18

Anthropogenic Threats to Australasian Coastal Salt Marshes

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Australasian coastal salt marshes have experienced intensive human modifications on a relatively short time scale compared to many modified marshes in America, Africa, Asia, and Europe. In this chapter, we review anthropogenic threats first to New Zealand and then Australian salt marshes and provide recommendations to abate these threats and improve restoration efforts of these important coastal communities. Initial impacts by Australian aboriginals (about forty thousand years ago) and New Zealand Maori (about eight hundred years ago) were likely to have been small and were limited to harvesting of plants and shellfish. However, following a few centuries of colonization, indirect effects of Maori avian hunting and coastal forest clearance could have reduced herbivore browsing considerably and caused increased runoff, with increased sedimentation and nutrient inputs in New Zealand marshes. With European colonization about two hundred years ago, reclamation of marshes became a major focus of human development (an imported habit from England), and large salt marshes were converted to urban structures (roads, buildings, ports, marinas) and agricultural grassland. Today, reclamation threats are largely under control and generally prohibited by planning legislation. Still, a suite of indirect anthropogenic stresses (e.g., livestock grazing, eutrophication, and invasions by nonindigenous species) threaten the marshes that remain. Quantitative distribution data of specific marsh species are sparse, descriptive studies dominate the literature, and only a few process-oriented experimental studies have been conducted to elucidate factors controlling community structure. Thus, human impacts on Australasian salt marshes and how they may impact community organization are largely based on anecdotal evidence, older semiquantitative surveys, and gray literature reports. Imminent visible threats include livestock grazing/trampling, human trampling and usage of recreational vehicles, waste dumping, invasions by nonindigenous species, and mangrove expansion. Less visible threats include storm water runoff and contamination with persistent organic molecules (e.g., pesticides) and heavy metals. Enhanced sedimentation and eutrophication are major problems in many larger estuaries with intense urbanization, but little is known about how these stressors affect bordering salt marsh structure and function. The consequences of climatic change (sea-level rise, temperature rise,

increased storminess, ozone destruction) will affect future Australasian salt marshes, although there is much to learn about both the changes to the environment and the biotic response. Clearly, large-scale inventories are still very much needed to track changes, but the application of high-resolution remote sensing and geographic information system (GIS) techniques will assist the analyses of changes. We suggest that managers outline and carry out long-term monitoring programs that capture present and future distribution ranges of key marshes, plants, and animals, but also that researchers supplement such programs with controlled experiments that test for effects of grazing, competition, facilitation, nutrient limitation, soil texture, and various human stressors, to ensure that we can detect cause-and-effect relationships and thereby create means to ameliorate and mitigate future threats. Ideally, there should also be continued collaboration and data interchange between stakeholders to ensure synergistic benefits for both the managers (e.g., use the experimental data to adjust monitoring programs and provide causation to their distribution data) and researchers (use the monitoring data as background information for experiments and to extrapolate findings to larger spatiotemporal scales).

Coastal salt marshes and salt meadow (hereafter salt marshes) are some of the most important habitats in the world. Marshes protect coastlands from erosion and floods, act as feeding grounds and shelter for aquatic and terrestrial species, provide carbon sinks that reduce global warming impacts, and are natural systems of wastewater treatment that reduce adverse effects of eutrophication and pollution (Chapman 1974; Adam 1995; Valiela and Cole 2002; Bertness, Silliman, and Jefferies 2004). Salt marshes are also important ecosystems within Australasia (New Zealand and Australia) and are likely to be increasingly impacted by humans as populations grow, urbanization increases, climatic change accelerates, and the coastal zone is further utilized for human development, fisheries, and aquaculture. To be able to protect marshes from these threats, a review of what is known about Australasian marshes and human threats to these marshes is timely. We address the review in separate sections for New Zealand and Australia partly because the two countries have been geographically separated for more than seventy million years and are distinctively different with respect to natural history (e.g., they vary widely in geology, tectonic activity, geomorphology, topography, climatology, oceanography, animal and plant distribution patterns, biogeography, degree of endemics), partly because of the two countries' different colonization histories (e.g., Australia was colonized forty thousand years ago, vs. New Zealand only about eight hundred years ago), degree of human-induced extinction rates, and different legislation and administration. During the review, we noted that quantitative scientific-based knowledge regarding New Zealand salt marsh ecology was relatively sparse. Hence, to advocate and stimulate salt marsh research from this minicontinent, we included a more thorough review of both the human colonization history and of the existing salt marsh literature from this country (e.g., we here include Web-based information and a questionnaire to local managers).

NEW ZEALAND SALT MARSHES, HISTORY, MANAGEMENT, AND HUMAN THREATS

LATE HUMAN COLONIZATION

New Zealand is considered the last large landmass to be colonized by humans (Diamond 1999). East Polynesians (ancestral Maori) discovered and colonized New Zealand only seven to eight hundred years ago, first settling around estuaries (King 2003; McFadgen 2003)—that is, in the vicinity of coastal salt marshes. The first few settlers likely had only a minor impact on salt marshes, mainly hunting birds, fishing, and collecting snails, mussels, oysters, and coastal flax for weaving. However, with a rapidly growing population, indirect effects on the marshes could have been dramatic within a few centuries (McGlone 1989). For example, in less than three hundred years from first colonization, more than thirty endemic bird species, many of which likely foraged in marshes—including coastal geese, swans, pelicans, rails, and eleven moa species—were probably hunted to extinction. An estimated thirty thousand to ninety thousand moa were killed around the mouth of the Waitaki River alone (King 2003). In addition, millions of nesting seabirds and coastal seals were killed (the former also by introduced rats and dogs) and large areas of coastal forest burned, probably for hunting purposes (85 percent forest cover reduced to 55 percent before European settlement; Hutching 1998; King 2003). Removal of an abundant avian megafauna coupled with alteration of drainage basin properties (e.g., increased sedimentation would be expected following coastal deforestation) is likely to have altered salt marsh community structure and ecosystem functioning. For example, in the Northern Hemisphere, it has been shown that geese can have dramatic impacts on marshes, including both a negative effect of grubbing rhizomes and a positive effect associated with nutrient enrichment (Jefferies 1988; Esselink et al. 1997; Dormann and Bakker 2000; Bos 2002; Bos et al. 2002; Jefferies and Rockwell 2002). Thus, the extermination of the prolific populations of ground-browsing birds has potentially created the present view of "naturally nongrazed" marshes in New Zealand. This removal of herbivore top-down control is analogous to the near-extermination of megaherbivores by humans in the Caribbean (manatees and turtles; Jackson 1997) and on the North American Plains, although the latter extinctions are also attributed to climate changes (Owen-Smith 1989).

Utilization and alterations of the coastal zone accelerated with the arrival of European whalers and farmers in the late eighteenth century. For example, nine additional birds species became extinct, seals and coastal birds in more remote areas were hunted to near-extinction, and intensive large-scale farming and urbanization commenced. Today, after only eight hundred years of turbulent human history, around four million people inhabit New Zealand's approximate 270,000 square kilometers, most living in close proximity to its 11,000-kilometer coastline (McLay 1976; Hume 2003) (some authors estimate up to 18,000 kilometers; Hutching 1998; Bell, Hume, and Hicks 2001; King 2003; Rouse, Nichol, and Goff 2003). This short human history has greatly modified the natural landscape, with examples such as clearance of more than 85 percent of buffering coastal forest (Auckland Regional Council 2000a). Such removal can have large-scale impacts on salt marshes by releasing unfiltered nutrient-rich waters into the marsh (Bertness, Ewanchuk, and Silliman 2002; Silliman and Bertness 2004); destruction of more than 85 percent of wetlands (Taylor and Smith 1997; Auckland Regional Council 2000b); "harvest" of millions of marine mammals and coastal birds (often to local extinctions); extinction of half of the endemic bird fauna (King 2003); introduction of twenty-five thousand nonnative plant species, of which about two thousand survive without human aid (Holland 2001); reduction of the native forest to a quarter of its prehistoric extent (Holland 2001); and conversion of almost fifteen million hectares to farmland (Statistics New Zealand 2002) inhabited by millions of sheep, cattle, and deer (Hutching 1998). Such short-term and large-scale landscape alterations can only have had dramatic impacts on the extent, community structure, and ecosystem functioning of New Zealand salt marshes. However, data showing direct linkages between this activity and changes in marsh community structure, as has been shown elsewhere, is largely lacking for New Zealand.

SALT MARSHES OF NEW ZEALAND

More than 80 percent of New Zealand's coastline is "energetically exposed" with sandy or rocky substratum. Still, some 300 to 350 low-energy bar-built estuaries, drowned river valleys, river mouths, and lagoons exist (hereafter estuaries) (McLay 1976; Bell et al. 2001), where siltation, wave protection, and semidiurnal tides, typically of one to three meters, provide conditions for salt marsh development. Sea level was about 120 meters below mean sea level (MSL) 18,000 years ago but stabilized at its present level about 6,500 years BP (Hume 2003), leaving a relatively short time for siltation, dispersals, and establishment of salt marshes along the present coastline. The estuaries vary from a few hectares to more than fifteen thousand hectares, with two-thirds being less than five hundred hectares (McLay 1976). The areal extent of salt marshes at regional and national scales has been determined from Landsat 7 images (30 × 30 – meters resolution, 1996–1997 images, LCDB1 category Coastal Wetland = salt marsh + coastal flax [*Phormium tenax*], minimum habitat size = 1 hectare). From this large-scale inventory, it is estimated that 10,400 hectares exist on the North Island and 9,700 hectares on the South Island (http://www.maf.govt.nz/statistics/primaryindustries/landcover). This total area of 20,100 hectares is slightly smaller than the extent of the mangrove *Avicennia marina* (=*A. resinifera*), only found north of 38° S of 22,500 hectares. However, because numerous New Zealand marshes are fringing and distributed in small pockets and along thin lines, cumulative large areas are likely to remain undetected by the Landsat resolution. A few large-scale semiquantitative inventories (Braun-Blanquet methodology) have described national distributions of species and communities (Thannheiser and Holland 1994; Haacks and Thannheiser 2003). According to Haacks and Thannheiser (2003), seventy-four salt marsh plant species exist in New Zealand, with almost half being nonnative and only seven species endemic (*Carex litorosa*, *Lachnogrostis littoralis*, *Leptinella* [= *Cotula*] *dioica*, *Leptocarpus* [= *Apodasmia*] *similis*, *Plagianthus divaricatus*, *Poa cita*, *Puccinellia walkeri*). This is a low number of endemics, compared to the national average of 80 percent endemics of all native species (Holland 2001), but not atypical for salt marshes. Many New Zealand salt marsh plant genera are cosmopolitan, and many species are shared with Australia. In general, distribution patterns resemble patterns from southern Australia (Chapman 1974; Adam 1990), with a typical zonation consisting of (from low to high elevation) the seagrass *Zostera* Capricornil (below midtide), the mangrove *A. marina* (only above 38°), *Sarcocornia quinqueflora*/*Samolus repens* (low marsh), *Juncus kraussii* (referred to in most New Zealand accounts as *J. maritimus*)/*Leptocarpus similis* (middle-high marsh), and bordered upland by the flax *P. tenax* and coastal shrubs (*Plagianthus divaricatus*, *Coprosma* spp., and *Leptospermum scoparium*) (fig. 18.1; Morton and Miller 1973; Chapman 1974; Thannheiser and Holland 1994; Haacks and Thannheiser 2003; Wardle 1991). *S. repens* and *S. quinqueflora* are considered pioneer species (Chapman 1974; Thannheiser and Holland 1994), and they can also be found at higher elevations—for example, in disturbed patches, and scattered on open shingle, sand, and mudflats. Vertical zonation pattern is not always as clear-cut as described from North American and European coastlines; and because species limits often are reversed, species distribution is sometimes referred to as "mosaic-like" (Chapman 1974; Thannheiser and Holland 1994). For example, *L. similis* is not found in all marshes and is sometimes found as the low marsh species, with *S. quinqueflora* occurring in the higher marsh in disturbance patches (e.g., in some marshes in the Catlins). It is possible that the shifting vertical patterns and competitive hierarchies among plants are modified and/or flip-flopped by local differences in freshwater input (Wilson et al. 1996), soil textures (Partridge and Wilson 1989), climatic

