

Marine Restoration Ecology

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Many new scientific disciplines emerge as new theory or technological breakthroughs lead to fundamental advances and demand new lines of inquiry, whereas others emerge out of pressing human need. This dichotomy in evolution often reflects the somewhat arbitrary labeling of “basic” and “applied” sciences. Few would argue against labeling ecology as a basic science and engineering as an applied field of inquiry. Modern marine restoration science borrows its theoretical constructs from ecology and shares its mandate to address pressing needs with engineering (Young et al. 2005). Few fields have grown as fast as restoration ecology over the last two decades. Characterizations of the field in the late 1980s and early 1990s as “a sort of gardening with wild species in a natural mosaic” (Allen and Hoekstra 1992) and an “expensive self indulgence” (Kirby 1994) reflect the early development of restoration ecology and the need for a conceptual and theoretical framework for the field. Two decades after this need was espoused by Hobbs and Norton (1996), restoration ecology has borrowed much from conservation biology and ecology to move beyond ad hoc pursuits of site-specific goals (Hobbs and Norton 1996) to a science that includes broader ecosystem (Bullock et al. 2011; Trabucchi et al. 2012) and socioeconomic contexts (Aronson et al. 2010). Restoration ecology in marine ecosystems has followed this same path of development with decades of species- and site-specific efforts that have only recently matured into landscape- and ecosystem-level endeavors (Peterson and Lipcius 2003).

Although much has been learned over the last 20 years, marine restoration ecologists are facing many pressing challenges from a legacy of environmental degradation (Jackson et al. 2001; Myers and Worm 2003) while bracing for unprecedented changes (e.g., sea-level rise, increased storm frequency, ocean acidification) over the coming decades. In this chapter we review and synthesize past restoration efforts in the marine environment and offer direction for meeting future challenges. Reflecting the evolution of marine restoration science, this chapter proceeds from efforts to restore individual populations to habitat-based initiatives to landscape- and ecosystem-level restoration attempts. Throughout the chapter we stress the necessary connection between ecological process and knowledge, the importance of experimentation, and the development of sound restoration strategies.

What Is Restoration?

The term “restoration” is used frequently in the literature to refer to a variety of human interventions intended to produce a positive biological response. This intent-based characterization incorporates numerous activities, many of which would not be considered restoration if any historical reference point were used as part of the definition. The

definition preferred by most funding agencies (explicitly or implicitly) is that advanced by the National Research Council (NRC, 1992): “returning a system to a close approximation of its condition prior to disturbance, with both the structure and function of the system recreated.” This definition’s focus on repairing some damage is narrower than the concept proposed by Hobbs and Norton (1996), that restoration represents one end of a continuum of conservation biology. Within this latter context, restoration ecology can be perceived as a strategy to conserve biological diversity (Jordan et al. 1988) and ecosystem integrity (Cairns and Heckman 1996). When intervention is necessary to achieve conservation goals and the intervention strategy is developed from an ecological perspective, the process, evolution, and fate of this intervention can be considered restoration ecology. Such interventions encompass a wide variety of activities in marine ecosystems with a broad range of financial and logistical costs: ceasing damaging activity (e.g., bottom trawling of live or relict reef habitats: Watling and Norse 1998); adopting mitigative strategies (e.g., installing permanent moorings for boat anchoring near coral reefs: Harriott et al. 1997); rebuilding habitats (e.g., oyster reefs: Lenihan 1999), or restoring the

hydrology of complete watersheds (e.g., Florida Everglades: Sklar et al. 2005).

The scope of intervention necessary for restoration varies as a function of the bottleneck that currently limits populations or habitats, the ultimate target or goal, and the societal value placed on the resource to be restored. In many cases, the bottleneck is not known, the goal is poorly defined (see Choi 2004), and the societal value a function of public education. Marine ecologists have played, and can play, critical roles in marine restoration initiatives by addressing the current bottleneck(s) (Figure 22.1), identifying positive effects that promote reestablishment of species (Figure 22.2), and working in a more public setting to communicate the ecosystem benefits of restoration to society. The increasingly abundant marine conservation literature and data (e.g., Jackson et al. 2001; Myers and Worm 2003; Pandolfi et al. 2003; Halpern et al. 2007) may be used to set goals and targets. Such goals and targets are generally set within a historical context (i.e., X percent of population or habitat levels recorded 20, 30, 100, or 200 years ago). The efficacy of such historical benchmarks in developing goals for restoration has been questioned by several investigators on the grounds that such static targets fail to acknowledge the uncertainty of historical reconstructions, the dynamic nature of ecosystems, losses of keystone species, establishment of invasive species, the added effects of climate change, or the high cost and logistical difficulty of many of these goals (Davis 2000; Choi 2004, 2007). In addition, historical fidelity can create unattainable benchmarks for success (Higgs 2003; Halvorson 2004; Choi 2007).

Although stringent application of historical benchmarks is rarely advisable, quantitative estimates of the extent and quality of past ecosystems (e.g., zu Ermgassen et al. 2012) can be very useful in developing restoration plans. Unfortunately, the detailed records needed to fully reconstruct past environments are limited to a few well-studied systems or global assessments (Orth et al. 2006; Beck et al. 2011). Still, the lack of site-specific historical information should not deter restoration when clear global declines have been identified and anecdotal historical evidence provides an understanding of the past. Such knowledge can be supplemented with examination of present-day reference sites that have undergone limited perturbation (e.g., Morgan and Short 2002; Steyer et al. 2003; Wigand et al. 2010). Increasingly, the delivery of specific ecosystem benefits has been set as an explicit goal in restoration plans. In measuring progress toward these goals, restoration projects are proving useful in elucidating the “production functions” of habitats ranging from denitrification to fish production (Grabowski and Peterson 2007).

The danger in ignoring historical or present-day reference points is that many activities that are characterized as restoration may have little relevance in a conservation framework. Marine resource agencies have a long history of conducting resource enhancement activities under

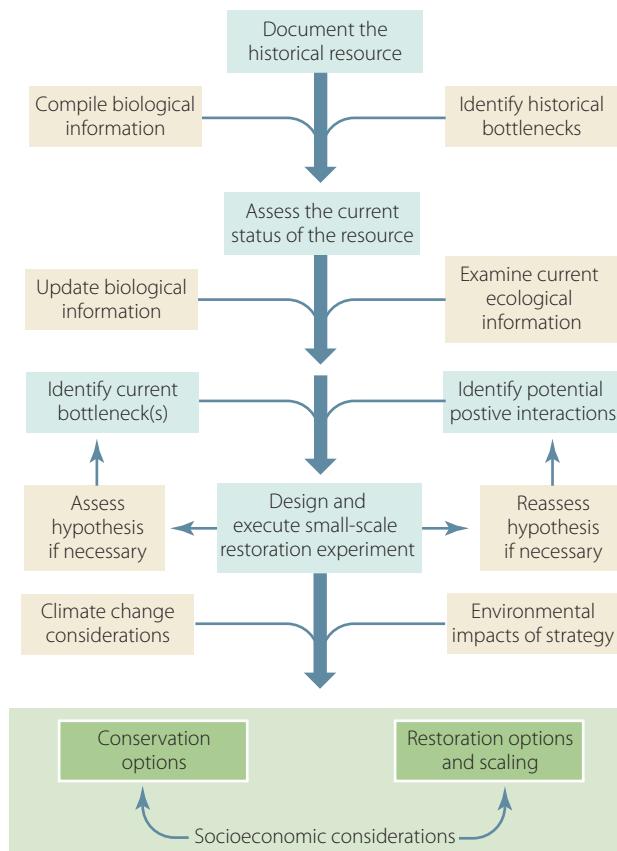


Figure 22.1 Conceptual diagram of the stages in the development and testing of a marine population-level or habitat-level restoration plan.

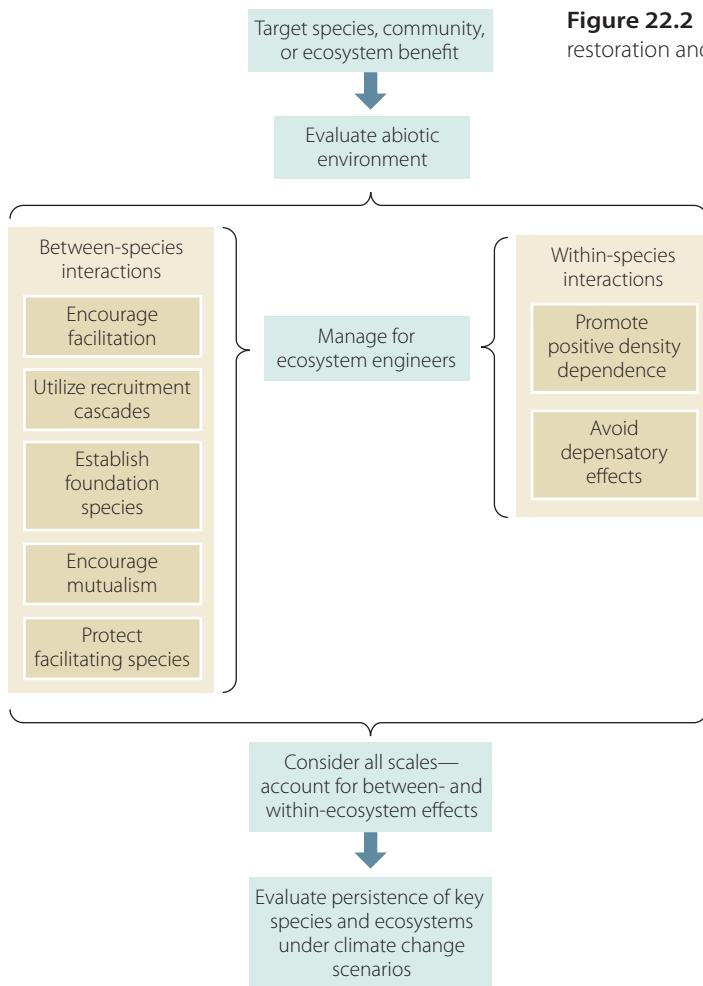


Figure 22.2 Conceptual diagram illustrating incorporation of positive effects in restoration and conservation strategies. (After Halpern et al. 2007.)

tal conditions. Environmental conditions for marine ecosystems will change, and the ability to predict these changes has increased dramatically over the last two decades. Comparisons of current conditions with those of past and future decades are now possible and should be made when assessing the feasibility of restoring species or habitats. Range shifts of several species have been documented as a result of increases in temperatures (Perry et al. 2005; Fodrie et al. 2010). These shifts are mainly poleward and may increase local biological diversity in the short term, but endemic species may eventually be lost as a result of their eventual poleward retreat (Hickling et al. 2006). Even if species are not lost, local predatory and competitive interactions may be significantly modified by the inclusion of additional species, similar to the effects documented for invasive species. Sea level is also expected to rise over the coming century (0.5 to 1.4 meters above the 1990 level: Rahmstorf 2007), which may reduce the sustainability of restoration projects. “Renourishment” of sandy beaches, for example, is a common practice on the U.S. Atlantic and Gulf coasts and is considered a restoration activity by many marine resource agencies. Sea-level rise, as well as increased severity of tropical cyclones (Webster et al. 2005), would render these activities ineffective over longer time scales and hence unsustainable. Within estuaries, sea-level rise will move salt water into regions that are currently brackish, leading to shifts in species distributions. Regional planning for restoration and conservation will need to consider how these changes in salinity distribution will affect acreages as well as proportions of different habitat types. Ecosystems have changed, and will change, in response to climate change (McCarty 2001), and hence restoration projects must maintain an appreciation for the past but be responsive to current and future conditions (Choi 2007).

the guise of restoration. Manipulation of populations or habitats for the primary purpose of increasing human exploitation is not restoration, although such activities may be sustainable and may be endorsed by numerous stakeholders. For example, while increased harvest of oysters is a valid fisheries management objective, it is at odds with other objectives of oyster reef restoration, such as shoreline protection and provision of fish habitat. Limiting the definition of restoration to reestablishment of a species or habitat in order to replace lost ecosystem function would eliminate the debatable inclusion of many fishery enhancement activities under the term. Larger landscape-based approaches, which can encompass a more complex set of management goals, could allow for both restoration and fisheries enhancement activities (e.g., habitat restoration in protected areas that serve as spawning sanctuaries may enhance recruitment on harvestable oyster reefs), but require a broader perspective than many current restoration initiatives take.

Once historical and present-day reference conditions are determined and a conservation framework established, the next priority is to determine whether restorations are sustainable under current and future environmen-

Population-Level Approaches

Population-level “restorations” of exploited marine and anadromous species have been attempted by marine scientists and resource agencies with mixed results. Many of these activities have been performed for the explicit purpose of short-term (2–3 years) increases in exploitation level, while others have been performed within the context of longer-term goals. Even if fisheries enhancement was one anticipated benefit, these longer-term efforts may in fact be considered in a restoration context depending on the restoration approach adopted.

A variety of approaches are available for population-level restorations (**Table 22.1**). Decreasing fishing mortality

TABLE 22.1 Common restoration options and rationale for population-level restoration/enhancement activities

RESTORATION APPROACH	POTENTIAL INTERVENTIONS	BOTTLENECK ASSUMED	POTENTIAL BENEFITS	POTENTIAL NEGATIVE EFFECT
Restrict harvest	Reduce fishing mortality via regulations	Larval supply	Increase spawning stock biomass	Economic losses to fishermen
Create spawning sanctuaries	Set aside no-take areas for target species	Larval supply	Protection of habitats by restricting harvest; potential disease resistance	Minimal
Introduce reared larvae	Hatchery production of larvae released into the environment	Larval supply; reduction of early postsettlement mortality	Increase local population abundance	Adverse population genetic effects; potential disease introductions
Relocate and concentrate juveniles and adults	Harvest target species from donor site and place at transplant site	Larval supply; reduction of early postsettlement mortality	Increase local population abundance	Potential disease introductions
Habitat conservation and restoration	Protect or reestablish marine habitats	Reduction of postsettlement mortality	Protection of nursery habitats	Minimal

is the most commonly accepted conservation measure to rebuild exploited stocks of fishery species. If stocks have not decreased to levels at which depensation is expected to occur (e.g., densities too low for high fertilization success of sessile invertebrates, difficulty in finding mates for more mobile fishes), then decreases in fishing mortality may result in rebuilding of exploited populations. When decreases in fishing mortality are not possible (either because of public resistance or because the species in question is not directly exploited), other mechanisms to increase population biomass must be used. Critical to the success of such endeavors is identification of the current population bottleneck (see Figure 22.1). If the bottleneck is unknown, then resources should first be devoted to resolving this issue. In these cases, experiments should be used to elucidate the bottleneck (e.g., Johnson et al. 2010), particularly when competing hypotheses emerge that are equally plausible (**Figure 22.3**).

In many cases, an implicit assumption of past restoration or enhancement strategies has been that recruitment limits the population. For species with planktonic larvae, recruitment encompasses two distinct components: supply of larvae (a function of horizontal advection and concentration of larvae) and postsettlement mortality. The relative importance of pre- versus postsettlement processes was the subject of intense debate among marine ecologists (Caley et al. 1996) in the 1980s and 1990s. The consensus that emerged from this debate supports dominance of postsettlement or density-dependent processes in most marine systems. Recruitment limitation is possible, but is most likely a temporary condition caused by extremely high exploitation rates for fisheries species or by episodic disturbances (e.g., red tide events: Peterson et al. 1996). This consensus has important implications for

marine restoration ecology, but it has been largely ignored by advocates of hatchery-based approaches to population-level enhancement and restoration. A substantive burden of proof should be met prior to engaging in restoration that relies on introduction of larvae. For endangered or threatened species, this burden of proof may be met, and there is support for the conclusion that hatcheries have played an important role in sustaining highly endangered salmonid populations (Naish et al. 2007). The most common marine species reared under hatchery conditions and released for enhancement or restoration purposes have been bivalve molluscs and marine teleosts. Few of these species have been demonstrated to be recruitment limited (bay scallops: Peterson et al. 2006) or to suffer from dramatically lowered spawning stock biomass (but see hard clams, *Mercenaria mercenaria*: Peterson 2002; blue crabs, *Callinectes sapidus*: Lipcius and Stockhausen 2002)—evidence that is critical to have before advancing options designed to increase larval supply.

Spawning sanctuaries are also promoted as an effective tool when larval supply is suspected as a bottleneck to population recovery. Unlike hatchery approaches, which may increase postsettlement survival for the period of hatchery grow-out, spawning sanctuaries address only larval supply. For sessile and less motile benthic invertebrates, concentrating adults within areas may increase fertilization success (Doall et al. 2008; Tettelbach et al. 2011) and hence the supply of larvae. Protecting spawning stock from harvest (Lambert et al. 2006) or predators (Fegley et al. 2010) would be expected to provide similar benefits by increasing the density of the targeted species, provided such protection is adequate (see Eggleston et al. 2009). Studies of the efficacy of spawner sanctuaries have primarily addressed the survival (Lambert et al. 2006; Tettelbach et al. 2011) and gonad

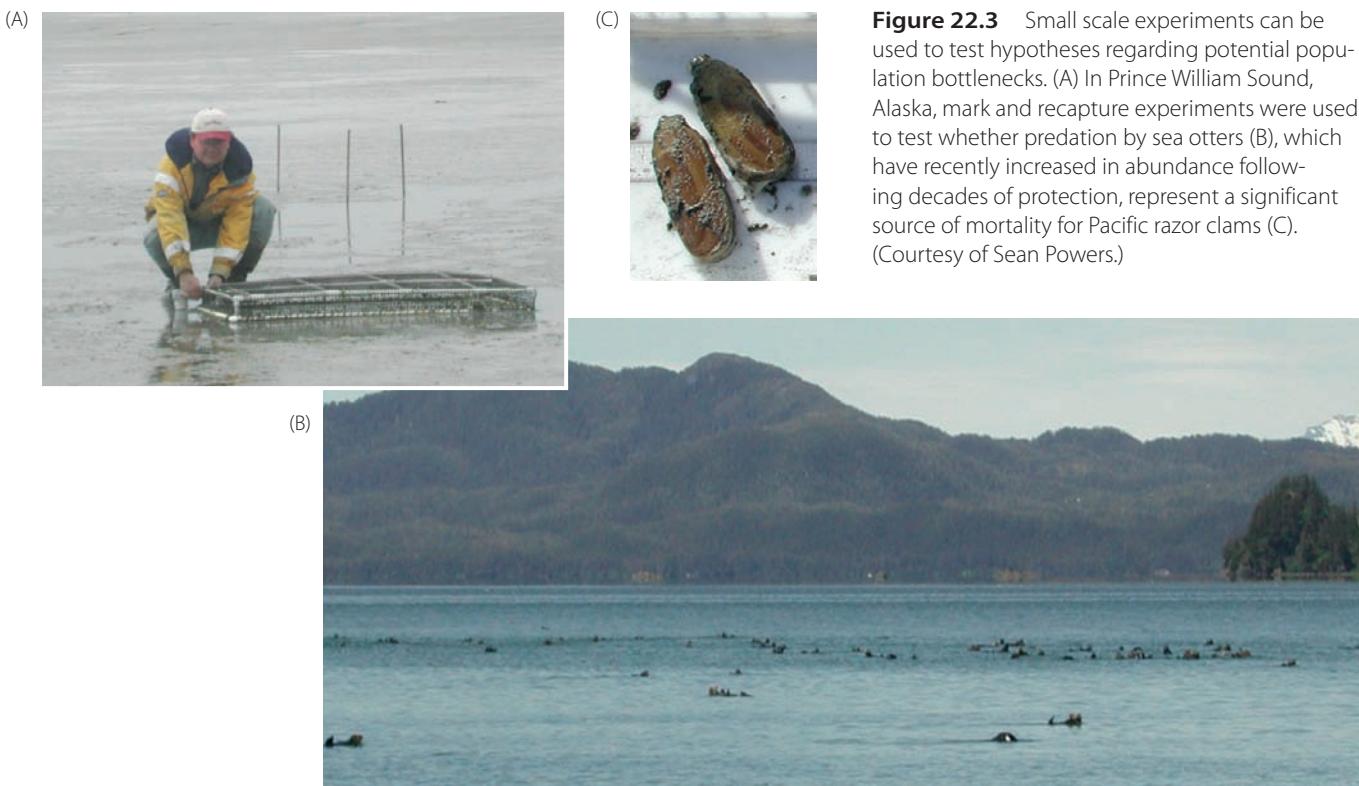


Figure 22.3 Small scale experiments can be used to test hypotheses regarding potential population bottlenecks. (A) In Prince William Sound, Alaska, mark and recapture experiments were used to test whether predation by sea otters (B), which have recently increased in abundance following decades of protection, represent a significant source of mortality for Pacific razor clams (C). (Courtesy of Sean Powers.)

condition of the transplanted adults (Doall et al. 2008), but have not established a linkage between the transplanted spawners and increased recruitment inside or outside the spawning sanctuary. Peterson and colleagues (1996) documented basin-wide increases in recruitment following transplantation of bay scallops into Bouge Sound, North Carolina; however, the linkage between the transplanted spawners and increased recruitment was correlative. Planes and colleagues (2009), using DNA markers, provided one of the few demonstrations of linkages between a no-take area and recruitment outside the reserve area. Advances in genetic tagging (Palumbi 2003; Planes et al. 2009) and the use of elemental chemicals as fingerprints (Becker et al. 2005), as well as traditional larval marking techniques (Jones et al. 1999), could be used to establish more concrete linkages between spawners and recruits; such evidence is critical in examining the efficacy of spawner sanctuaries as a restoration tool.