FIGURE 18.1 Border between *Samolus repens* (front, low marsh) and *Juncus kraussii* (back, middle marsh) zones. Note how drift log can accumulate in marshes (center), causing disturbances and opening up spaces for pioneer species like *Sarcocornia quinqueflora* and *S. repens*.

conditions (Haacks and Thannheiser 2003), and nutrient levels (Levine, Brewer, and Bertness 1998). For example, it is likely that *S. quinqueflora* gains a "competitive edge" over *S. repens* on coarse sandy soils and *L. similis* over *J. kraussii* with high freshwater inputs (personal observation). Similar freshwater–saltwater dominance reversals have been documented experimentally for North American salt marsh plants (Crain et al. 2004; Pennings, Grant, and Bertness 2005). Additional common native species include *Selliera radicans*, *Baumea juncea*, *Suaeda novae-zelandiae*, *Leptinella dioica*, *Schoenoplectus pungens*, *Bolboschoenus medianus*, *Mimulus repens*, *Puccinella stricta*, *P. walkeri*, *Triglochin striatum*, and *Apium prostratum* (Johnson 1989; Wilson et al. 1996; Haacks and Thannheiser 2003). Several accounts exist from local salt marshes (but unfortunately often not presenting biomass values with standard errors)—for example, Pollen Island, Auckland (Chapman and Ronaldson 1958); Avon-Heathcote Estuary, Christchurch (Mason 1969; Knox 1992; Webster 1997; Thomsen, Marsden, and Sparrow 2005); Lake Ellesmere, Canterbury (Evans 1953), Hapuka Estuary, West Coast (Dickinson and Mark 1999); and Nelson Haven, Nelson/Marlborough (Davies 1931; Doak 1931). The best-described marshes are from the Otago region where it has been shown that vertical zonation patterns correlate with salinity tolerances, freshwater inflow, soil texture, and "local peculiarities" (Paviour-Smith 1956; Partridge and Wilson 1987a, 1987b, 1988a, 1988b, 1989; King, Wilson, and Sykes 1990; Wilson et al. 1996). In one of the few published manipulative experiments, Partridge and Wilson (1988a) showed that in Otago, most species survived up-elevation transplantation but typically died following down-transplantation, suggesting competitive limitation upward and physiological stress limitation downward. However, because neighbor presence (i.e., competitive and facilitative interactions) was not manipulated, this conclusion is not definitive.

There are few data on animals utilizing New Zealand salt marshes, and this research area is therefore wide open for eager scientists. Due to a seventy- to eighty-million-year-long geographic isolation following the break-off from Australia and dramatic ongoing changes in geology and climate (e.g., thirty-five million years ago, New Zealand was reduced to a few small flat islands), mammals (except bats) never occupied New Zealand (Harvey 2001). Thus, browsing would have been by the recently extinct moa (*Anomalopteryngineа* spp.) and geese (*Cnemiornis* spp.) (Atkinson and Greenwood 1989). The few scientific papers we are aware of that quantify salt marsh fauna showed that amphipods, nematodes, oligochaetes, and diptera larvae were the most common in terms of biomass in an Otago salt meadow (Paviour-Smith 1956) and that the marine invertebrate fauna of Christchurch marshes is sparse, with only four mollusks, five crustaceans, and five polychaetes. None of these species were found exclusively in salt marshes (Marsden and Heremaia 1998). Qualitative observations and data from adjacent estuarine ecosystems suggest that key salt marsh species with potential large-scale ecological effects likely are the burrowing crab *Helice crassa* and the gastropods *Amphibola crenata*, *Ophicardelus costellaris*, and *Potamopyrgus estuarinus* (due to their ubiquity throughout New Zealand—personal observation; Morton and Miller 1973; Jones and Simons 1983; Juniper 1986; Marsden and Heremaia 1998). The crab (*H. crassa*) oxygenates soils and increases bioturbation (Morrisey et al. 1999; Williamson et al. 1999; Gibbs, Thrush, and Ellis 2001) and as a consequence may increase primary production particularly in poorly drained soils (Bertness 1985), whereas the gastropods are likely to control decomposition and/or bacterial and primary production (Juniper 1987a, 1987b; Silliman and Bortolus 2003; Silliman and Newell 2003). Few data exist on how birds utilize New Zealand salt marshes, but Paviour-Smith (1956) note that migrating dotterels winter on salt meadows and that gulls, oystercatchers, stilts, and godwits commonly feed in salt meadows or seek shelter behind sedges/rushes at high tide and under strong winds (we have observed similar patterns in both North and South Island salt marshes). Also, Lowe (1997; Marsden and Heremaia 1998) has shown that densities of oystercatchers were highest in a marsh not separated by stop banks and that stilts were most abundant in a newly artificially formed marsh. In this study, densities, foraging time and prey capture efficiency varied between bird species, salt marsh sites, and seasons, but not in any simple consistent manner. Also, little is known about fish utilization, but Nairn (1998; Marsden and Heremaia 1998) found abundant juvenile flounders and mullets in Christchurch marsh channels during summer months. We expect fish utilization to be as important as found in New South Wales (NSW), Australia, where up to sixteen fish species, six of commercial importance, were caught in pop nets at high tide in the *Sarcocornia* zone, with a mean density of 0.6 individual per square meter (Mazumder, Saintilan, and Williams 2005a, 2005b). Similar measurements should be made in different New Zealand salt marsh zones, preferentially coupled with experiments that document interactions between visiting fish and birds and permanent salt marsh inhabitants.

MANAGEMENT OF NEW ZEALAND SALT MARSHES

New Zealand salt marshes are mainly managed by regional councils and some city councils, but a large proportion is also under management by the Department of Conservation (some marshes are also under private ownership). For example, to conduct scientific research in a salt marsh, resource consent must be obtained from the appropriate council; and if the marsh is also a Department of Conservation wildlife reserve, an additional research permit must be obtained. Some councils, the Department of Conservation, and local trusts have carried out salt marsh restoration projects (Bergin 1994; Auckland Regional Council 2000a, 2000b; fig. 18.2), and seeds of the most common salt marsh species can be bought from commercial nurseries (e.g., *J. kraussii*, *L. similis*, New Zealand Tree Seeds), obtained from local voluntary groups/trusts (e.g., by the Guardians of Pauatahanui Inlet, personal observation), or attained from council nurseries (Thomsen et al. 2005).

It has been shown that transplantation success of *J. kraussii* and *L. similis* depended on initial transplant size (the larger the better) but not on transplant spacing or nutrient additions (twenty-five grams of slow-release NPK pellets added to each transplant; Bergin 1994). The study by Thomsen et al. (2005) confirmed that

FIGURE 18.2 Example of a restorations project in Pauatahanui inlet near Wellington. *Juncus kraussii* and *Leptocarpus similis* have been transplanted to restore local marshes.

FIGURE 18.3 *Leptocarpus similus* is relatively easy to transplant to high marshes, particularly around freshwater sources. Several councils and trusts grow the species in nurseries, making this species readily available for restoration projects.

these two key species are suitable for use in restoration (fig. 18.3), whereas *Schoenoplectus pungens*, despite an initial transplant survival, failed to regenerate the next spring following seasonal dieback. Thomsen et al. (2005) also showed that the survival and growth of *J. kraussii* and *L. similis* did not depend on soil type (dredged estuarine soils vs. marsh soils) but that plants from natural populations had higher biomass than plants out-transplanted from a local nursery, presumably because the nursery plants were "softer"—that is, less resistant to wind, currents, and waves and/or were more exposed to rabbit grazing. Still, much applied ecological information is needed to ensure efficient restoration, reestablishment, and management of New Zealand salt marshes, such as by testing for competition and facilitation, repeated nutrient additions, timing of transplantation, and effects of multispecies versus single-species plantings.

ANTHROPOGENIC THREATS

Without information on the prehuman extent of nationwide salt marsh area, it is not possible to quantify with certainty the area of salt marsh lost. However, in Bay of Plenty, an estimated 56 percent of estuarine wetlands on harbor margins has been lost/reclaimed (calculated from Park 2000 and the New Zealand Land Cover Data Base, http://www.mfe.govt.nz/issues/land, accessed 2006). This regional loss is less than intensively human alienated North American coastal areas (e.g., up to 80 percent lost salt marshes in New England; Bertness et al. 2002) and much less than the more than 90 percent salt marsh loss suggested by Marsden and Heremaia for the whole of New Zealand (1998, but no data and/or references follow this high suggested loss).

Anthropogenic impacts to New Zealand salt marshes can be grouped into four general lines of threats: (1) land reclamation and impacts associated with utilization of watershed (e.g., livestock grazing, eutrophication, and enhanced sedimentation); (2) pollution and human trampling effects; (3) biological invasions; and (4) climatic changes. Because of the scarcity in the literature of hard data, we submitted a questionnaire to managers in the regional councils regarding local salt marsh management and perceived threats. Six councils replied, and we have included their perception about threats to local and regional salt marshes in the text.

LAND CLAIM AND HUMAN WATERSHED UTILIZATION

Land claim has probably had the most severe impact on New Zealand marshes (Haacks and Thannheiser 2003; see estimates presented earlier). For example, six major ports and numerous smaller harbors have been built in New Zealand estuaries, and these constructions typically involved considerably foreshore reclamation (Hume 2003). Also, at least 164 causeways have been constructed for roads and railways in estuaries (Hume 2003). The causeways reduce/cutoff tidal flow and the hinterland marshes have often been converted into farmland or urban buildings (personal observation). Even where wetlands remain behind the causeway, vegetation composition will typically be very different with degraded water quality, decreased salt water input, and reduced abundance of fish and macroinvertebrates (Roman, Garvine, and Portnoy 1995; fig. 18.4), as has also been documented in tidally cutoff or restricted mangroves (Layman et al. 2004; Layman, Arrington, and Blackwell 2005). Most large New Zealand urban areas are coastal, and the development of these areas would certainly have included reclamation of salt marsh areas. The growing population of New Zealand (e.g., estimated 4.4 million in 2021, "Part 9: Sub-national Demographic Projections," *Demographic Trends 2001*, Statistics New Zealand), and a trend of people seeking coastal residency will continue to put pressure on existing salt marshes. Still, many salt marshes are under management by the Department of Conservation and/or regional councils, and this is probably why contemporary land reclamation for urban development are presently only considered a relatively low risk by managers (average 2.2 [out of 2, 2, 3, 1, 2, 2 [where score 1 = not considered a problem, score 2 = considered a minor problem, score 3 = considered a major problem, and score ? = potentially a problem, but considered unknown]).

FIGURE 18.4 A causeway in Pauatahanui inlet that cut through a marsh, reducing tidal flow and habitat connectivity. The vegetation composition in the marsh behind the causeway will typically be different, sometimes with degraded water quality, decreased saltwater input, and reduced abundance of fish and macroinvertebrates.

Agricultural land claims have probably also taken a high toll on New Zealand salt marshes. Enormous areas of New Zealand's landscape have, in a few centuries, been converted to pasture (11.5 million hectares) and other agricultural practices (an additional 4 million hectares) (Statistics New Zealand 2002). Again, it is unknown how much salt mash has been converted to grassland, but agricultural land claim was considered slightly less of a threat than urban land claims by regional managers (average 1.8 [2, 3, 2, 2, 2, 1]). In addition to conversion of salt marshes to pastoral grassland, agriculture also affect salt marshes directly by livestock grazing of existing salt marsh vegetation and by indirect nutrient runoff effects (Jensen 1985; Andresen et al. 1990; Kiehl, Esselink, and Bakker 1996; Kiehl et al. 1997; Levine et al. 1998; Esselink, Fresco, and Dijkema 2002). For example, in Europe, grazing at low to medium livestock densities increase species richness by reducing cover of the tall *Elymus athericus* and *Atriplex portulacoides* and thereby decreasing competition and allowing for the coexistence of smaller species like *Puccinellia maritima*, *Triglochin maritima*, and *Plantago maritima* (Bos et al. 2002).