In the absence of such linkages, several investigators have used knowledge of local water circulation or hydrodynamic transport models to establish areas of potentially high larval retention. Given that documentation of low larval supply is usually limited to local scales, such evidence can be extremely useful in directing placement of spawning sanctuaries. These approaches are further supported by emerging evidence of “self-recruitment” in some marine species (Jones et al. 1999; James et al. 2002). In nearshore areas, water in embayments may have sufficiently long residence times to promote localized increases

in larval supply (Schulte et al. 2009; Kim et al. 2013). In more open-water settings, Sponaugle and colleagues (2002) have suggested that areas where water flows depart from unidirectional, depth-uniform flows provide the highest potential opportunity for retention of larvae. Further demonstration of the linkages between spawner sanctuaries and regional recruitment benefit will probably be documented in the upcoming years; however, it is still important to note that the soundness of spawning sanctuaries as a restoration strategy depends on the assumption that larval supply, not postsettlement mortality, limits the population. This observation should not discourage the establishment of no-take zones or marine protected areas, as the benefits of restricting the harvest and degradation of large areas of the ocean extend well beyond the singular benefit of a spawning sanctuary to one population.

Habitat-Level Approaches

Like population-level restoration strategies, habitat-level restoration plans focus on one or two species; however, the targeted species in this case are known to form habitat for other species (i.e., ecosystem engineers that then act as foundation species) and hence can fundamentally change the ecosystem (**Figure 22.4**). The recovery of many species of fish and invertebrates may be limited by bottlenecks caused by the loss and degradation of marine biogenic habitats over the last two centuries (Jackson et al. 2001; Aioldi and Beck 2007). Declines and degradation of salt

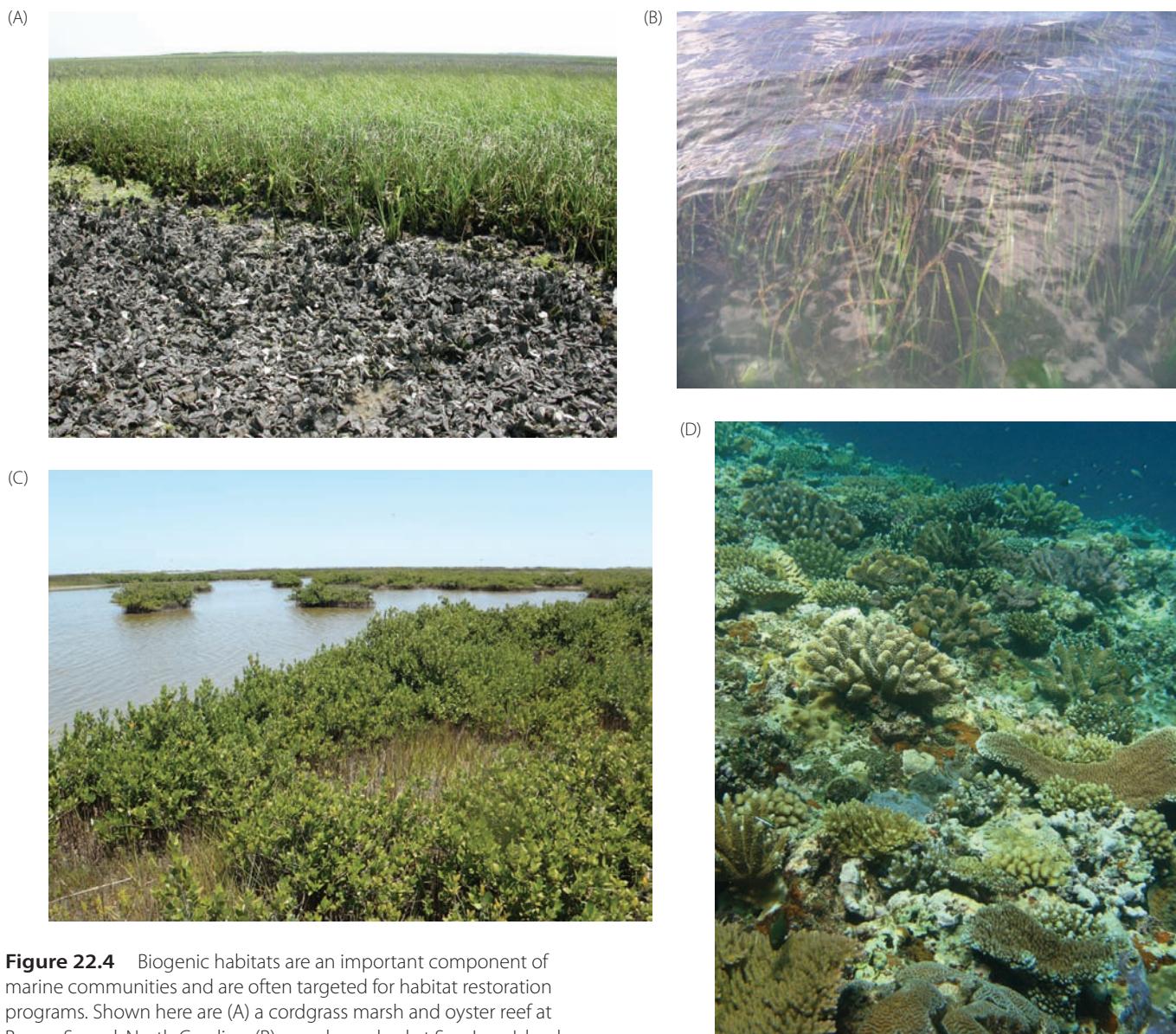


Figure 22.4 Biogenic habitats are an important component of marine communities and are often targeted for habitat restoration programs. Shown here are (A) a cordgrass marsh and oyster reef at Bogue Sound, North Carolina, (B) an eelgrass bed at San Juan Island, Washington, (C) mangroves in the Chandeleur Islands, Louisiana, and (D) a coral reef in Fiji. (A,B courtesy of Katharyn Boyer; C courtesy of Kenneth Heck, Jr.; D courtesy of Isabelle Côté.)

marshes (Adam 2002), seagrass meadows (Short and Wyllie-Echeverria 1996; Orth et al. 2006), oyster reefs, (Beck et al. 2011; zu Ermgassen et al. 2012), coral reefs (Jackson 1997; Pandolfi et al. 2003), and mangrove forests (Alongi 2002) are a major global conservation crisis. The need for suitable settlement substrate for sessile marine invertebrates and the high postsettlement mortality of many juvenile fish and invertebrates in the absence of refuge are well established in the marine ecological literature (Heck et al. 2003; Powers et al. 2009; Hixon 2011). In addition to serving as habitat for juvenile fish and invertebrates, healthy biogenic habitats can fundamentally change benthic–pelagic coupling, nutrient dynamics, predator–prey

dynamics, and competitive interactions (Grabowski 2004; Grabowski and Powers 2004; Hixon 2011). Consequently, habitat-level restorations have the capacity to benefit multiple species, multiple trophic levels, energy flows, and ecosystem function.

The vast majority of habitat-level restoration efforts have focused on reestablishment of nearshore and estuarine foundation species, such as seagrasses, salt marsh grasses, oysters, mangroves, and kelp. These species are an integral component of coastal and nearshore ecosystems, serving as essential nursery and feeding grounds for numerous marine, avian, and terrestrial species. These highly productive habitats are characterized by high densities of invertebrate prey, which in turn provide feeding opportunities for fish, crabs, shorebirds, waterfowl, and mammals. Because many nearshore habitats are intertidal, these communities are open to consumers from a

variety of adjacent habitats, including marine, terrestrial, and aerial predators (Leigh et al. 1987). With the recognition of the important role that structured nearshore habitats play in coastal food webs, efforts to restore them have increased dramatically over the last decades. The rationale most often cited for these restorations has been the role of these ecosystems as nursery and foraging habitats for birds, fish, and invertebrates; to a lesser extent, other equally important ecosystem functions (e.g., nutrient cycling) have been cited as well. Expectations of increased fish or invertebrate production have rarely been quantified (but see Peterson et al. 2003; Powers et al. 2003), and the population-level consequences of such habitat restorations have rarely been empirically tested beyond simple documentation of higher densities of animals in structured habitats.

Decades of efforts to restore nearshore habitats have revealed several points of commonality. First, conservation and rehabilitation of existing habitats is the most cost-effective approach to ensuring that the ecosystem functions of these habitats are preserved. Second, the design and construction of heavily engineered efforts and structures has stressed persistence, with restoration of ecological function often a distant secondary concern and the potential for adverse effects high. Third, few restoration projects have included specific and measurable performance criteria for judging their “success,” and even fewer have included adequate monitoring to measure the performance of restored habitats. Fourth, although habitat restoration efforts should have historical justification, they must also be designed and prioritized under the predictions of near-term and long-term climate change. Finally, in our review of marine ecology in the context of restoration, we have identified an increasingly common finding of positive interactions among species. These positive interactions should be used in restoration settings to enhance establishment and persistence of foundational habitats and associated species (Halpern et al. 2007; see Figure 22.2).

Salt marshes

Restoration of salt marshes has perhaps the longest history of any marine habitat restoration effort and illustrates many of the notable lessons from past restorations. Coastal wetlands are a major component of the landscape of most estuaries, and their loss as a result of human development of coastal areas as well as large-scale changes in hydrology has been well established (Gedan and Silliman 2009a; Teal and Peterson 2009). Restoration of salt marshes may require reengineering of system hydrology as well as establishment of halophytic vegetation via seeding, transplants from donor sites, or natural recruitment. A limited number of studies of restored salt marshes and natural reference areas have been conducted so as to allow evaluation of the time course over which ecological function is restored (**Figure 22.5**), although the number of such studies should increase with the development of comprehensive monitoring protocols (e.g., Neckles et al. 2002). It is clear that some abiotic conditions strongly affecting the functioning of natural marshes can develop rather quickly (5–15 years), while others can take decades, if not longer, to become established in restoration sites (e.g., organic carbon and nitrogen in soils; Craft et al. 2003). Further, plant species richness can be as much as 50% lower in restoration sites than in prehistoric remnant marshes, even decades after restoration was initiated (Boyer and Thornton 2012). Some measures of success will simply take time to achieve, while others may require specific interventions that move a restoration along a desired trajectory. As much as possible, we recommend that ecological understanding, informed by experimentation, be used when determining the form of such interventions (e.g., Palmer 2009).

A number of ecological concepts and aspects of general theory have been, or could be, applied in salt marsh restoration settings to manipulate species composition or functions toward restoration goals. There is increasing recognition that positive species interactions in salt marshes could be beneficial to the restoration of targeted species

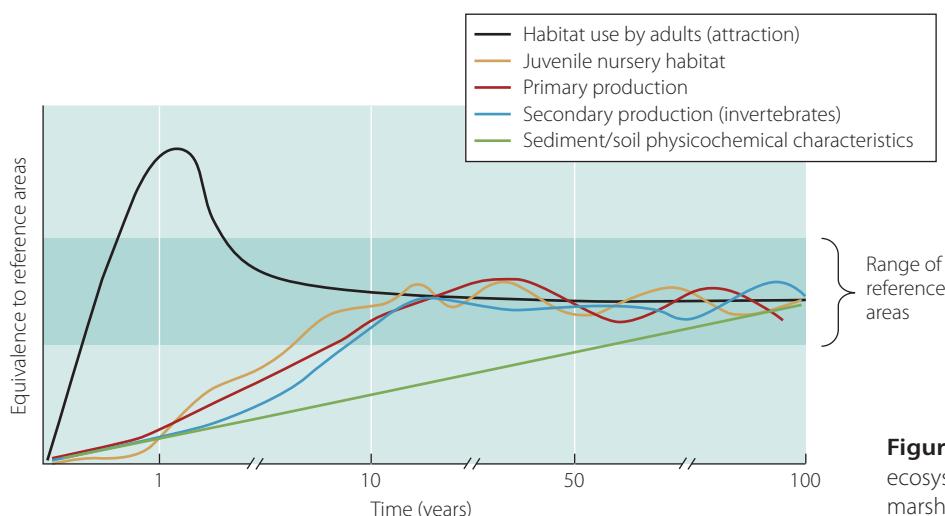


Figure 22.5 Generalized response time for ecosystem services following restoration of salt marsh habitats.

assemblages and functions. Species that ameliorate harsh physical conditions can affect salt marsh community composition, and knowledge of these interactions could be exploited in restoration settings. For example, in New England, the black rush *Juncus gerardii* shades soil surfaces and reduces salinities, permitting the marsh elder *Iva frutescens* to extend its range from terrestrial borders down to mid-marsh elevations (Bertness and Hacker 1994). Studies in the same region showed that the recovery of saline bare patches produced by wrack can depend on less common species that are able to colonize and reduce salinities (Bertness et al. 1992). Some uncommon species are able to concentrate nitrogen in their tissues, which could lead to local reductions in coverage of dominant species and enhanced species richness (Sullivan et al. 2007). Further, species that concentrate nitrogen may later release it, benefiting other species that differ in timing of N demand (Morzaria-Luna 2005). Evaluation of the applicability of positive interactions to the establishment and maintenance of restored marshes will rely on experimental tests in restoration settings as well as consideration of climate and abiotic stress, which can determine the strength of facilitative and other interactions (Bertness and Ewanchuk 2002).

Establishing native species diversity is a common goal in salt marsh restoration, in part to match historical or present-day reference conditions, but also because experimental evidence across many marine and terrestrial habitats supports the idea that species richness has a positive effect on productivity or other measures of ecosystem functioning (Balvanera et al. 2006; Cardinale et al. 2006). Further, species with different functional roles or responses to disturbance may help to ensure community persistence (Naeem 1998; Yachi and Loreau 1999; Elmquist et al. 2003). Wetland plants have been shown to differ in the types and magnitudes of their functions, so maximizing richness may maximize the range of functions in a plant assemblage even if it does not increase plant productivity (e.g., Engelhardt and Richie 2002). In a southern California salt marsh restoration experiment, increasing species richness led to greater canopy complexity (Keer and Zedler 2002) as well as with biomass production and nutrient retention (Callaway et al. 2003); further, richness maximized various attributes of structure and function (Sullivan et al. 2007). In addition, higher richness at the level of primary producers can increase the abundance, richness, and stability of assemblages at higher trophic levels (e.g., Haddad et al. 2011); for example, in Tomales Bay (northern California), Traut (2005) showed that plant species richness was correlated with richness of spider species. Genetic variation within a species can also provide important ecosystem functions, including increased resilience to disturbance (e.g., Hughes and Stachowicz 2004), and may be important to establishing species in restoration sites that present novel conditions (Lesica and Allendorf 1999).

Complexity of physical features can lead to increased species richness by providing more physical space or re-

sources (e.g., MacArthur 1970). The greater complexity of sediment textures, creek networks, pools, and microtopographic features found in prehistoric salt marsh remnants and known from historical records may encourage a greater number of species by supporting a wide range of habitat requirements (Boyer and Thornton 2012). Marshes that have deposits of coarse sediments due to connections to fluvial sources or overwash of barrier beaches of sand or shell can provide conditions most suitable to some rarer plant species (Baye et al. 2000). Channels, creeks, and pools are associated with greater plant species richness than adjacent plains (Zedler et al. 1999; Sanderson et al. 2001), and may increase habitat for fish species (Larkin et al. 2008). Microtopographic variation across marsh surfaces can also enhance the diversity of the plant community (Vivian-Smith 1997; Morzaria-Luna et al. 2004). In view of these potential benefits, maximizing topographic variation should be a routine consideration in marsh restoration projects. In addition, diverse plantings may increase opportunity for less common species to be present in marsh restoration sites (Armitage et al. 2006; O'Brien and Zedler 2006).

While harsh physical conditions in salt marshes reduce the number of potential invaders, several non-native species, invasive genotypes, and hybrids have established in marshes and modified their environments extensively. These species represent significant threats to present and future restoration projects. In San Francisco Bay, hybridization of the introduced cordgrass *Spartina alterniflora* with the native *Spartina foliosa* produces plants with taller stems and greater biomass over a larger tidal range than the native *S. foliosa*; the hybrid also supports significantly lower densities of infaunal invertebrates and shifts trophic relationships (Brusati and Grosholz 2006; Levin et al. 2006). Invaders of estuarine salt marshes may reinforce their persistence and spread through positive feedbacks such as sediment accretion and pollen swamping (*Spartina* hybrids: Neira et al. 2006; Sloop et al. 2005), and legacy effects of altered sediment conditions after they are removed may further reduce the potential to restore native species (e.g., thick root mats after removal of aboveground tissues of *Spartina* hybrids). A number of invaders outcompete natives for space (e.g., perennial pepperweed, *Lepidium latifolium*: Boyer and Burdick 2010), which could be particularly detrimental when the natives are rare species, as in the case of the invasive *L. latifolium*, and the very rare *Chloropyron molle* subsp. *molle* (soft bird's beak) and *Cirsium hydrophilum* var. *hydrophilum* (Suisun thistle) in the San Francisco Estuary (Grewell et al. 2003; Schneider 2013). In New England, invasive genotypes of the reed *Phragmites australis* compete with native marsh plants at higher elevations (Bertness et al. 2002), perhaps in part through release of an allelopathic compound that persists after the invader is removed (Rudrappa et al. 2007). In many cases, there are no immediate plans or funds to control salt marsh invaders on a regional scale, and local control is an ongoing management challenge.

In recent decades, ecologists have identified the importance of top-down control on marsh plant production. Herbivores from snow geese to snails have been shown through observations and experimentation to produce major damage to the foundational marsh plants (e.g., Silliman et al. 2005; Henry and Jefferies 2009), and a reduction in predators has been implicated in runaway herbivore damage in some cases (Silliman and Bertness 2002). Restoration projects need to consider the local context of predator and herbivore abundance and the potential for grazing damage, especially during the vulnerable period of early establishment of tender new transplants or seedling recruits. For example, large efforts to restore *Spartina foliosa* in San Francisco Bay in 2012–2013 have incorporated fencing to deter Canada geese (California State Coastal Conservancy 2013). These grazers have few predators along the urbanized Bay shorelines, preferentially consume the native cordgrass over invasive hybrids (Grosholz 2010), and have been shown experimentally to decimate new restoration plantings in the absence of exclosures (Thorton 2013).

A number of climate change effects are likely to alter coastal marsh distributions and functions and thus are considerations in the restoration of these habitats. Sea-level rise in many coastal areas will affect salt marshes more than inland brackish or tidal freshwater marshes, as rates of vertical accretion in salt marshes are lowest and may prevent them from keeping pace with sea-level rise (Craft 2007). Salt marshes in some regions are predicted to decline by 20% or 45% by 2100, according to the minimum and maximum estimates for sea-level rise by the Intergovernmental Panel on Climate Change (IPCC; Craft et al. 2009). Actual losses will depend on local sediment supply, as marshes that receive high sediment loads are more likely to keep pace. This ability will relate in part to tidal range, as marshes with greater tidal range tend to have higher sediment delivery and are less reliant on organic matter accumulation than those with microtidal (<2 m) inundation (Stevenson and Kearney 2009). For restoration to be successful in an era of rapidly increasing sea level, measures and predictions of accretion rates will be critical to determining the engineering needed at a site for it to gradually accumulate local sediment, establish and maintain vegetation, and evolve creek networks (Williams and Orr 2002). It is possible that some historical marshes that were diked to permit farming, solar salt evaporation, or other uses could simply have their hydrological connection to tidal flows restored, and that sediment accretion will be adequate for their elevations to rise along with rising seas. In other locations with low predicted sediment supply or where marsh surfaces have subsided (e.g., due to regional groundwater pumping or decomposition of organic matter through drying and years of farming; Teal and Weishar 2005), estuarine marshes may require sediment placement to establish appropriate elevations. In all cases, in the face of sea-level rise, long-term persistence

of coastal marshes will depend on their ability to increase in elevation over time or to migrate landward as inundation increases. In urbanized estuaries, there may be few marshes or potential marsh restoration sites that are free of upslope development or other barriers to marsh migration, and such considerations are increasingly contributing to restoration planning.