Following land claim, intensive utilization of the watershed is likely to have had the second-most dramatic impacts on New Zealand salt marshes (Haacks and Thannheiser 2003). The main livestock are sheep (39.5 million), cattle (9.7 million), and deer (1.6 million) (Statistics New Zealand 2002), with the latter mainly kept under strict fenced conditions. These large numbers correspond to some of the highest densities in the world (Taylor and Smith 1997), and because farmers often do not restrict sheep access to marshes, browsing and trampling likely have important effects on salt marshes, particularly as they evolved without browsing mammals. This is reflected in livestock grazing being considered a medium to high risk (average 2.3 [3, 2, 3, 2, 2, 2]). More specifically, Hacks and Thannheiser (2003) suggest that *C. coronopifolia* is facilitated by grazing but that *S. repens* and *S. radicans* are relative insensitive to grazing. They also observed that *S. quinqueflora*, *S. novae-zealandica*, *M. repens*, and *Puccinellia* spp. were found very rarely in intensively grazed areas. Wraight (1964) reports that in a salt meadow in Lake Ellesmere, abundant *A. stolonifera* and *Trifolium fragiferum* were found with little grazing, whereas *J. kraussii*, *Hordeum marinum*, *S. radicans*, *P. coronopus*, and *C. dioica* were more abundant under moderate to heavy grazing. Finally, *A. prostratum*, *Atriplex patula*, *M. repens*, and *T. fragiferum* were most impacted by grazing. Despite these observations, it was concluded in a recent review undertaken to estimate livestock effects on New Zealand wetlands that little information was available (not a single reference was found that documented effects on salt marshes; Reeves and Champion 2004). Clearly, it is important to conduct large-scale correlative surveys to quantify the extent of salt marsh grazing and couple these surveys with cage exclusion experiments and feeding preference trials. Given that cattle tend to be relatively unselective grass "tearers" compared to sheep, which are more selective "biters" (Bos et al. 2002), it cannot be assumed that the effect of grazing on New Zealand salt marsh plants is similar for sheep and cattle grazing.

A by-product of intensive large-scale agriculture is nutrient enrichment, which has become an increasing problem in New Zealand freshwater and coastal waters (Taylor and Smith 1997). Point source sewage from the larger urban areas has previously been a large-scale nutrient contributor but is today somewhat limited due to sewage treatment, although storm water runoff still causes frequent peaks of high nutrient outflow (Taylor and Smith 1997). Due to the high livestock densities grazing on steep and shallow soils (more than 70 percent of New Zealand is considered steep topography; Hutching 1998) with high applied fertilizer rates, which accelerates nutrient runoff, the occurrence of algal blooms are an increasing problem in freshwater and estuarine habitats (Taylor and Smith 1997; Parkyn et al. 2002). This is probably why eutrophication is considered a medium to high threat to New Zealand salt marshes by the regional managers (average 2.3 [?, 2, 2, 2, 3, ?]). Although no studies have linked nutrient availability to New Zealand salt marsh ecology, we expect that enhanced nutrient supply will favor epibenthic diatoms and ephemeral green algae (Fletcher 1996; Raffaelli, Raven, and Poole 1998), facilitate nonnative opportunistic marsh plants (Silliman and Bertness 2004), increase salt marsh productivity (Kiehl et al. 1997; Silliman and Zieman 2001; Brewer 2003), and alter competitive hierarchies and thereby dominance patterns (Levine et al. 1998). Indeed, it is important to reiterate that salt marshes are buffer zones between land and sea, typically facilitating nutrient uptake and transformation as well as sediment deposition (see later discussion) and thereby reduce these stressors on adjacent, more susceptible seagrass, macroalgal, and oyster beds (Valiela and Bowen 2002; Valiela and Cole 2002). Thus, maintaining and/or restoring these buffer zones could be a partial management option to reduce local problems of enhanced sedimentation and eutrophication.

New Zealand watersheds are prone to erosion and high sediment transport, given the steep topography combined with relatively soft bedrocks, relatively high precipitation rates (typical values of 800 to 1,200 mmy^{-1} but up to 11,000 mmy^{-1} in Fjordland; Stuarman, Owens, and Fitzharris 2001), and a high tectonic activity (i.e., situated on the convergence zone of the Australian continental plate and the Pacific oceanic plate). However, due to a high forest cover, prehuman estuarine sedimentation rates were relatively low (less than one millimeter per year; Hume 2003). Following forest clearings, urbanization, and intensive livestock farming, it is today estimated that more than eighteen million hectares are threatened by erosion (Hutching 1998), and estuarine sedimentation rates have typically doubled or tripled (Hume 2003). We are not aware of any studies that have tested for effects of sedimentation on New Zealand salt marsh community structure, but Swales, MacDonald, and Green (2004) showed that, at wave-exposed sites, *Spartina* facilitates shell accumulations, which feedbacks to again stabilize *Spartina*. In other countries, it has been shown that sedimentation can enhance salt marsh development (Gibblin, Valiela, and Teal 1983; DeLaune et al. 1990), presumably by reducing drowning and enhancing nutrient supply. New Zealand salt marshes have been shown to accumulate three to twelve millimeters per year (Lee and Partridge 1983, although this is for invasive *Spartina* species), documenting that at least this species does well under medium sedimentation levels. Today, sedimentation is considered the most important "contaminant" in the coastal zone (Williamson et al. 2003) and is also considered a high risk to salt marshes by regional managers (average 2.7 [3, 3, 2, 2, 3, 3]). Nevertheless, we disagree with this survey and predict salt marsh plants to be relatively robust to sedimentation (DeLaune et al. 1990; French and Spencer 1993; Kastler and Wiberg 1996), for example, compared to rocky shores, oyster reefs, seagrass beds, and ecosystems inhabited by filter feeders (Airoldi 2003). Clearly, specific tests should be conducted to evaluate the sediment sensitivity of native New Zealand salt marsh plants and animals.

An indirect effect of sedimentation on salt marsh plant distribution is competitive displacement by *A. marina* in northern estuaries. Here, *A. marina* has been observed to expand onto open mudflats (Young and Harveya 1996; Nicholls and Ellis 2002; Ellis et al. 2004; Park 2004) and potentially also salt marshes. The expansion has been associated with high estuarine sedimentation rates ("infilling," partly natural, partly human caused), but additional factors such as altered precipitation patterns, sea-level rise, eutrophication, and temperature rise are possible alternative causes (Saintilan and Williams 1999). If *A. marina* expands onto higher ground and the hinterland is fixed (e.g., by stop banks and causeways), salt marshes will be squeezed and eventually outcompeted (sea-level rise–coastal squeeze scenarios). No studies have tested for competitive effects between *A. marina* and salt marsh plants, and we are only aware of one report that shows subtle mangrove expansion into a salt marsh area (Park 2004). Still, mangrove expansion was considered a high risk by northern council managers (average 2.7 [3, 3, 2]).

The construction of about eighty large hydrodams throughout New Zealand (providing nearly 75 percent of the national electricity demand; Taylor and Smith 1997) has clearly also altered watershed properties. Such large-scale control of flow conditions likely affect coastal river mouth salt marshes by controlling the extent, duration, and frequency of freshwater flooding, but we are not aware of any study that have linked dam-induced altered freshwater flow patterns to salt marsh ecology. Alteration of river mouth salinity regimes will ultimately control species patterns in fringing marshes, as most species have optimal growth in freshwater, but different tolerances to salinity and thereby different competitive advantages; for example, reducing freshwater river outlets will favor the slower-growing, more salt-tolerant species (Partridge and Wilson 1987a, 1987b; Wilson et al. 1996; Crain et al. 2004; Pennings et al. 2005).

POLLUTION AND TRAMPLING

Pollution with heavy metals, oil, pesticides, and persistent organic pollutants (POP, such as PAH, PCB, and DDT) is relatively low compared to more populated and heavily industrialized Northern Hemisphere countries (Williamson et al. 2003). Heavy metals have been discharged from mining operations, tanneries, fertilizer work, and other industries primarily in the period from 1890 to 1960. Today, many point source pollutions have been cleaned up or routed through sewage treatment plants, and surface sediment concentrations are lower than in the past because pollutants have been diluted or buried by less polluted sediments (Williamson et al. 2003). Contemporary sources of heavy metals are mainly urban storm water runoff, industrial spills, boating and antifouling paints, and geothermal discharges (Roper, Thrush, and Smith 1988; Taylor and Smith 1997). Pollution by toxic organic chemicals is also considered relatively low in New Zealand, with slowly degrading organochlorine pesticides mainly being applied in agriculture between 1940 and 1970, but today phased out by legislation. For example, use of PCB has been illegal since 1995 and its pollution effect is therefore diminishing as residuals are transformed and/or buried. In contrast, it is expected that polycyclic aromatic hydrocarbons released from combustion of fossil fuel and from oil spills will increase in importance due to continued coastal urbanization and an increasing demand for fossil fuels (Williamson et al. 2003).

The ultimate fate of heavy metal and POP discharges is incorporation into in- and offshore sediments, and data on metal and POP levels in estuarine sediments are available both in regional council reports and refereed literature (Roper et al. 1988), but little is known about concentrations in New Zealand salt marsh sediment. Despite most metals and POP showing increased levels in near-shore sediments compared to preindustrialization levels, it has been concluded that compared to impacted overseas areas, New Zealand contamination is relatively low (although highly impacted sites have been found around harbors; Williamson et al. 2003). Still, POPs from urban storm waters and agricultural runoff will continue to enter estuaries and salt marshes in the future. Accreting salt marshes are generally considered sinks for these pollutants (Leendertse, Scholten, and van der Wal 1996), although resuspension events during storms can still transport contaminated sediments out of the marsh. A net pollutant loss is particularly expected in wave-exposed retreating salt marshes (Swales, MacDonald, and Green 2004; Swales, Ovenden, MacDonald, Lohrer, et al. 2005) or salt marshes disturbed by human or livestock trampling. In general, we expect elevated levels of pollutants in New Zealand marshes, but we are not aware of data that verify this or of New Zealand field studies about how marsh structure or functions are affected by pollutants. Indeed, typical salt marsh invertebrates (*H. crassa*, *A. crenata*, and *P. estuarinus*) were found to be relatively resistant to the herbicide mixture dalapon/weedazol in laboratory toxicity tests (Gillespie 1989). Still, it is too simplistic to extrapolate the laboratory LC50 tests to *in situ* food web structures and community interactions. Nevertheless, the regional managers consider pollution and pesticide runoff to be only medium threats to the salt marsh health (average 2.0 [2, 2, 2, ?, 2, ?] and 1.8 [1, 2, 2, ?, 2, ?], respectively).

Dumping of human rubbish is a much more visual pollutant. Dumping is illegal and also considered a medium threat to salt marshes (average 2.0 [1, 2, 3, 2, 2, 2]). Ecological effects are probably minor as long as the dumping are of small quantities and do not contain degrading toxic substances, but the aesthetic impacts are disproportionately large, particularly because marshes typically are considered “wild nature” primarily visited by naturalists. We are not aware of any data that show the extent or effects of rubbish dumping in New Zealand marshes. Of potential more harm is the usage of salt marshes for recreational vehicles and trampling where it can take many years for a marsh to recover (Adam 2002). These negative effects are partly mitigated in marshes managed by the Department of Conservation by designated boardwalks, information boards, and special legislation. Again, we are not aware of any studies that have quantified the extent or impacts of these stressors, but it is considered a medium to high salt marsh threat by the regional managers (average 2.4 [?, 2, 2, 3, 3, 2]).

BIOLOGICAL INVASIONS

Almost half of the New Zealand salt marsh plants are nonnative. This matches the general level of nonnative angiosperms where 2,000 nonnative species persist without human aid, compared to 2,300 native angiosperms (Holland 2001). Of all the nonnative plant species, around two hundred are considered pests that pose a risk to native plants. Of the nonnative salt marsh species, *Spartina anglica* and *S. alterniflora* in particular are considered highly invasive. *S. alterniflora* is mainly found in North Island estuaries, whereas *S. anglica* can be found occasionally in estuaries throughout the country (Partridge 1987; Swales, Ovenden, MacDonald, Lohrer, et al. 2005). *Spartina* was deliberately planted in New Zealand from the beginning of the twentieth century to reduce erosion and claim land, and the first scientific reports focused on these "advantages" ("converting mudflats to productive grass lands"; Allan 1924, 1930; Harbord 1949). From the late 1960s, this viewpoint was challenged, and researchers began to emphasize problems associated with mudflat destruction and alteration of native salt marsh communities (Bascand 1968; Bascand 1970). Today, *Spartina* is considered a pest species, and most regional councils have implemented eradication programs, typically spraying with herbicides (Lee and Partridge 1983; Partridge 1987; Jamieson 1994; Roper et al. 1996; Shaw and Gosling 1996, 1997; Turner and Hewitt 1997; Swales, Ovenden, MacDonald, Burt, et al. 2002; Swales, MacDonald, and Green 2004; Swales, Ovenden, MacDonald, Lohrer, et al. 2005). Despite many reports on *Spartina* in New Zealand (see previously listed references), we are not aware of any scientific studies that test for effects of *Spartina* on native salt marsh species.

Other common nonnative plants include *Paspalum vaginatum*, *Agrostis stolonifera*, *Atriplex prostrata*, *Festuca arundinacea*, *Juncus acutus*, *J. gerardii*, *Plantago coronopus*, *Plantago australis*, *Puccinellia distans*, and *P. fasciculata* (it is sometimes argued that *Cotula coronopifolia* is nonnative, but until this has be rigorously established, we consider it a native) (Johnson 1989; Wilson et al. 1996; Graeme and Kendal 2001; Haacks and Thannheiser 2003; Shaw and Allen 2003). Several of these species can be locally abundant (Graeme and Kendal 2001). The list of salt marsh invaders continues to grow, with, for example, the recent detection of *Limonium companyonis* in the Avon-Heathcote Estuary (Heenan et al. 1999; McCombs and von Tippelskirch 2004). It is interesting that despite New Zealand managers and scientist being highly aware and respondent to problems associated with nonnative species, such as *Spartina* eradication programs (Forrest, Taylor, and Hay 1997; Shaw and Gosling 1997; Taylor and Smith 1997; Clout 1999; Jaya, Moradb, and Bel 2003; Hewitt et al. 2004) only a single study has quantified the abundance of nonnative salt marsh plants as a whole (Wilson et al. 1996), and no studies have tested for impact of nonnatives on the distribution and performance of native salt marsh species. This high awareness of problems with nonnative invasive species is also reflected in the high rankings given by the regional managers (average 2.8 [2, 3, 3, 3, ?, 3]). Little is known about nonnative animals in salt marshes, but it is possible that introduced rabbits graze on salt marsh seedlings (Thomsen et al. 2005). Again, controlled experiments are called for, such as mammal exclusion and nonnative plant removals.

CLIMATE CHANGE

Climate change is primarily associated with anthropogenic pollution with infrared-absorbing gases (International Panel on Climate Change [IPCC] 2001). Projected climatic changes include rising temperature and sea levels, increased evapotranspiration, higher frequency and/or intensities of storms, and altered precipitation patterns (Michener et al. 1997; Bell et al. 2001; IPCC 2001). Most of these effects are likely to stress salt marshes (although higher temperature and carbon dioxide levels may stimulate primary production; van de Staaij et al. 1993), for example, by increasing immersion and mangrove competition at low elevations and hindrance to upward expansion by stop banks, sea walls, causeways, and other urban structures (coastal squeeze). However, the overall community effects in specific marshes are expected to be complex and to depend on latitude, specific community structure, and other local environmental stressors (Vestergaard 1997; Donnelly and Bertness 2001; Simas, Nunes, and Ferreira 2001; Bertness and Ewanchuk 2002). In New Zealand, potential coastal climatic changes and impacts have been reviewed (relative sea-level rise in New Zealand is expected to be 0.14 to 0.18 meter by 2050 and 0.31 to 0.49 meter by 2100), resources allocated, and research encouraged to provide information for future coastal management (National Science Strategy Committee 2000; Bell et al. 2001). On a more "salt marsh–positive" note, it is expected that increased salinization of lands around estuaries will transform terrestrial land into salt meadows (including previously reclaimed land), salt marshes, and eventually estuary mudflats, particularly in areas without seawalls or stop banks (planned retreat strategy; Bell et al. 2001). Again, no specific studies have yet related climatic changes to New Zealand salt marsh ecology.

In addition to global warming, the degradation of the ozone hole is another human-induced "diffuse" global climate change. The degradation of the ozone layer, mainly due to CFC emissions, has caused an increase in ultraviolet-B (UV-B) radiation, particularly in the Southern Hemisphere as the ozone layer is thinnest over Antarctica. Thus, New Zealand has high UV-B radiation (Howard-Williams et al. 1997; McKenzie, Connor, and Bodeker 1999), partly contributing to a high incidence of human skin cancer (about 1,800 melanoma cases and 45,000 nonmelanoma cases confirmed by laboratory tests, plus 20,000 nonmelanoma skin cancer cases treated without laboratory tests; Cancer Society of New Zealand 2004). There are no New Zealand studies that link high UV radiation to salt marsh ecology, but given quantified adverse growth effects on South American *Sarcocornia* plants (Bianciotto et al. 2003) and Dutch *Elymus athericus* (van de Staaij et al. 1993), we expect similar negative effects in New Zealand. Due to the complexity of the problems associated with multifactorial climate changes, it is obviously difficult to link these stressors to salt marsh performance (Caldwell and Flint 1994) and is therefore not surprising that the regional managers considered these threats likely, but generally with unknown consequences to local salt marshes (global warming average 2.5 [?, 2, 3, ?, ?, ?] and ozone average 2 [?, ?, 2, ?, ?, ?]).

AUSTRALIAN SALT MARSHES, MANAGEMENT, AND HUMAN THREATS

SALT MARSHES OF AUSTRALIA

Unlike New Zealand, Australia is relatively flat, with low runoff. Rivers are mostly small, and a particular feature of southern Australian coasts is the number of intermittently open coastal lagoons (Brearley 2005).

Northern Australia is tropical, with a strongly seasonal rainfall pattern with a pronounced summer maximum. Tropical coasts are exposed to major storms (cyclones) in mid- to late summer. In more temperate latitudes, rainfall is generally lower than in the tropics and more evenly distributed throughout the year, although in parts of southern Australia, there is a winter rainfall-maximum Mediterranean climate. There are extensive stretches of arid coastline, with no permanently discharging rivers, but where topography permits, there are salt marshes on these apparently inhospitable coasts.

In northern Australia, the upper part of the intertidal zone often takes the form of extensive hypersaline flats, with vascular plants either very sparse or absent, and the sediment surface encrusted with microalgae and cyanobacteria (Saenger et al. 1977). The lower part of the intertidal zone in many mainland estuaries and sheltered soft open coasts is occupied by mangroves. The most species-rich mangroves occur in northeast Queensland, with species richness declining both westward and southward. At the highest latitudes, only a single species, *Avicennia marina*, occurs. There is also a marked decline in stature with increasing latitude. In northeast Queensland, mangroves may be up to thirty meters or more; but in southern Victoria *Avicennia*, they take the form of low shrubs about one meter tall. There are no mangroves in Tasmania.

The total area of salt marsh in Australia is not known with any degree of certainty. Adam (1995) used the estimate presented by Bucher and Saenger (1991) of between thirteen thousand and fourteen thousand square kilometers (see also Bucher and Saenger 1989, 1994). The majority of this area occurs in tropical Australia (Queensland, the Northern Territory, and northern Western Australia). Bucher and Saenger (1994) point out that their inventory was not able to distinguish between "true salt marsh, with its flora of salt-tolerant herbs and grasses, and the bare saline clay pan which may have a seasonal plant cover." The difficulty of distinguishing the two is affected not only by seasonal variation but also by the timing of image accession relative to tidal flooding and rainfall events. There are very extensive, frequently hypersaline, upper intertidal salt flats (a habitat referred to elsewhere as *sabkha*) in northern Australia, probably at least as extensive in area as the permanently vegetated salt marshes. It is not worth investing too much effort for purposes of inventory into distinguishing between salt marsh and salt flat as both are habitats of conservation value that are vulnerable to anthropogenic impacts. Indeed, it would be unfortunate if, in an attempt to protect salt marsh, developments were relocated to salt flats.

Even in temperate Australia, determination of the area of salt marsh is affected by the spatial scale of investigation, the nature and quality of the images assessed, and differences in interpretation between operators. In NSW, the frequently cited figure for total salt marsh area of 5,716 hectares was derived by West et al. (1985) from air photo interpretation and ground truthing of some 130 estuaries; but a later study by Williams et al. (1998) identified some 950 water bodies discharging from NSW into the Tasman Sea, and they suggested that because of the dynamic nature of the NSW coast (as a result of both human and natural influences), "The actual number of water bodies present on the NSW coast may never be known." More recently, Williams (2006) has highlighted substantial differences in both total area and number of recognized salt marsh sites between studies of the same estuaries utilizing different methodologies so that meaningful detection of trends is difficult.

(A)

Unlike mangroves, in which species diversity decreases with increasing latitude, in Australian salt marshes, species richness is highest in Tasmania and lowest in the tropics. General accounts of the flora and vegetation of Australian salt marshes are provided by Saenger et al. (1977) and Adam (1990). More detailed studies are reported in Kirkpatrick and Glasby (1981); Bridgewater (1982); Adam, Wilson, and Huntley (1988); Cresswell and Bridgewater (1998); Kirkpatrick and Harris (1999); and Jaensch (2005). The pattern of regional variation in Australian salt marshes can be placed in the context of a global scheme (Adam 1990) and particularly in temperate latitudes, where there are similar patterns of species and generic occurrence across Gondwana continents (southern South America, South Africa, Australia, New Zealand) (Adam 1990). Most of the species or genera on temperate Australian marshes are also found in New Zealand, although there are some striking differences. The three species of *Wilsonia* (Convolvulaceae) are endemic to Australia, and the important grasses of southern Australian salt marshes, *Sporobolus virginicus* and *Distichlis distichophylla*, are notably absent from New Zealand. Thus, the native grasslands, which can be extensive in Australian salt marshes, are not a feature across the Tasman. The characteristic shrub of the upper marsh in New Zealand, *Plagianthus divaricatus*, does not have an obvious parallel in Australia.

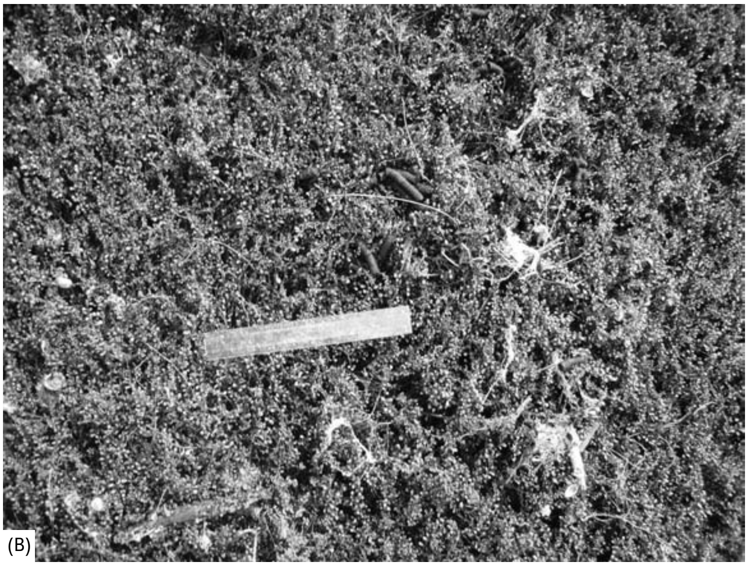

FIGURE 18.5 Black swans (*Cygnus atratus*) can occasionally be found in salt marshes. Swans are common in Pauatahanui inlet, New Zealand, where they are often seen resting (and sometimes foraging) on *Samolus repens* patches (A). The numerous swan droppings in the marsh (B) provide a visible fertilization impact that likely facilitates marsh growth.