Other factors changing along with rising seas include CO₂, temperature, and salinity, all of which affect the distribution and production of marsh plants and associated animal communities (see Chapter 11). Increasing CO₂ should enhance production and water use efficiency, especially by C₃ vegetation, which could alter patterns in community composition (Körner 2006). A general shift in salt-tolerant species up estuaries is expected to occur as salinity rises with sea level (e.g., Watson and Byrne 2009). Further, in addition to climate-related changes, nutrient pollution can affect species dominance patterns and the vertical distribution of plant species (Bertness et al. 2002; Ryan and Boyer 2012). The ability to photosynthesize with high water use efficiency (important in the drought-like conditions of a salt marsh) is highly variable among plants, and C₄ plants use considerably less N for photosynthesis (5–9% of total leaf N) than most C₃ plants (20–30%) and thus can allocate more for growth (Makino et al. 2003). Given the variety of mechanisms influencing plant responses and the presence of both C₃ and C₄ species in most salt marshes, predicting patterns in plant richness and dominance under these simultaneous changes in multiple important abiotic forcing factors is a challenge in marsh conservation and in planning for restored marshes to support particular species assemblages.

Seagrasses

Following decades of degradation and loss, seagrass beds have been declining at an accelerated rate over the last 20 years in many parts of the world (Waycott et al. 2009). Restoration of seagrasses has been attempted in many regions worldwide in an effort to reverse this trend. Overall, there has been limited success in establishing seagrasses that persist over time (i.e., for more than 5 years in Chesapeake Bay: Orth et al. 2010). In a 2010 workshop to share seagrass restoration experience among European researchers and managers, organizers concluded that success in most projects was very low and that monitoring periods were too short (<1 year) to confidently claim success in many others (Cunha et al. 2012). Despite these difficulties, some seagrass restoration programs have been very successful; for example, following 11 years of eelgrass (*Zostera marina*) seeding efforts in the Virginia coastal bays, where disease previously (1930s) led to a massive die-off of the plant, eelgrass now covers 1700 hectares (Orth et al. 2012). Such successes provide reassurance that with careful planning, experimentation, and a phased approach that scales up following smaller successes, additional larger efforts may be warranted to offset previous losses and to increase

the numerous ecosystem functions and services that seagrasses provide (Orth et al. 2006; McGlathery et al. 2007; Hughes et al. 2009).

In general, seagrass restoration should be attempted only in places where recruitment is thought to be limited or where the factor or factors leading to a previous loss of seagrasses (e.g., eutrophication, shading, disease) have been eliminated or greatly reduced (Meehan and West 2002). A paucity of information on historical distributions can make site selection for seagrass restoration a difficult step. Guidelines for determining appropriate restoration sites have emerged in several regions (e.g., New England: Short et al. 2002b), with much of the advice applicable elsewhere. In addition, most scientists involved in seagrass restoration have taken a cautious approach that includes multiple steps toward the understanding of restoration constraints. For example, it has become standard practice to use replicate small-scale (<1 to a few square meters; e.g., Short et al. 2002a) plots of seagrass plantings over multiple years to test the suitability of a site for restoration before deciding whether to scale up to planting or seeding of larger areas.

Several scientists have noted that seagrass restoration efforts have not capitalized on past work, although much of it is unpublished (especially the failures) and difficult to access (Fonseca 2011; Cunha et al. 2012). This situation is changing, however, with restoration programs in new regions calling on the experience of other programs through workshops and other forums (e.g., San Francisco Bay in 2006; the European workshop in 2010). Further, a growing body of literature documenting experiences and testing techniques for establishing seagrasses (e.g., Paling et al. 2001; Short et al. 2002a; Pickerell et al. 2005; Marion and Orth 2010) is helping to keep restoration efforts from “reinventing the wheel.” Still, different coastal regions of the world present site-specific challenges that require site-specific investigations. For example, in some areas, herbivory and bioturbation have emerged as substantial impediments to restoration, but the animal species involved and the types and magnitudes of effects vary among regions. Recent studies have documented effects ranging from bioturbating invertebrates burying seeds and seedlings (Valdesmarsen et al. 2011) to outbreaks of invasive amphipods consuming flowering and vegetative shoots (Reynolds et al. 2012a) to Canada geese decimating donors or transplants (Rivers and Short 2007; Kiriakopolos 2013). As in salt marshes, the effects of animals and their local abundances are increasingly recognized as important considerations in the restoration of seagrass habitat structure.

A better understanding of the roles of genetic variation in seagrasses in recent years has applications to restoration methods and outcomes. Efforts to maximize genetic diversity, such as collecting transplants for restoration over large areas of a donor bed, are expected to increase the success of seagrass restoration efforts (Williams 2001; Procaccini and Piazzi 2001). Increased genotypic diversity,

as tested by inclusion of multiple genotypes in experimental plots, has been found to enhance resilience to disturbances such as high temperatures and brant goose herbivory (Hughes and Stachowicz 2004; Reusch et al. 2005). This idea was recently applied in a restoration experiment in the coastal bays of Virginia, in which knowledge of genetic variation was used to select seed sources; more diverse seeding combinations increased valued ecosystem functions, including biomass production and invertebrate support (Reynolds et al. 2012b). Because seeds provide enhanced diversity via sexual recombination, they are inherently a greater source of variation than vegetative shoot transplants. In locations where seagrasses flower extensively, seeding can also be the most effective method for establishing seagrasses (e.g., in Chesapeake Bay: Marion and Orth 2010) while also maximizing genetic diversity among recruits. In other locations seeding is used to supplement whole-shoot transplants to increase the potential for genetic diversity in the restored populations (Long Island Sound, New York: C. Pickerell, pers. comm.). In estuaries where seagrass beds exhibit significant population genetic structure (e.g., San Francisco Bay: Ort et al. 2012), it may be advisable to use multiple populations as donors in a single restoration site in order to improve the probability of including genotypes suitable for that site. However, information about the genetics of the potential donors and the habitats to be restored should inform choices of specific donors or donor mixes for any project.

The effects of climate change on seagrasses and their restoration will also vary regionally. Increased temperatures may enhance seagrass growth in some regions and be very detrimental in others. Sustained high temperatures (>30°C) in July–August 2005 in Chesapeake Bay are believed to have been largely responsible for a massive, nearly complete dieback of *Zostera marina* (Moore and Jarvis 2008). On the other hand, temperatures up to 35°C for 4 to 5 hours at a time produce no noticeable damage in New Hampshire eelgrass beds (F. Short, University of New Hampshire, pers. comm.). As for the effects of increased CO₂ concentrations, *Z. marina* showed higher growth rates under short-term experimental CO₂ enrichment (Thom 1996), but longer-term effects are less clear (Palacios and Zimmerman 2007). As with salt marshes, the ability of seagrass beds to keep pace with rising sea levels will depend on local and estuary-wide sediment supply, and increasing salinities should shift seagrass distributions up estuaries. These changes will need to be considered in planning for restoration in particular regions and locations within estuarine and coastal zones.

Oyster reefs

Like salt marshes and seagrass beds, oyster reefs provide complex three-dimensional habitat, but unlike these habitats, oyster reefs are targeted as a fishery. The relatively recent addition of reefs formed by the gregarious settlement of bivalves (e.g., mussel beds, oyster reefs) to the

marine and estuarine habitats that are actively restored for their ecological benefit is a function of the increasing recognition of the importance of dense assemblages of bivalves in estuarine ecosystems (Dame et al. 1984; Jackson et al. 2001). The eastern oyster (*Crassostrea virginica*) is a key component of coastal ecosystems and local economies along the Atlantic and Gulf coasts of the United States (Rothschild et al. 1994; Coen and Luckenbach 2000; Lenihan et al. 2001) and has the longest history of restoration. Recent efforts to restore the native Olympian oyster (*Ostrea lurida*) on the Pacific coast have also expanded for similar ecological benefits (Brumbaugh and Coen 2009; Dinnel et al. 2009; Zabin et al. 2010). Biogenic reefs formed by the aggregation of oysters and mussels provide three-dimensional habitat for numerous species of fish and invertebrates (Peterson et al. 2003; Coen et al. 2007). Oysters also filter estuarine waters and have probably exerted a high level of control on energy flow and nutrient fluxes (Dame et al. 1984, 1989) in estuarine ecosystems in the past (Jackson et al. 2001). Further, shallow subtidal and intertidal oyster reefs can facilitate the maintenance and expansion of emergent (Meyer et al. 1997; Meyer and Townsend 2000) and possibly submerged vegetation (Newell et al. 2002; Newell and Koch 2004) in estuarine systems. Nearshore oyster reefs may benefit emergent shoreline vegetation such as *Spartina patens* and *S. alterniflora* by reducing wave energy that would normally result in shoreline erosion (Meyer et al. 1997; Scyphers et al. 2011).

The dramatic decline in oyster populations throughout the world (Beck et al. 2011) has prompted increased efforts to restore oyster habitat (Figure 22.6). Much of this decline has resulted from a century-long legacy of overharvesting (Kirby 2004), destructive fishing practices (Rothschild et al. 1994; Lenihan and Micheli 2000; Lenihan and Peterson 2004), declines in water quality (Lenihan and Peterson 1998), and increased exposure to pathogens (Ford 1996). Many of these processes have led to bottlenecks for restoration. Chief among these bottlenecks is the removal of hard substrate via harvest. Oysters are gregarious settlers, and the absence of hard substrate, preferably bivalve shell, will limit any restoration. Thus all restorations involve placing substrate in the environment. In addition to the loss of hard substrate, several areas have been sufficiently depleted of oysters via harvest to create regionally low larval supplies, a bottleneck that is difficult to overcome if sustainability is the goal. Although bottlenecks that work on much larger scales, such as poor water quality, must be addressed by ecosystem-level restorations, design and siting of reefs can mitigate some of their effects. Lenihan (1999) demonstrated how increased vertical relief of oyster reefs can enhance oyster survival in the presence of low-oxygen bottom waters, increase the overall fitness of oysters, and increase recruitment by elevating the substrate above the bottom, where water flow, and hence larval supply, will be greater. The benefits of elevated vertical relief have also been demonstrated by Gregalis and colleagues

(2008) in Mobile Bay, Alabama, and by Schulte and colleagues (2009) in Chesapeake Bay, Virginia.

Historically, oyster reef enhancement and restoration efforts have focused primarily on fisheries enhancement (i.e., returning substrate to areas of high harvest pressure) or have been responses to major natural disasters (e.g., hurricanes). Although the economic loss of the oyster fishery is striking and has been largely responsible for public concern (Kirby 2004), the loss of ecological services provided by oyster reefs may have far greater effects on estuarine ecosystems (Jackson et al. 2001; Coen et al. 2007; Grabowski and Peterson 2007), and those services are increasingly used as the primary rationale for restoration. As in other nearshore habitats, few studies have tracked the long-term fate of oyster reef restoration efforts, quantified the anticipated benefits of restoration, or provided clear metrics to measure performance in terms of the ecosystem function of restored oyster reefs. Short-term studies (2–3 years) have demonstrated that these restored habitats attract a large transient fauna and provide refuge for recently settled invertebrates and fishes (Peterson et al. 2003; Grabowski and Peterson 2007). Using a synthesis of these observations, Peterson and colleagues (2003) concluded that 10 m² of oyster reefs would result in 2.6 kg per year of added fish production. Such scaling of restoration initiatives should form the basis for discrete hypothesis testing in the future and may be of significant value in developing criteria for success. Few success criteria have been published, and those publications that exist have offered only minimal performance criteria (Powers et al. 2009) or have discussed generic issues (Coen and Luckenbach 2000) associated with setting more rigorous targets.

Studies that have examined the ecological benefits of natural and restored oyster reef restoration have primarily demonstrated effects that are localized to the reef area. Demonstrations of chlorophyll *a* decreases near reefs are common in the literature, but the degree to which these localized effects result in measurable changes in primary production or water quality remain a point of substantial debate (Pomeroy et al. 2006; Newell et al. 2007; Coen et al. 2007). Dame and colleagues (2000), as well as Gerald and colleagues (2009) and Plutchak and colleagues (2010), used *before-after/control-impact* (BACI) designs to evaluate the effects of oyster removal (Dame et al. 2000) or addition (Gerald et al. 2009; Plutchak et al. 2010) within well-defined tidal creeks on a variety of ecological parameters, including primary production. Neither study found consistent and measurable changes in primary production, and there were relatively minor effects on transient and mobile fauna. These studies demonstrate the need for rigorous examination of hypotheses associated with the benefits of restored habitats as well as the need to consider the scale of restoration in this quantification. The expectation of Gerald and colleagues (2009) and Plutchak and colleagues (2010) that restoring 10% of the reef bottom with oyster reefs, a goal of several estuary-wide initiatives, would have

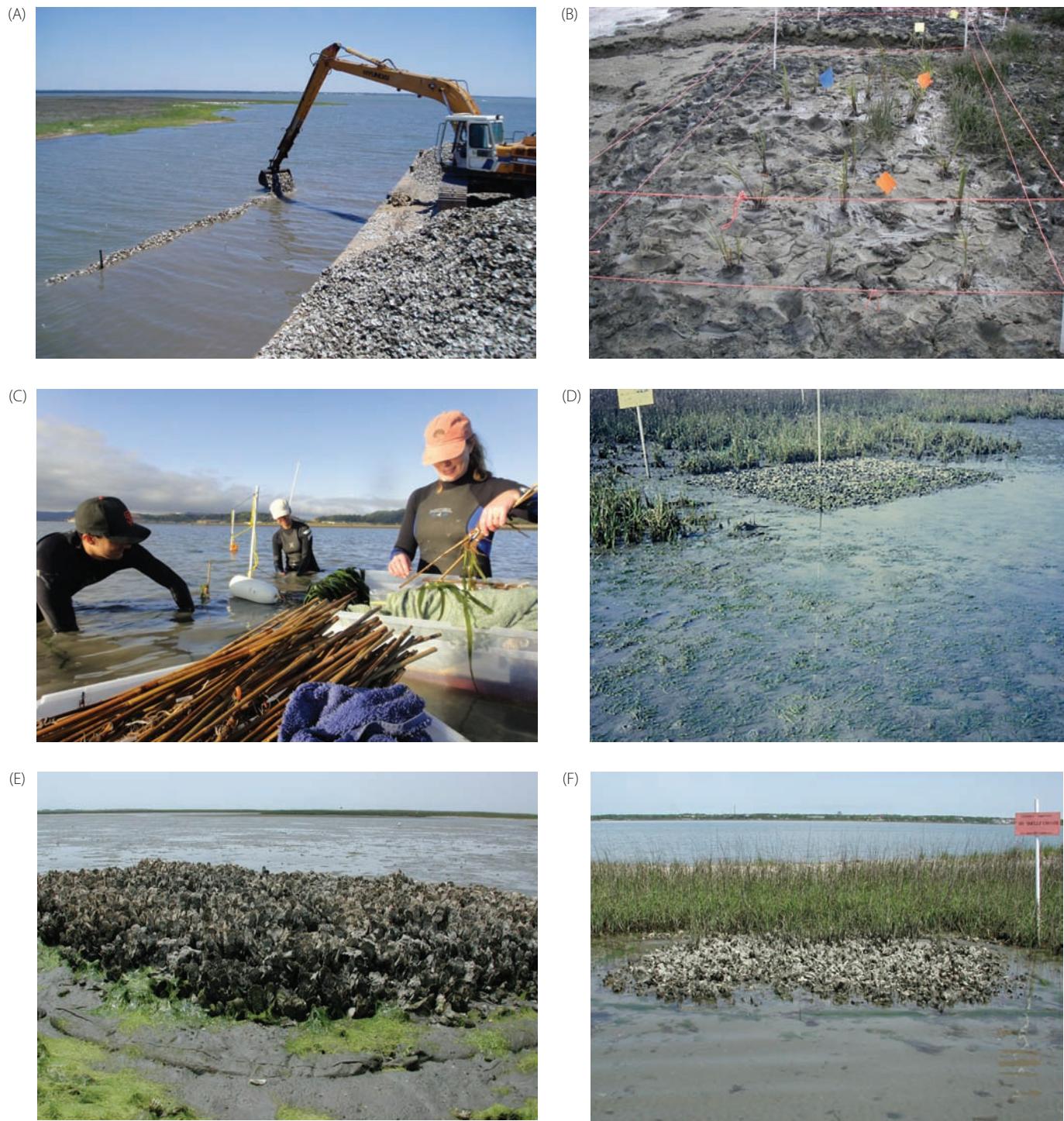


Figure 22.6 Small scale restoration experiments can be used to determine optimal design and placement of habitats as well as to test ecological concepts. (A) Restoration of oyster reef at Mobile Bay, Alabama, (B) experimental plot of native cordgrass in the San Francisco Bay with geese-excluding rope fencing, and (C) experimental planting of eelgrass as part of a living shorelines project in San Francisco Bay. Bottom row of photos illustrate different settings for oyster

reef restoration: (D) near seagrass and saltmarsh, (E) on unstructured mudflats, and (F) near saltmarsh. Monitoring following restoration of these three North Carolina oyster reefs demonstrated differing levels of ecological benefit in the three settings. (A courtesy of Steven Scyphers; B courtesy of Whitney Thornton; C courtesy of Stephanie Kiriakopulos; D–F courtesy of Jon Grabowski.)

a measurable effect on primary production appears not to be supported. Examination of such benefits as well as evaluation of other ecological processes (e.g., denitrification: Smyth et al. 2013) at a range of potential restoration locations is needed.

Coral reefs

The imperiled state of coral reef ecosystems throughout the tropics has been documented and reviewed in several seminal papers over the last decade (Jackson 1997; Jackson et al. 2001; Pandolfi 2003). Many of the problems facing coral reefs (e.g., increased ocean temperatures, ocean acidification) must be addressed through international and regional (e.g., overfishing: Jackson et al. 2001; nutrient enhancement: Szmant 2002) conservation plans (Bellwood et al. 2004) and are addressed in other chapters in this book. In this section we focus on local restoration efforts following discrete disturbances. As with the other biogenic habitats we have discussed, the need for restoration of coral reefs has increased over the last century as damage to reefs has accelerated. Localized damage to reefs can occur naturally from tropical storms, ENSO events, earthquakes, and lava flows, with the scale of damage proportional to the strength and duration of the event (Jaap 2000). Anthropogenic disturbances that physically alter reefs (e.g., by sedimentation or removal of material) primarily include activities related to dredging (ship channels or sand mining for beach restoration), improper vessel activity (grounding or anchor-related damage), fishing (blast fishing or trawling), or tourism (direct removal of coral or inadvertent damage; McManus et al. 1997; Jaap 2000).

Efforts to restore coral reefs following disturbances have primarily focused on repair strategies. The success of such efforts depends on a quick response to salvage and rescue damaged corals. According to Jaap (2000), efforts immediately following the event should focus on righting overturned corals, collecting live fragmented pieces of corals, short-term husbandry of coral fragments, and careful removal of sediment that rests over hard substrate. The process

is exceptionally labor-intensive and requires hundreds to thousands of hours of dive time depending on the scale of the disturbance. Reattachment of fragmented hard corals can be accomplished using epoxy, stainless steel rods, and concrete or mortar mixes. Soft corals and sponges are usually transplanted intact with a portion of the substrate to which they were attached. Using an approach analogous to hatchery production of fish or bivalves, coral nurseries have recently been established to provide coral species representing different growth forms (branching, leaflike, and sub-massive forms) for transplant (Epstein et al. 2001; Shaish et al. 2008). The concept of “gardening with corals” (Epstein et al. 2001) offers great promise, given that recruitment of stony corals into barren zones is very low (Jaap 2000). This low recruitment is a function of both low supply (larval source populations are primarily local) and the lack of the chemical signatures necessary for larval settlement in barren areas.