There are relatively few studies of fauna on Australian salt marshes. Tropical salt marshes provide habitat for the saltwater crocodile (*Crocodylus porosus*), which may be a disincentive for fieldwork! Macropods (kangaroos and wallabies) are obvious grazers at many sites, and on urban fringes salt marsh may provide an important refuge for macropods. The native water rat (*Hydromys chrysogaster*) and the false water rat (*Xeromys myoides*) both utilize salt marsh and mangrove habitats. A recent study has shown that temperate salt mashes are likely to be important foraging sites for bats (Laegdsgaard, Monamy, and Saintilan 2004). More study, including of tropical marshes, is required, but the previously noted findings strengthen the case for salt marsh conservation. Salt marshes provide important high-tide roosting for migratory wading birds, and a number of sites have been incorporated in conservation reserves for this reason. Unlike many Northern Hemisphere sites, Australian salt marshes are not visited by migratory waterfowl. Australian ducks are nomadic rather than migratory and are predominantly birds of the inland. The chestnut teal (*Anas castanea*) occurs in both inland and coastal sites, but of Australian ducks, it is the species that shows a preference for coastal habitats including salt marsh, although it is rarely found in large flocks. Black swans (*Cygnus atratus*) frequent coastal lagoons, particularly during the molt, and feed primarily on sea grasses, but they can occasionally be found on salt marshes (fig. 18.5). The Cape Barren Goose (*Cereopsis novaehollandiae*) has a restricted distribution in southern Australia, but population numbers have been increasing, and locally it can exert heavy grazing pressure on vegetation, including salt marsh. One of Australia's rarest birds, the orange-bellied parrot (*Neophema chrysogaster*), overwinters on the mainland on salt marshes, where it feeds on the seeds of chenopod (Cousins 1989; Garnett and Crowley 2000). A number of small passerine birds utilize salt marsh habitat including cisticolas (*Cisticola juncidis*, *C. exilis*) and chats (*Ephianura albifrons* and the very rare *E. crocea* var. *macgregori*).

The invertebrate fauna of salt marshes includes both marine and terrestrial components, of which the marine element has been better studied, even though the total number of studies is small. A diversity of marine mollusks (Robinson and Gibbs 1982) and crabs (Mazumder et al. 2005a, 2005b) are characteristic of salt marshes. Given the distribution of salt marsh in the upper intertidal above the mangrove zone, and the paucity of creek and pan systems in most Australian salt marshes (Adam 1997), until recently the value of Australian salt marshes as fish habitat was assumed to be small. Several studies (Morton, Pollock, and Beumer 1987; Connolly, Dalton, and Bass 1997; Mazumder et al. 2005a, 2005b) have dispelled this myth. Although utilization by fish is temporally limited, a diversity of fish species, including a number of commercial importance, has now been shown to be present in salt marsh during flooding tides (Mazumder et al. 2005a, 2005b).

MANAGEMENT OF AUSTRALIAN SALT MARSHES

Tenure and regulatory control of Australian salt marshes is complex. In general, freehold and leasehold titles extend to high water (in most cases defined as mean high water springs [MHWS]), so that many salt marshes straddle the boundary between private and Crown land. There are, however, numerous exceptions where, because of historical accidents or inaccurate early surveys, private ownership continues below the current MHWS.

Even where salt marsh is in private ownership, there may be controls on development applied through the planning system. For example, in NSW, all salt marsh is listed as an Endangered Ecological Community under the Threatened Species Conservation Act, and many sites are individually mapped under State Environment Planning Policy 14—Coastal Wetlands. The Fisheries Management Act in NSW and the Fisheries Act in Queensland have provisions that give the fisheries agencies considerable control over any development that may occur in salt marshes.

Around Australia, a number of salt marshes are incorporated in conservation reserves under the control of national park agencies or in some form of aquatic or marine reserve managed by fisheries agencies. Some sites are part of wetlands listed under the Ramsar Convention on Wetlands. Despite the diversity of state laws and policies affecting salt marshes, local government also has considerable influence, not only in approving (or refusing) some development within marshes but, more important, through control of development in the catchment. The research permit system vary between states. In NSW, salt marshes are generally classified as Endangered Ecological Communities, and a permit is needed from the Department of Environment and Conservation. In addition, if the research impinges on fisheries matters, a permit from the Department of Primary Industry would be required.

ANTHROPOGENIC THREATS

DEVELOPMENT

European Australia is essentially a coastal nation, the overwhelming majority of the population living on, or close to, the coast. All state capital cities are situated on estuaries (only the national capital, Canberra, is inland). This means that the coast and estuaries have been very heavily impacted by development (Turner et al. 2004). The majority of impacts has been in temperate and subtropical regions, the regions where salt marsh is more floristically rich, but where it would also have been relatively limited in extent. The much more extensive tropical salt marshes have received relatively little direct impact, although there are some tropical coasts where there has been extensive port and industrial development, most notably in northwest Western Australia (Turner et al. 2004), where in addition to development of ports and associated infrastructure to export iron ore, salt production facilities have resulted in loss of salt flats and salt marsh.

In southern Australia, much of the loss of salt marsh has been through infilling for urban and industrial development (e.g., Fotheringham 1994). Unlike northern Europe, where the reason, over centuries, for much reclamation has been for agriculture, farming, although it has driven some reclamation, has not been such a major factor in Australia.

From the 1970s onward, a major activity resulting in salt marsh loss, often in locations that at time of development were distant from major cities, was the development of canal estates (see illustrations in Turner et al. 2004; Brearley 2005). Many canal estate developments were controversial and viewed by many conservation groups as the epitome of the worst excesses of real estate entrepreneurs. In response to public concerns, planning regulations in NSW currently prohibit any new canal estates, but construction is still possible in other states (e.g., Western Australia; personal observation).

Australia is a dry continent, so that management of water resources has been a major preoccupation of governments for the last two hundred years. For much of this period, concepts of ecological sustainability were not considered; even today, when environmental concerns are high, we still have the legacy and infrastructure of past decisions. Freshwater flows into estuaries have been affected both by dam construction and abstraction of groundwater.

Tidal flow of saline water into upper estuaries and tributary creeks is often restricted through construction of weirs, floodgates, and other barriers. In NSW, Williams and Watford (1996) identified more than four thousand such barriers. Any areas of salt marsh cutoff from tidal influence by such barriers are likely to change to a more brackish vegetation type, although individual plants of more halophytic species such as *Sarcocornia quinqueflora* or *Suaeda australis* may survive at low density among tall *Phragmites*, *Bulboschoenus*, or *Schoenoplectus* for many years (personal observation), suggesting that removal of the barriers may permit reestablishment of salt marsh.

Australia's largest river system is the Murray-Darling, with a catchment covering 14 percent of the continent. Water from the Murray-Darling supports the population in the catchment and three-quarters of Australia's irrigated farmland (Turner et al. 2004). In addition, water is transported out of the catchment to supply Adelaide (the capital of South Australia) and the major industrial centers in South Australia. Flow to the mouth of the river has been reduced by about 80 percent compared to the pre-European period (Turner et al. 2004) such that the mouth is now frequently closed. As a result of barrage construction and reduced flow, the area of the estuary has been reduced by 90 percent. Not only is the volume of freshwater reaching the estuary reduced, but the timing of its arrival has been modified as a result of the management of release from upstream impoundments (Turner et al. 2004). The estuaries of all managed rivers will have experienced modified freshwater inputs, although less dramatic than those in the Murray. The impacts on salt marshes have, however, been little studied.

Although inputs of water into estuaries may have been reduced, within urban areas or adjacent to major roads, inputs into individual marshes may have increased due to storm water discharge. Except in the most recently constructed examples, there are rarely gross pollutant traps. Storm water is thus a source of a range of pollutants, including heavy metals, oils, pesticides, and larger plastic pieces and other rubbish. Discharge also affects the salinity regime within the marsh. In many locations, storm water pipes are associated with the establishment of patches of *Phragmites* or *Typha* within marshes (see Zedler, Paling, and McComb 1990). Spread of *Phragmites* changes the structure and diversity of salt marshes, but, unlike the situation in New Zealand (McCombs 2004), it is not generally treated as a weed in Australia.

EUTROPHICATION

The Australia landscape has been transformed over the last two hundred years by the development of agriculture. On a continent of largely nutrient-poor soils, this has involved extensive use of fertilizers. While the nutrient loads now entering estuaries have undoubtedly increased, any consequences for salt marshes have been little studied. The most dramatic impacts have been in southwestern Western Australia, particularly in the Peel-Harvey Estuary (Deeley and Paling 1998; Hodgkin and Hamilton 1998; Brearley 2005), where, as a result of considerable increases of nutrients but particularly of phosphorus, algae productivity greatly increased, smothering fringing salt marshes and, on dying, producing a malodorous mess that substantially reduced the amenity of the area. This problem has been addressed not only through education and better practices of farmers in the catchment, but by major engineering works to construct a new opening to the estuary (the Dawesville Cut), altering the circulation and flushing patterns (Hodgkin and Hamilton 1998). In addition to fertilizer runoff, there may be runoff of agricultural chemicals, but, again, there has been no study of possible impacts on salt marsh biota.

OIL POLLUTION

Salt marshes are potentially at risk from oil or chemical spills. These spills could contribute to low background levels of pollution (e.g., from routine port operations and refueling) or be a consequence of major accidents. In addition to shipping accidents, road or rail transport accidents resulting in oil or chemicals entering the storm water system could impact salt marshes.

Australia has been fortunate to date in not having suffered major incidents affecting salt marshes. In Botany Bay, where Sydney's major oil refinery is situated, there have been a number of shipping incidents resulting in minor (by world standards) oil spills that have had adverse impacts on mangroves (Anink et al. 1985; Allaway 1992), although tidal conditions at the time of the incidents meant that oil did not extend into salt marshes. Australia has comprehensive contingency plans for responding to oil spills (Australian Maritime Safety Authority [AMSA] 1993) that recognize the importance and sensitivity of salt marsh. Booms, skimmers, other equipment, and chemicals are available for rapid deployment in the event of any spill.

Shipping operations present other potential hazards to salt marshes, including the possibility of introduction of exotic species in ballast water. Although many ballast water introductions have been recorded in Australian waters, no species that might have a direct impact on salt marshes has been reported so far. Australia is active both nationally and internationally in moving to reduce to overall environmental impacts of shipping (Australia and New Zealand Environment and Conservation Council [ANZECC] 1996).

HEAVY METALS

Australia's major industrial centers are located close to estuaries, and as a result, there have been considerable increases in the amounts of bioavailable heavy metals in estuaries. (This has arisen from atmospheric fallout, runoff, and, in some cases, deliberate discharge and dumping.) However, while heavy metals in sediments have been documented (Chenhall et al. 2004), and uptake by salt marsh plants has been described (Chenhall, Yassini, and Jones 1992), the ecosystem consequences have not been explored. The various state and national environmental protection agencies have established guideline concentrations for a range of pollutants in estuarine and marine waters (ANZECC and Agricultural and Resource Management Council of Australia and New Zealand [ARMCANZ] 2000), largely based on human health considerations, with little knowledge about the responses of native biota.

Chenhall et al. (2004) showed that surface sediments in Lake Illawarra had higher levels of heavy metals, reflecting anthropogenic inputs. However, under the ANZECC and ARMCANZ (2000) guidelines, most sites were classified as low risk, although there were some "hot spots" with much higher values. In Lake Macquarie (another large coastal lagoon in NSW), zinc and lead levels very much higher than those in Lake Illawarra have been recorded at Cockle Creek (Roy and Crawford 1984; Batley 1987), associated with smelter operation. Similar stories could be told for other long-established industrial regions, but many of Australia's estuaries do not have heavy industry.

INVASIVE SPECIES

The salt marsh flora of Australia contains a large number of introduced species, although probably not as high a percentage as in New Zealand. Many of these are found in the upper salt marsh fringe, particularly on dry sandy coasts (Bridgewater and Kaeshagen 1979); and although there have been no detailed studies, they do not appear to pose a serious threat to the integrity of the ecosystem. A few more widespread species give rise to more serious concern. As in New Zealand, *Spartina anglica* is of major concern in Victoria and Tasmania. The history of the introduction of *Spartina* to Australia was documented by Boston (1981). In Tasmania, the original plantings were for mudflat stabilization (Phillips 1975; Pringle 1982) and from that perspective were a great success. In Victoria, a particular concern is that at some localities it has spread onto mudflats below the previous seaward limit of mangroves. Control of *Spartina* is now actively practiced, and the basis for control measures are discussed in Rash, Williamson, and Taylor (1996) and Kriwoken and Hedge (2000). The primary control technique in Tasmania is application of herbicide (Fusilade®), supplemented by physical removal and smothering (Tasmanian Department of Primary Industries, Water, and Environment 2002).

In southern Queensland and continuing to spread southward into NSW, a major salt marsh weed is groundsel bush (*Baccharis halimifolia*; Natural Resources, Mines, and Water 2006). An added incentive for the control of this species is its importance as a trigger for hay fever. *Juncus acutus* can displace the native *Juncus kraussii* and form dense monospecific stands in upper and mid–salt marshes (Paul and Young 2006). Pampas grass, *Cortaderia selloana,* is both salt-tolerant and recovers vigorously after fire (personal observation). In NSW, it occurs in a number of salt marshes and has considerable potential for further spread.

A particular problem for the control of weeds in salt marshes is that most herbicides are not routinely approved for use in the intertidal environment, and the research necessary to ascertain toxicity of herbicides (and any accompanying surfactants) to intertidal biota and to the sensitivity of native plants to herbicide application would be expensive and is unlikely, given the very limited sales market, to be carried out by manufacturers.

CLIMATE CHANGE

Except where sea cliffs occur, the Australian coast will be vulnerable to a greenhouse-induced rise in sea level (Short 1988), and salt marshes are likely to be particularly sensitive (Vanderaee 1988). As elsewhere, "coastal squeeze" is likely to occur if the seaward edge of marshes retreats, but the landward boundary is constrained by either natural topography or artificial structures. The widespread spread of mangroves into salt marshes in southeastern Australia (Saintilan and Williams 1999) may in part be an early indication of sea level rise. Sea-level rise will be not the only consequence of the greenhouse effect. Increased carbon dioxide and temperature are likely to alter the performance of individual species, changing the composition of assemblages. Changes in rainfall (both increases and decreases) may have impacts on the composition of upper marsh communities, and changes to the frequency and intensity of cyclones could result in extreme disturbance of some salt marshes in northern Australia. Coastal squeeze will be a particular problem on the developed coastline of southern Australia. In the north, there are long stretches of coast where landward shift may be possible, but the composition of the marshes may still change in response to changes in temperature, carbon dioxide concentration, and freshwater inputs.

FIRE

Australia is a notoriously fire-prone continent. Salt marsh would not, however, be generally considered likely to burn. Nevertheless, upper salt marsh communities dominated by *Juncus kraussii* or

Baumea juncea can carry hot fires. Documentation of fires in salt marshes is poor, and recovery has been little studied, although, anecdotally, vegetative regeneration is slow (as after the 1994 fires that burned both mangroves and salt marsh on the Hawkesbury River; personal observation). In tropical northern Queensland, fire is used as a management tool in *Sporobolus virginicus* grasslands to promote "green pick" for grazing cattle (Anning 1980). Kirkpatrick and Harris (1999) suggest that fire in Tasmanian salt marshes may have caused a decline in the shrubby chenopod *Sclerostegia arbuscula*.

MOSQUITO CONTROL

Mosquito control is becoming an increasingly important issue in the management of Australian salt marshes. A number of mosquito species breed in salt marshes, including *Ochlerotatus vigilax*, *Anopheles hillii*, and *Culex sitiens*, which can act as vectors for a number of arboviruses causing debilitating diseases of humans (Webb and Russell 2006). Currently, the major diseases transmitted in this way are Ross River fever and Barmah Forest disease, and under global warming, their incidence is expected to increase. There is also the potential for malaria to become established. Given the increasing human population living within close proximity to estuaries, there is great pressure on local councils to implement control measures. Although pesticides (Webb and Russell 2006) are still widely used, there is public concern about the possible consequences, and alternative nonchemical approaches have been developed. Runneling is the creation of shallow drainage channels to increase tidal flushing and remove standing water, which is the mosquito larval habitat. Runneling has been widely practiced in Queensland (Dale 1994; Jaensch 2005) and has also been instituted in southwestern Australia (Latchford 1998). Studies to date indicate that the ecological impact of runneling on nontarget species is slight, although whether this would be true if mosquito control in more species-rich higher latitude marshes were required is not known.

SALT PRODUCTION

Australia is one of the world's leading producers of solar salt. The majority of production is in northwestern Australia, but plants have also been developed in southwestern Australia, Victoria Queensland, and South Australia. Evaporation ponds have been constructed on high-level salt flats and on salt marshes (and this has been a major factor in the decline of South Australian salt marshes; Edyvane 1996). Further expansion of the industry could threaten the persistence of salt marsh at some sites.

OFF-ROAD VEHICLES

A major threat to many Australian salt marshes is from off-road recreational vehicles (BMX bicycles, trail bikes, and four-wheel drive vehicles). There is a variety of consequences of passage of vehicles across marshes, including crushing of vegetation and alteration to drainage patterns, with consequent changes to habitats. Examples of damage are illustrated in Jaensch (2005) and Kelleway (2005). The scale of the problem is increasing. Kelleway (2005) has shown that in 1966, the proportion of salt marsh affected by track damage in the Georges River (NSW) was 1.5 percent, but by 1998, this figure had increased to 23.2 percent, notwithstanding that some of the sites were in a national park. Tracks may persist for many years, even when access is prevented.

AQUACULTURE

Aquaculture is a major growth industry internationally. There are a range of possible impacts on salt marshes, directly from loss of sites for construction of ponds, and indirectly from changed water quality from effluent. A number of early aquaculture facilities in Australia involved loss of salt marsh (e.g., construction of prawn ponds on Micalo Island in NSW; personal observation), but the protection now afforded salt marsh through the planning systems makes it unlikely that similar facilities would be approved.

LIVESTOCK GRAZING

Livestock have access to many Australian salt marshes; particularly in the tropical north, salt marsh is regarded as an important resource for rangeland cattle (Anning 1980). In some locations, sheep have access, but there is no practice of intensive grazing by sheep, as was the case in much of northern Europe. Indeed, one attempt to intensively graze sheep, on the Towra Point salt marsh in Botany Bay, was a complete failure, resulting in the burial of carcasses on the site (Holt 1972). In Europe, grazing profoundly changes the composition of salt marsh vegetation. There is little evidence of such impacts in Australia, and in species-poor tropical marshes, there are too few species to ring the changes. Howarth (2002) has made the interesting suggestion that past cattle grazing may have been a factor limiting the invasion of salt marsh by mangroves. Trampling damage is visible in the marshes on which cattle grazing occur.

CONCLUSION AND RECOMMENDATIONS

Few quantitative data exist that directly link anthropogenic threats to Australasian salt marshes. Nevertheless, it is certain that large salt marsh areas have been drained and cleared and that many remaining marshes are under threats from an array of anthropogenic stressors. A lack of quantitative distribution data, such as species-specific biomass/cover values with standard errors, implies that it is difficult to evaluate past and future changes (Gurevitch and Hedges 1999; Osenberg et al. 1999). A further lack of process-oriented manipulative experiments also implies that we have poor knowledge about the processes that control present distribution patterns. These shortcomings cause conservation and restoration effort to some extent to be based on anecdotal and personal experiences and the gray literature (local reports).

We recommend that managers combine remote sensing techniques with ground truth surveys (species identification, stem length, biomass, with standard errors) to establish baseline data on the extent, zones, and borders of salt marshes. Preferably, data should be added on salt marsh fauna, sediment characteristics, and degree of pollution/anthropogenic impacts. Such data should be digitized and entered into a standardized database to allow for easy cross-regional comparisons and for efficient documentation of future large-scale changes (many of these suggestions are currently being implemented in regional councils in New Zealand). It is equally important that research institutions support baseline monitoring data with process-oriented manipulative experiments that test for effects of anthropogenic stressors on salt marsh plant health, community patterns, and ecosystem functioning. For example, the most abundant and ubiquitous plants and animals should be manipulated (inclusion/exclusions) to uncover mechanisms of zonation patterns and assess community impacts, and multiple stressors should be manipulated simultaneously using factorial design to elucidate relative importance of stressors as well as to test for interaction effects (e.g., Bertness 1984, 1985; Bertness and Yeh 1994; Silliman and Zieman 2001; Bertness and Ewanchuk 2002; Silliman and Bertness 2004). Lastly, results obtained from these tasks, should be published in peer-reviewed journals to ensure high-quality data and open access for researchers, and to avoid valuable information remaining hidden in local gray literature.

The threats facing Australasian salt marshes are similar to those operating throughout the world (Adam 2002). Increased awareness of the ecological values of salt marshes has led to the introduction of measures to prevent actual loss of sites. These are likely to be effective in some circumstances, but the controls are rarely absolute and have escape clauses to permit development where the economic value of a project is perceived to be greater than the loss of environmental value, or where the national interest is thought to be best served by development. Further losses for major infrastructure, such as roads or ports, are likely to occur. However, any such development will probably include mitigation measures, such as rehabilitation of other damaged marshes or creation of new marshes.

Research on restoration techniques is in its infancy in Australasia, but it will become increasingly necessary as the continuing population increase causes growing stress on coastal salt marshes. This highlights the need for strong management. Part of the solution will involve development planning and public education, but control of insect populations is likely to be called for (particularly in Australia). Given political reality, such calls will be heeded, but hopefully the methods used will minimize collateral damage to salt marsh and estuarine ecosystems.

At the site-specific level, there is likely to be a tyranny of small decisions, causing attrition of the salt marsh resource. A boat ramp here, a coastal cycle track there are likely to be seen as socially desirable and having minimal environmental impact, although the cumulative impact of many such decisions could be considerable. Decisions made by individuals to carry out activities such as illegal dumping or using off-road vehicles in salt marshes could also have large cumulative impacts. Given increased visitation to northern Australia, damage from off-road vehicles could increase stress on large tropical salt flats and salt marshes significantly. Addressing these individually minor but cumulatively large impacts, both approved and illegal, will require heightened public awareness and action.

Management of changes to hydrology and of pollution, both point source and diffuse, will require action at the catchments scale. The major challenge will be to address the continuing consequences of historic decisions and actions. The effects of human modification of the atmosphere will be felt by salt marshes globally. The affects include global warming, changes to patterns of storminess, rainfall and fire, increases in atmospheric carbon dioxide concentration, and increase in UV radiation due to a reduction in upper atmospheric ozone. The consequences are difficult to predict with certainty due to complex interaction between the various affects. In southern Australia and many estuaries in New Zealand, sea-level rise is likely to create coastal squeeze, resulting in salt marsh loss. In Australian tropical north and remote low-populated New Zealand estuaries, there will be greater opportunities for salt mash area to be sustained by landward retreat. Changes in temperature, rainfall, and storm occurrence and an increase in carbon dioxide will give rise to species-specific response, so that future salt marsh communities may have different compositions from those of today. Particularly in southern Australia and northern New Zealand, there is likely to be a continuance of the trend in the expansion of mangroves into salt marsh (Saintilan and Williams 1999).

In short, salt marshes in New Zealand and Australia are, like salt marshes throughout the world, under continued pressure as coastal human populations and economies continue to increase. However, with heightened attention on monitoring, regulation, planning, protection, research, and restoration, it should be possible to ensure that these important sea–land transition ecosystems remain abundant and healthy.

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REFERENCES

Adam, P. 1990. *Saltmarsh Ecology*. Cambridge: Cambridge University Press.

———. 1995. Saltmarsh. Pages 97–105 *in* L. P. Zann and P. Kailola (eds.), *The State of Marine Environment Report for Australia*. Canberra: Department of Education Science and Training, Technical Annex 3, State and Territory Issues.

———. 1997. Absence of creeks and pans in temperate Australian salt marshes. *Mangroves and Salt Marshes* 1: 239–241.

———. 2002. Saltmarshes in a time of change. *Environmental Conservation* 29: 39–61.