As with oyster reef restoration, when damage to coral reefs removes or pulverizes their three-dimensional structure, restoration efforts must include reestablishment of this vertical relief as the first step in successfully transforming barren expanses back into coral reefs. New artificial structures that can be used to establish vertical structure are generally made of limestone or cement. Limestone boulders (~1 m in diameter) or prefabricated concrete–limestone modules have been the most popular structures deployed. Establishment of a diverse coral reef assemblage requires decades. The long recovery times of both denuded natural substrate and prefabricated structures, coupled with the high financial cost of restoration and repair activities, highlight the importance of mitigation of future damage as well as the adoption of broader, more comprehensive conservation strategies (**Figure 22.7**).



Figure 22.7 Restoring complex and diverse habitats such as coral reefs can be a daunting task. Conservation efforts of healthy coral reefs like these in Fiji are far more cost effective than restoration. (Courtesy of Isabelle Côté.)

Mangroves

Mangroves, salt-tolerant trees that grow along sheltered coastlines in tropical and subtropical regions throughout the world, include approximately 70 species from 27 genera (Tomlinson 1986). These trees have been exploited for hundreds of years for their wood (as lumber, charcoal, and pulpwood) and for the invertebrates living on or among their prop roots (crabs, prawns, and molluscs; reviewed by Ellison 2008). They are also cleared from coastlines to make room for the development of resorts and aquaculture facilities (shrimp and fish farms), and are therefore disappearing globally at a rate of 1% to 2% per year (Alongi 2002; Spalding 2010). This rate of loss is cause for concern, especially considering the many functions of these trees, including high rates of productivity, which support both local and offshore benthic and pelagic food webs (Mumby et al. 2004; Manson et al. 2005); provision of habitat by the prop roots for numerous invertebrate and fish species, including vulnerable juvenile stages (Dorenbosch et al. 2004; MacDonald and Weis 2013); substantial sequestration of carbon, far exceeding that of terrestrial forests (Mcleod et al. 2011); and reduction of coastal erosion and a degree of shoreline protection during cyclones and tsunamis (e.g., Danielsen et al. 2005; but see Dahdouh-Guebas et al. 2005).

Mangrove plantations have long been maintained for rotational harvest of forest products, and this process has led to experience in planting methodologies and in following detailed management plans (Watson 1928). These plantations are typically uniform plantings of one or a few species, which limits their structural complexity and potential habitat value, although continued harvesting confers limitations on development of habitat (Ellison 2000). Similarly, restoration practitioners tend to plant only one or a few mangrove species, even though local richness (typically 4–8 species) is higher and, as in plantation settings, lower structural complexity may limit the richness and abundance of associated fauna (Ellison 2000, 2008). Recent work has revealed the importance of planting species mixes as well as clustering plants to maximize intraspecific facilitation. For example, planting seedlings closely in groups, rather than widely spaced as in plantations, may help to ameliorate low-oxygen conditions among neighboring plants (Gedan and Silliman 2009b; Huxham et al. 2010).

Site selection for mangrove restoration has been problematic in some regions, with plans made to plant mangroves along vast coastlines (e.g., in Southeast Asia) giving little consideration to current site conditions or historical mangrove presence (Dahdouh-Guebas et al. 2005; Barbier 2006). Mangrove planting on previously unvegetated mudflats frequently fails (Erfemeijer and Lewis 2000). In many locations, planting is deemed unnecessary if site conditions are appropriate for establishment and recruitment potential is high (Lewis 2005).

Loss of mangroves is correlated with a reduction in the abundance of associated fauna (Manson et al. 2005), which is reflected in reduced landings of fisheries species

such as shrimp (Loneragan et al. 2005). There is evidence that restoration of mangroves can reverse this pattern through natural recruitment of some crabs and gastropods (Macintosh et al. 2002; Ashton et al. 2003; Bosire et al. 2004), and it has been suggested that the structure of benthic assemblages may be a useful indicator for measuring the progress of restoration efforts, though it must be combined with careful consideration of historical and current land use and environmental context (Macintosh et al. 2002; Ashton et al. 2003). Performance measures in addition to faunal community composition are needed to evaluate restoration progress, and simulation modeling of factors such as vegetation dynamics, hydrology, and nutrient biogeochemistry can aid in assessment as well as prediction of future responses to climate change and other perturbations of both natural and restored mangrove habitats (Twilley and Rivera-Monroy 2005).

Extraction of invertebrates for food not only can damage mangrove habitats, but also greatly reduces or completely removes species at middle trophic levels, such as crabs, shrimps, and molluscs (Ellison 2008). These mid-level consumers of detritus, algae, or live mangrove tissues are important in the transformation and exchange of materials and energy, so their removal could have implications for both community- and ecosystem-level processes within the immediate habitat (reviewed by Ellison 2008). In addition, local removal of these invertebrates could affect offshore food webs, as well as reduce their recruitment to other mangrove stands in the region, consequences that must be considered in both conservation and restoration of these species and their dynamics.

As most research on animal interactions has focused on the effects of the mangrove habitat on benthic aquaculture and offshore fisheries, much less is known about the effects of the fauna on the mangroves themselves. Several invertebrates associated with mangroves have been shown experimentally to enhance the growth of these trees. For example, in Caribbean (Belize) red mangroves (*Rhizophora mangle*), several sponge species growing on the roots provide nitrogen to the trees and reduce boring into the roots by isopods, enhancing root growth (Ellison et al. 1996). In addition, burrowing by grapsid crabs in Australian (North Queensland) *Rhizophora* forests appears to increase soil aeration, which in turn enhances the productivity and reproductive output of the trees (Smith et al. 1991); however, grapsid crabs can also consume mangrove propagules (Farnsworth and Ellison 1991). Additional study is needed to determine whether manipulations of these or other candidate species could be used to maximize positive interactions that could increase mangrove restoration success.

Landscape-Level Approaches

Nearshore structured habitats often appear in a natural mosaic with other structured and unstructured habitats (i.e., intertidal and subtidal mudflats). As the marine

ecological literature began to document the importance of these natural landscapes to predator–prey dynamics, recruitment, and competitive interactions (Irlandi and Crawford 1997; Micheli and Peterson 1999; Mumby et al. 2004), restoration ecologists began examining the potential to link multiple habitats within one overall restoration. In many cases, the expectation for such restorations was that one habitat might facilitate (*sensu* Bruno et al. 2003) the maintenance and expansion of other biogenic habitats as well as increasing their refuge and foraging value to fishes and invertebrates (Scyphers et al. 2011). Several habitats within nearshore landscapes appear to be naturally linked or connected by faunal associations. Coral reefs located near seagrasses (Heck et al. 2008) or mangroves (Mumby et al. 2004; Olds et al. 2012) have higher fish utilization and energy flow than coral reefs in isolation. Including efforts

to maximize the potential benefits of habitat connectivity is becoming a design criterion in many marine restorations.

Within more estuarine landscapes, oyster reefs are frequently close to seagrass beds and salt marshes, and some ecological services appear to be enhanced by this proximity, while others remain unchanged and suggest redundancy. Restoration of oyster reefs close to shorelines may reduce erosive wave action and mitigate shoreline retreat while simultaneously increasing fish and invertebrate utilization (Meyer et al. 1997; Scyphers et al. 2011). The presence of an oyster reef may also stabilize sediments in the surrounding area, and this stabilization could facilitate the expansion of seagrasses and macroalgae (Figure 22.8). Further facilitation of seagrasses may occur if suspension feeding by oysters has a measurable effect on water clarity or local nutrient dynamics (Newell and Koch 2004).

For some parameters, however, pairing of habitats has not resulted in increased ecological benefits. Grabowski and colleagues (2005) found that oyster reefs restored in areas away from other structured habitats (seagrass beds and salt marshes) had higher densities of oysters and higher fish utilization. Similar results were shown by Gerald and colleagues (2009), who found minimal enhancement of fish utilization for oyster reefs restored within tidal creeks with abundant marsh edge habitat. They suggested that functional redundancy of the two habitats explained the lack of enhancement of fishes expected with oyster reef restoration in a landscape without existing emergent structure.

In salt marshes, the zone of transition from high-elevation marsh to upland supports a relatively diverse assemblage of plant and animal species and provides refuge for resident birds and mammals during high tides (Holland et al. 1990; Baye et al. 2000; Traut 2005). A number of rare or uncommon marsh plant species are found along a gently sloping upper marsh transition zone in prehistoric remnant marshes, but not in restored marshes that lack this feature (Boyer and Thornton 2012). Restoring gradual transitions to upland is

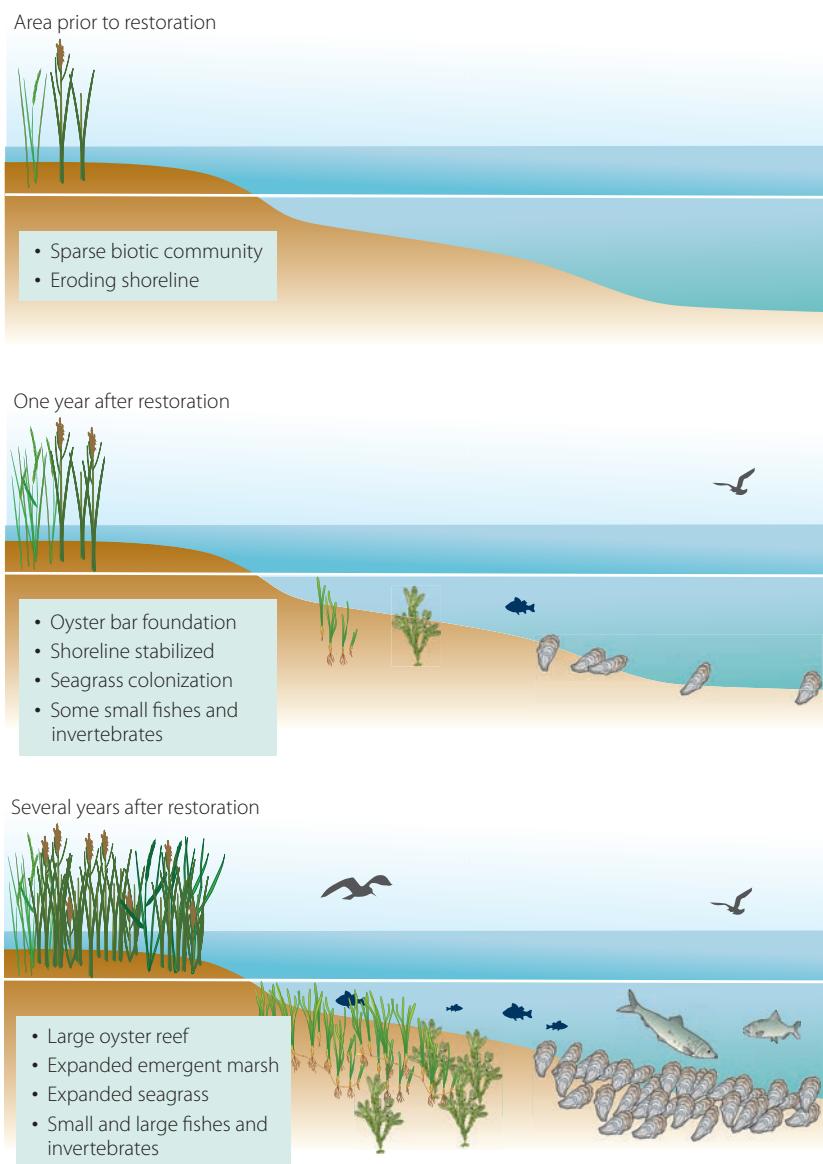


Figure 22.8 Linking multiple habitats in a landscape-level restoration could increase biological benefits compared with single-habitat approaches. In this diagram, the addition of a oyster reef near the shoreline mitigates shoreline erosion and enhances salt marsh vegetation. Stabilization of sediments in the area may also facilitate seagrass growth.