Adam, P., N. C. Wilson, and B. Huntley. 1988. The phytosociology of coastal saltmarshes in New

South Wales, Australia. *Wetlands (Australia)* 7: 35–84.
Airoldi, L. 2003. The effects of sedimentation on rocky coast assemblages. *Oceanography and Marine Biology Annual Review* 41.
Allan, H. H. 1924. *Spartina* grass and its introduction into New Zealand. *New Zealand Journal of Science and Technology* 1: 253–256.
———. 1930. *Spartina townsendii*: A valuable grass for reclamation of tidal mudflats. II Experience in New Zealand. *New Zealand Journal of Agriculture* 40: 189–196.
Allaway, W. G. 1992. Mangrove die-back in Botany Bay. *Wetlands (Australia)* 2: 2–7.
Andresen, H., J. P. Bakker, M. Brongers, B. Heydemann, and U. Irmler. 1990. Long-term changes of salt marsh communities by cattle grazing. *Vegetatio* 89: 137–148.
Anink, P. J., D. E. Roberts, D. R. Hunt, and N. F. Jacobs. 1985. Oil spill in Botany Bay: Short-term effects and long-term implications. *Wetlands (Australia)* 5: 32–41.
Anning, P. 1980. Pastures for Cape York Peninsula. *Queensland Agricultural Journal* (March–April): 148–171.
Atkinson, I. A. E., and R. M. Greenwood. 1989. Relationships between moas and plants. *New Zealand Journal of Ecology* 12: 67–96.
Auckland Regional Council. 2000a. *Coastal Planting Guide No. 1*. Auckland: Author.
———. 2000b. *Wonderful Wetlands: Wetland Restoration and Planting Guide for the Auckland Region*. Auckland: Author.
Australia and New Zealand Environment and Conservation Council. 1996. *Working Together to Reduce Impacts from Shipping Operations: ANZECC Strategy to Protect the Marine Environment*. Canberra: Author.
Australia and New Zealand Environment and Conservation Council and Agricultural and Resource Management Council of Australia and New Zealand. 2000. *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. Canberra: Authors.
Australian Maritime Safety Authority. 1993. *Review of the National Plan to Combat Pollution of the Sea by Oil*. Canberra: Author.
Bascand, L. D. 1968. The control of *Spartina* species. *Proceedings of the Twenty-first New Zealand Weed and Pest Control Conference*: 108–118.
———. 1970. The roles of *Spartina* species in New Zealand. *Proceedings of the New Zealand Ecological Society* 17: 33–40.
Batley, G. R. 1987. Heavy metal speciation in waters, sediments and biota from Lake Macquarie, New South Wales. *Australian Journal of Marine and Freshwater Research* 38: 591–606.
Bell, R. G., T. M. Hume, and D. M. Hicks. 2001. Planning for climate change effects on coastal margins. Report prepared for the Ministry for the Environment as part of the New Zealand Climate Change Program, Ministry for the Environment, Wellington.
Bergin, D. O. 1994. Performance of transplanted indigenous salt marsh species, Maketu estuary. New Zealand Forest Research Institute Contract Report, Wellington.
Bertness, M. D. 1985. Fiddler crab regulation of *Spartina alterniflora* production on a New England salt marsh. *Ecology* 66: 1042–1055.
Bertness, M. D., and P. J. Ewanchuk. 2002. Latitudinal and climate-driven variation in the strength and nature of biological interactions in New England salt marshes. *Oecologia* 132: 392–401.
Bertness, M. D., P. J. Ewanchuk, and B. R. Silliman. 2002. Anthropogenic modification of New England salt marsh landscapes. *Proceedings of the National Academy of Sciences of the USA* 99: 1395–1398.
Bertness, M., R. B. Silliman, and R. Jefferies. 2004. Salt marshes under siege. *American Scientist* 92: 54–61.
Bianciotto, O. A., L. B. Pinedo, N. A. San Roman, A. Y. Blessio, and M. B. Collantes. 2003. The effect of natural UV-B radiation on a perennial *Salicornia* salt-marsh in Bahia San Sebastian Tierra del Fuego, Argentina: A 3-year field study. *Journal of Photochemistry and Photobiology B: Biology* 70: 177–185.
Bos, D. 2002. Grazing in coastal grasslands. Brent Geese and facilitation by herbivory. Unpublished PhD diss., University of Groningen, Holland.
Bos, D., J. P. Bakker, Y. De Vries, and S. Van Lieshout. 2002. Long-term vegetation changes in experimentally grazed and ungrazed back-barrier marshes in the Wadden Sea. *Applied Vegetation Science* 5: 45–54.
Boston, K. G. 1981. The introduction of *Spartina townsendii* (s.l.) to Australia. *Occasional Paper, Melbourne State College* 6: 1–57.
Brearley, A. 2005. *Ernest Hodgkin's Swanland: Estuaries and Coastal Lagoons in South Western Australia*. Perth: University of Western Australia Press.
Brewer, J. S. 2003. Nitrogen addition does not reduce belowground competition in a salt marsh clonal plant community in Mississippi (USA). *Plant Ecology* 168: 93–106.
Bridgewater, P. B. 1982. Phytosociology of coastal salt-marshes in the Mediterranean climatic region of Australia. *Phytocoenologia* 10: 257–296.

Bridgewater, P. B., and D. Kaeshagen. 1979. Changes induced by adventive species in Australia plant communities. Pages 561–579 *in* O. Wilmanns and R. Tuxen (eds.), *Werden und Vergehen von Pflanzengesellschaften*. Vaduz, Lichtenstein: Cramer.

Bucher, D. J., and P. Saenger. 1989. An inventory of Australian estuaries and enclosed marine waters. Report and database prepared for the Australian Recreational and Sport Fishing Confederation, Australian National Parks and Wildlife Service and the Centre for Coastal Management, Lismore, New South Wales.

———. 1991. An inventory of Australian estuaries and enclosed marine waters: An overview of results. *Australian Geographical Studies* 29: 370–381.

———. 1994. A classification of tropical and subtropical Australian estuaries. *Aquatic Conservation: Marine and Freshwater Ecosystems* 4: 1–19.

Caldwell, M. M., and S. D. Flint. 1994. Stratospheric ozone reduction, solar UV-B radiation and terrestrial ecosystems. *Climatic Change* 28: 375–394.

Cancer Society of New Zealand. 2004. *The Costs of Skin Cancer to New Zealand*. Wellington: Author.

Chapman, V. J. 1974. Salt marshes and salt deserts of the world. *Cramer, Lehre*: 310–331.

Chapman, V. J., and J. W. Ronaldson. 1958. The mangrove and saltmarsh flats of the Auckland Isthmus. New Zealand Department of Scientific and Industrial Research Bulletin 125, Auckland.

Chenhall, B. E., B. G. Jones, C. R. Gloss, M. O'Donnell, M. Payne, M. Murrie, D. Garnett, and H. Waldron. 2004. Trace metals in sediments from Lake Illawarra, New South Wales Australia. *Wetlands (Australia)* 21: 198–208.

Chenhall, B. E., I. Yassini, and B. G. Jones. 1992. Heavy metal concentrations in lagoonal saltmarsh species, Illawarra region, southeastern Australia. *Science of the Total Environment* 125: 203–225.

Clout, M. N. 1999. Biodiversity conservation and the management of invasive animals in New Zealand. Pages 349–361 *in* O. T. Sandlund (ed.), *Invasive Species and Biodiversity Management*. Dordrecht: Kluwer Academic.

Connolly, R. M., A. Dalton, and D. A. Bass. 1997. Fish use of an inundated saltmarsh flat in a temperate Australian estuary. *Australian Journal of Ecology* 22: 222–226.

Cousins, J. 1989. *Issues in Geography*. Marrickville, Australia: Science Press.

Crain, C. M., B. R. Silliman, S. L. Bertness, and M. D. Bertness. 2004. Physical and biotic drivers of plant distribution across salinity gradients. *Ecology* 85: 2539–2549.

Cresswell, I. D., and P. B. Bridgewater. 1998. Major plant communities of coastal saltmarsh vegetation in Western Australia. Pages 297–325 *in* A. J. McComb and J. A. Davis (eds.), *Wetlands for the Future*. Adelaide: Gleneagles.

Dale, P. E. R. 1994. An Australian perspective on coastal wetland management and vector control. Pages 771–780 *in* W. J. Mitsch (ed.), *Global Wetlands: Old World and New*. New York: Elsevier.

Davies, W. C. 1931. Tidal flat and salt marsh studies in Nelson Haven. *New Zealand Journal of Science and Technology* 12: 338–360.

Deeley, D. M., and E. I. Paling. 1998. Assessing the ecological health of estuaries in southwest Australia. Pages 257–271 *in* A. J. McComb and J. A. Davis (eds.), *Wetlands for the Future*. Adelaide: Gleneagles.

DeLaune, R. D., S. R. Pezeshki, J. H. Pardue, J. H. Whitcomb, and J. W. H. Patrick. 1990. Some influences of sediment addition to a deteriorating salt marsh in the Mississippi River deltaic plain: A pilot study. *Journal of Coastal Research* 6: 181–188.

Diamond, J. 1999. *Guns, Germs and Steel: The Fate of Human Societies*. New York: Norton.

Dickinson, K. J. M., and A. F. Mark. 1999. Interpreting ecological patterns in an intact estuary, South-west New Zealand World Heritage Area. *Journal of Biogeography* 26: 913–932.

Doak, B. W. 1931. Tidal-flat and salt-marsh studies in Nelson Haven. II. *New Zealand Journal of Science and Technology* 13: 1–11.

Donnelly, J. P., and M. D. Bertness. 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated rise. *Proceedings of the National Academy of Sciences of the USA* 98: 14218–14223.

Dormann, C. F., and J. P. Bakker. 2000. The impact of herbivory and competition on flowering and survival during saltmarsh succession. *Plant Biology* 2: 68–76.

Edyvane, K. 1996. Issues in the South Australian marine environment. Pages 61–88 *in* L. P. Zann and D. Sutton (eds.), *The State of the Marine Environment Report for Australia*. Canberra: Department of Education Science and Training, Technical Annex 3, State and Territory Issues.

Ellis, J., P. Nicholls, R. D. H. Craggs, and J. Hewitt. 2004. Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. *Marine Ecology Progress Series* 270: 71–82.

Esselink, P., L. F. M. Fresco, and K. S. Dijkema. 2002. Vegetation change in a man-made salt marsh affected by a reduction in both grazing and drainage. *Applied Vegetation Science* 5: 17–32.

Esselink, P., G. J. F. Helder, B. A. Aerts, and K. Gerdes. 1997. The impact of grubbing by greylag geese (*Anser anser*) on the vegetation dynamics of a tidal salt marsh. *Aquatic Botany* 55: 261–279.

Evans, L. T. 1953. The ecology of the halophytic vegetation of Lake Ellesmere, New Zealand *Journal of Ecology* 41: 106–122.

Fletcher, R. L. 1996. The occurrence of "green tides": A review. W. Schramm and P. H. Nienhuis (eds.), *Marine Benthic Vegetation: Recent Changes and the Effects of Eutrophication*. Ecological Studies 123. New York: Springer.

Forrest, B., M. Taylor, and C. Hay. 1997. Foreign marine species in New Zealand: Towards a risk assessment management model. Cawthron Report No. 424.

Fotheringham, D. 1994. A vegetation survey of Barker Inlet, Gulf St. Vincent and recommendations. Coastal Management Branch Technical Report 94/1, South Australian Department of Environment and Natural Resources, Adelaide.

French, J. R., and T. Spencer. 1993. Dynamics of sedimentation in a tide-dominated back barrier salt marsh, Norfolk, U.K. *Marine Geology* 110: 315–331.

Garnett, S. T., and G. M. Crowley. 2000. *The Action Plan for Australian Birds*. Canberra: Environment Australia.

Gibblin, A. E., I. Valiela, and J. M. Teal. 1983. The fate of metals introduced into a New England salt marsh. *Water, Air, and Soil Pollution* 20: 81–98.

Gibbs, M., S. Thrush, and J. Ellis. 2001. Terrigenous clay deposition on estuarine sandflats: Using stable isotopes to determine the role of the mud crab, *Helice crassa* Dana, in the recovery process. *Isotopes in Environmental and Health Studies* 37: 113–131.

Gillespie, P. A. 1989. Bioassay of acute toxicity of herbicide mixture dalapon/weedazol 4L on estuarine fauna. *Environmental Toxicology and Chemistry* 8: 809–815.