important to support these species and to accommodate upslope migration of the habitat with sea-level rise (Orson et al. 1987). In addition, the high marsh–upland ecotone is particularly subject to fragmentation through the building of roads and other infrastructure, disturbances such as trampling by humans or cattle, and invasion by non-native species; hence it is a habitat in need of focused conservation and restoration attention (Larson 1995; Weinstein et al. 2005; Wasson and Woolfolk 2011). Some mitigation projects have overlooked this transition zone in an effort to maximize acreage of the lower-elevation wetlands that support the specific rare animals driving the mitigation (e.g., Zedler 1998). Such practices can lead transitions to uplands in restored marshes to be steep or abrupt, with little to no support of the species or functions that would naturally be present in this habitat.

Ecosystem-Level Restorations

The realization that bottlenecks to population-level or habitat-level restoration projects may exist at a larger scale than more localized projects can address has led to restoration efforts at a much larger ecosystem scale. Changes in the hydrology of watersheds and estuaries, chronic pollution and nutrient inputs, shoreline development, and other large-scale environmental insults require major intervention. Many of the projects that address these issues require substantial engineering, construction, and environmental impact considerations. The potential advantages of these projects are as large as the complexities associated with their design and implementation. Many of the largest-scale efforts focus on reestablishing the connectivity of nearshore marine and estuarine areas with upstream watersheds or on improving the quality of these connections (e.g., decreasing nutrient loads of rivers). Removing or mitigating obstacles to anadromous fish passage in rivers is a key element of plans to restore wild salmon runs. Reestablishing freshwater inflow and sediment delivery to Louisiana coastal wetlands is the only remedy likely to mitigate the high rate of wetland loss there (DeLaune et al. 2003). To date, most of the efforts in Louisiana have focused on implementation of small-scale freshwater diversions (Das et al. 2012); plans for large-scale expansion of the technique are now proposed in the wake of the 2005 landfalls of Hurricanes Katrina and Rita. The largest and most ambitious restoration program in the United States, the Florida Everglades project, also involves substantive efforts to restore freshwater connections to estuarine and marine areas (Sklar et al. 2005).

Understanding the complexities of hydrological alteration of estuarine habitats is critical in assessing the efficacy of ecosystem-level restoration strategies. The anticipated changes associated with efforts to restore the Louisiana coastal wetlands and the Florida Everglades provide examples of such environmental complexities. Existing marsh habitats in Louisiana will experience changes in

flora and fauna as a result of freshwater diversions, and these changes will affect the delivery of ecosystem goods and services. Decreases in salinity and temperature associated with riverine input are predicted to reduce the yield of brown shrimp currently produced by the higher-salinity marsh areas (Adamack et al. 2012). Oyster reefs will be degraded as a result of lower salinity levels. These changes must be balanced with the gain in brackish-water habitats as well as the long-term reality of encroaching salt water, which is also a hindrance to oysters, which thrive in the relatively narrow range of 10–22‰ (Melancon et al. 1998). The expectation that many of the ecosystem services delivered by mid- and higher-salinity areas will simply be shifted down the estuary needs to be rigorously examined and planned for. For example, oyster reef habitat will need to be created downstream to realize this expansion. Restoration of the Florida Everglades provides another example of the complexities of restoring connectivity between rivers and coastal habitats. After decades of conversion of these wetlands to agricultural and suburban lands, as well as diversions of fresh water to support urban and agricultural development, the realization that restoration of the Everglades requires removal of many of these hydrological bottlenecks has led to restoration planning on a truly ecosystem scale (Davis and Ogden 1994). Changes in nutrient regimes (particularly phosphorus: Noe et al. 2001), water quality, and vegetation (Childers et al. 2003) within the Everglades will affect the ecosystem services and goods that are now produced, and those changes must be weighed against the promise and, ultimately, the realization of future benefits.

Although smaller-scale habitat restoration and mitigation projects cannot make a substantive dent in the legacy of habitat and species loss, they may be able to keep pace with existing environmental degradation if they are linked together under an ecosystem-level goal. Community-based restoration initiatives for marine and estuarine habitats are funded through several federal agencies and nongovernmental associations. Directly involving stakeholders in the restoration of marine habitats has substantive public outreach benefits. If these small-scale projects (1) are based on a clear understanding of the current bottlenecks to restoration, (2) have a rigorous and appropriate siting and design plan to overcome those bottlenecks, (3) are coordinated to reach ecosystem-level targets, (4) establish clearly defined criteria to judge success, and (5) implement a monitoring program to assess performance, then they are likely to have a positive effect in combating habitat and species loss. Unfortunately, few small-scale or even large-scale restoration projects meet all of these criteria. Reviews of past restorations have pointed to the absence of clearly defined goals and the lack of post-restoration monitoring (Coen and Luckenbach 2000; Hobbs and Harris 2001; Ruiz-Jaen and Aide 2005) as major omissions in many restoration initiatives. Establishing these criteria,

conducting the necessary monitoring over multiple years, and publication of the results are fundamental necessities in the practice of restoration ecology. As an applied science, its progress is contingent on the experience of its practitioners, the availability of published findings on rigorously designed restorations, and syntheses of these findings. An example of the potential usefulness of such syntheses can be found in a review by Cunha and colleagues (2012) of seagrass restoration programs in Europe. They found that none of the seagrass restoration programs implemented over the last 10 years were successful despite the use of numerous restoration decision models, frameworks, and guidelines. Their synthesis calls for a new paradigm in which restoration of seagrasses is not the first option considered in mitigation and conservation plans.

Socioeconomic Considerations in Marine Restoration

Restoration is a human-based venture that must take social values into account, and restoration scientists must recognize that most goals are dependent on some level of social acceptance (Choi 2007). Restoration of the extensive marsh areas lost to the footprints of the many urban areas along the coast would probably fail to gain general acceptance when relocations of property owners became a major component of the plan. Even when natural disasters (e.g., hurricanes) highlight the danger of living in nearshore areas, the reaction of the public is not to relocate their homes and restore their coastal property to natural marsh or sandy shores. In many cases, the response is to reengineer the area to be more resistant to disturbance so that human uses can continue. Often such projects take an engineering-first approach, in which persistence of structures is placed above any ecological considerations (Scyphers et al. 2011) despite the established negative geomorphologic (increased erosion surrounding the structure) and ecological consequences (loss of nutrient regulation, reduced foraging and nursery habitats; Bilkovic and Roggero 2008). The armoring of vast expanses of shorelines throughout urban areas is the product of such approaches. The societal response to the adverse effects of such armoring is often more engineering: beach "renourishment" to replace eroding sands. In turn, the beach renourishment process results in ecological impacts, particularly when the composition of the sediment used in renourishment fails to match the pre-renourishment conditions (Schlacher et al. 2007). This spiral of engineering and reengineering can be avoided if the first priority becomes conservation and if sustainability is a key element defining success of any restoration initiatives. Changing social attitudes is often difficult, but such changes must be included among the goals of future restoration projects, forcing restoration ecologists to embrace a truly interdisciplinary approach incorporating behavioral, natural, physical, and social sciences.

CONCLUSION

Marine restoration ecology is fundamentally challenging because of the need for immediate answers. Throughout this chapter we have focused on examples of how ecological theory and experimentation can help to establish restoration practices that support targeted assemblages and functioning of marine systems. Moving beyond trial and error should be the goal, but doing so will require the adoption of a more exciting and circumspect approach by restoration practitioners. The setting of realistic and quantifiable goals and the monitoring of progress toward those goals must be incorporated. When judged in this manner, it is likely that many past restoration projects have failed to meet their goals or have had unintended consequences. If restoration ecology embraces a more rigorous approach, then lessons from experiments, pilot studies, and unexpected results can be used to improve ecological function in new projects as well as in former restoration sites.

Finally, marine restoration ecology must recognize the challenges that current and future global change present. Conservation of existing populations and habitats must remain the priority; however, the persistence and expansion of many nearshore environments will require some level of intervention. Future marine restorations should be considered within the context of the rigorous legacy of marine ecological experimentation. By adopting such an approach, restoration ecology can move from a field that borrows much from marine ecology to a field that contributes substantially to new knowledge. Future restorations can capitalize on the large scale of many new restoration projects to evaluate whether patterns revealed in small-scale restorations function at landscape and ecosystem scales. Further, concurrent restoration projects in different locations can be used to test the context dependency of ecological and methodological patterns and inform future restoration activities.

ACKNOWLEDGMENTS

The authors wish to thank Brian Silliman for his editorial assistance and guidance as well as two anonymous reviewers for their helpful comments. Funding for the authors was provided through various grant agencies including the National Science Foundation, National Marine Fisheries Service, and Sea Grant. Numerous graduate and undergraduate students have participated in restoration projects led by the authors and these projects have shaped the authors' opinions on project successes and failures. Among these individuals are Lindsey Carr, F. Joel Fodrie, Nathan Geraldi, Kevan Gregalis, Stephanie Kiriakopoulos, Cassie Pinnell, Rosa Schneider, Steven Scyphers, and Whitney Thornton.

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