Graeme, M., and H. Kendal. 2001. Saltwater paspalum (*Paspalum vaginatum*): A weed review. Unpublished report prepared for Environment Waikato.

Gurevitch, J., and L. V. Hedges. 1999. Statistical issues in ecological meta-analyses. *Ecology* 80: 1142–1149.

Haacks, M., and D. Thannheiser. 2003. The saltmarsh vegetation of New Zealand. *Phytocoenologia* 33: 267–288.

Harbord, W. L. 1949. *Spartina townsendii*: A valuable grass on tidal mudflats. *New Zealand Journal of Agriculture* 78: 507–508.

Harvey, E. 2001. Animals in the physical environment. Pages 398–414 *in* A. Sturman and R. Spronken-Smith (eds.), *The Physical Environment: A New Zealand Perspective*. New York: Oxford University Press.

Heenan, P. B., P. J. deLange, D. S. Glenny, I. Breitwieser, P. J. Brownsey, and C. C. Ogle. 1999. Checklist of dicotyledons, gymnosperms and pteridophytes naturalised or casual in New Zealand: Additional records 1997–1998. *New Zealand Journal of Botany* 37: 629–642.

Hewitt, C. L., J. Willing, A. Baucham, A. M. Cassidy, C. M. S. Cox, L. Jones, and D. M. Wotton. 2004. New Zealand's marine biosecurity: Delivering outcomes in a fluid environment. *New Zealand Journal of Marine and Freshwater Research* 38: 429–438.

Hodgkin, E., and B. Hamilton. 1998. Changing estuarine wetlands: A long term perspective for management. Pages 243–255 *in* A. J. McComb and J. A. Davis (eds.), *Wetlands for the Future*. Adelaide: Gleneagles.

Holland, P. 2001. The biosphere and the role of vegetation. Pages 376–397 *in* A. Sturman and R. Spronken-Smith (eds.), *The Physical Environment: A New Zealand Perspective*. New York: Oxford University Press.

Holt, H. E. 1972. *An Energetic Colonist*. Melbourne: Hawthorn.

Howard-Williams, C., I. Hawes, A.-M. Schwarz, and W. F. Vincent. 1997. Ultraviolet radiation in New Zealand and Antarctic lakes. In *UV Radiation and Its Effects: An Update*. Royal Society of New Zealand Miscellaneous Series 49, pp. 79–81.

Howarth, R. 2002. Changes in mangrove/salt marsh distribution in the Georges River Estuary, Southern Sydney, 1930–1970. *Wetlands (Australia)* 20: 80–103.

Hume, T. 2003. Estuaries and tidal inlets. Pages 191–213 *in* J. R. Goff, S. L. Nichol, and H. L. Rouse (eds.), *The New Zealand Coast*. Palmerston North, New Zealand: Dunmore.

Hutching, G. 1998. *The Natural World of New Zealand*. Auckland: Penguin Books.

International Panel on Climate Change. 2001 *Climate Change 2001: The Scientific Basis—Summary for Policymakers and Technical Summary of the Working Group I Report*. Cambridge: Cambridge University Press.

Jackson, J. B. C. 1997. Reefs since Columbus. *Coral Reefs* 16: 23–32.

Jaensch, R. 2005. Conservation management profile: Saltmarsh wetlands. www.epa.qld.gov.au/nature_conservation/habitats/wetlands/wetland_management.

Jamieson, A. 1994. Survey of *Spartina* in the Auckland Region. Coastal Resource Section, Auckland Regional Council, Auckland, New Zealand.

Jaya, M., B. Moradb, and A. Bel. 2003. Biosecurity, a policy dilemma for New Zealand. *Land Use Policy* 20: 121–129.

Jefferies, R. L. 1988. Pattern and process in arctic coastal vegetation in response to foraging by lesser snow geese. Pages 341–369 *in* L. D. Gottlieb and S. K. Jain (eds.), *Plant Evolutionary Biology*. London: Chapman & Hall.

Jefferies, R. L., and A. R. Rockwell. 2002. Foraging geese, vegetation loss and soil degradation in an Arctic salt marsh. *Applied Vegetation Science* 5: 7–16.

Jensen, A. 1985. The effect of cattle and sheep grazing on salt-marsh vegetation at Skallingen, Denmark. *Vegetatio* 60: 37–48.

Johnson, P. N. 1989. *Wetland Plants in New Zealand*. Wellington: Department of Science and Industrial Research.

Jones, M. B., and M. J. Simons. 1983. Latitudinal variation in reproductive characteristics of a mud crab, *Helice crassa* (Grapsidae). *Bulletin of Marine Science* 33: 656–670.

Juniper, S. K. 1986. Deposit feeding strategy of *Amphibola crenata*: Feeding behaviour, selective feeding and digestion. *Mauri Ora* 13: 103–115.

———. 1987a. Deposit feeding ecology of *Amphibola crenata*. 1. Long-term effects of deposit feeding on sediment micro-organisms. *New Zealand Journal of Marine and Freshwater Research* 21: 235–246.

———. 1987b. Deposit feeding ecology of *Amphibola crenata*. 2. Contribution of microbial carbon to *Amphibola*'s carbon requirements. *New Zealand Journal of Marine and Freshwater Research* 21: 247–251.

Kastler, J. A., and P. L. Wiberg. 1996. Sedimentation and boundary changes of Virginia salt marshes. *Estuarine, Coastal and Shelf Science* 42: 683–700.

Kelleway, J. 2005. Ecological impacts of recreational vehicle use on saltmarshes of the Georges River, Sydney. *Wetlands (Australia)* 22: 52–66.

Kiehl, K., I. Eischeid, S. Gettner, and J. Walter. 1996. Impact of different sheep grazing intensities on salt marsh vegetation in Northern Germany. *Journal of Vegetation Science* 7: 99–106.

Kiehl, K., P. Esselink, and J. P. Bakker. 1997. Nutrient limitation and plant-species composition in temperate salt marshes. *Oecologia* 111: 325–330.

King, M. 2003. *The Penguin History of New Zealand*. Auckland: Penguin Books.

King, W. M., J. B. Wilson, and M. T. Sykes. 1990. A vegetation zonation from saltmarsh to riverbank in New Zealand. *Journal of Vegetation Science* 1: 411–418.

Kirkpatrick, J. B., and J. Glasby. 1981. Saltmarshes in Tasmania. University of Tasmania, Department of Geography, Occasional Paper 8.

Kirkpatrick, J. B., and S. Harris. 1999. Coastal health and wetland vegetation. Pages 304–332 *in* J. B. Reid, R. S. Hill, M. J. Brown, and M. J. Hovenden (eds.), *Vegetation of Tasmania*. Australian Biological Resources Study.

Knox, G. A. 1992. The ecology of the Avon-Heathcote estuary. Report for the Christchurch City Council and the Canterbury Regional Council, New Zealand.

Kriwoken, L. K., and P. Hedge. 2000. Exotic species and estuaries: Managing *Spartina anglica* in Tasmania, Australia. *Ocean and Coastal Management* 43: 533–584.

Laegdsgaard, P., V. Monamy, and N. Saintilan. 2004. Investigating the presence of threatened insectivorous bats on coastal NSW saltmarsh habitat. *Wetlands (Australia)* 22: 29–41.

Latchford, J. A. 1998. The impacts of the runnelling on saltmarsh vegetation and substrate. Pages 327–334 *in* A. J. McComb and J. A. Davis (eds.), *Wetlands for the Future*. Adelaide: Gleneagles.

Layman, C. A., D. A. Arrington, and M. Blackwell. 2005. Community-based collaboration restores tidal flow to an island estuary (Bahamas). *Ecological Restoration* 23: 58–59.

Layman, C. A., D. A. Arrington, B. R. Silliman, and R. B. Langerhans. 2004. Degree of fragmentation affects assemblage structure in Andros Island (Bahamas) estuaries. *Caribbean Journal of Science* 40: 232–244.

Lee, W. G., and T. R. Partridge. 1983. Rates of spread of *Spartina anglica* and sediment accretion in the New River estuary, Invercargill, New Zealand. *New Zealand Journal of Botany* 21: 231–236.

Leendertse, P. C., M. C. T. Scholten, and J. T. van der Wal. 1996. Fate and effects of nutrients and heavy metals in experimental salt marsh ecosystems. *Environmental Pollution* 94: 19–29.

Levine, J., S. Brewer, and M. D. Bertness. 1998. Nutrient availability and the zonation of marsh plant communities. *Journal of Ecology* 86: 285–292.

Lowe, A. 1997. Food resources and habitat use by wading birds of remnant and restored wetlands at the Avon-Heathcote estuary, Christchurch. Unpublished MS thesis, University of Canterbury, Christchurch, New Zealand.

Marsden, I. D., and C. Heremaia. 1998. Saltmarsh restoration and management. *Proceedings of the February 1998 Sea Views Conference*: 136–147.

Mason, R. 1969. The vegetation of the coast. Pages 95–105 *in* G. A. Knox and W. A. Reed (eds.), *The Natural History of Canterbury*. Wellington: University of Canterbury Press.

Mazumder, D., N. Saintilan, and R. J. Williams. 2005a. Comparisons of fish catches using fyke nets and buoyant popnets in a vegetated shallow water saltmarsh flat at Towra Point, NSW. *Wetlands (Australia)*: 37–46.

———. 2005b. Temporal variations in fish catch using pop nets in mangrove and saltmarsh flats at Towra Point, NSW, Australia. *Wetlands Ecology and Management* 13: 457–467.

McCombs, K. 2004. Weed plan for *Phragmites* (*Phragmites australis*). 2004 CCCECO 04/60.

McCombs, K., and M. von Tippelskirch. 2004. Weed plan for sea lavender (*Limonium companyonis*). CCCECO 0411. www.ccc.govt.nz/reports/parksbotanist/CCCEO0411.pdf.

McFadgen, B. G. 2003. Pre-European archaeology of the coast. Pages 265–296 *in* J. R. Goff, S. L. Nichol, and H. L. Rouse (eds.), *The New Zealand Coast*. Palmerston North, New Zealand: Dunmore.

McGlone, M. S. 1989. The Polynesian settlement of New Zealand in relation to environmental and biotic changes. *New Zealand journal of Ecology (Suppl.)* 12: 115–219.

McKenzie, R., B. Connor, and G. Bodeker. 1999. Increased summertime UV radiation in New Zealand in response to ozone loss. *Science* 285: 1647–1649.

McLay, C. L. 1976. An inventory of the status and origin of New Zealand estuarine systems. *Proceedings of New Zealand Ecological Society* 23: 8–26.

Michener, W. K., E. R. Blood, K. L. Bildstein, M. M. Brinson, and L. R. Gardner. 1997. Climate change, hurricanes and tropical storms, and rising sea level in coastal wetlands. *Ecological Application* 7: 770–801.

Morrisey, D. J., T. H. DeWitt, D. S. Roper, and R. B. Williamson. 1999. Variation in the depth and morphology of burrows of the mud crab *Helice crassa* among different types of intertidal sediment in New Zealand. *Marine Ecology Progress Series* 182: 231–242.

Morton, J. E., and M. C. Miller. 1973. The New Zealand Sea Shore. Auckland: Collins.

Morton, R. M., B. R. Pollock, and J. P. Beumer. 1987. The occurrence and diet of fishes in a tidal inlet to a saltmarsh in southern Moreton Bay, Queensland. *Australian Journal of Ecology* 12: 217–237.

Nairn, H. J. 1998. Fish fauna of the Avon-Heathcote Estuary, Christchurch. Unpublished MS thesis, University of Canterbury, Christchurch, New Zealand.

National Science Strategy Committee for Climate Change. 2000. *Climate Change: Research Needs and Strategy*. Wellington: Ministry for the Environment.

Natural Resources, Mines, and Water. 2006. Groundsel Bush *Baccharis halimifolia* declared class 2. Facts Pest Series PP13, Queensland Government, Natural Resources, Mines and Water, Brisbane.

Nicholls, P., and J. Ellis. 2002. Fringing habitats in estuaries: The sediment–mangrove connection. *NIWA Water and Atmosphere* 10: 24–25.

Osenberg, C. W., O. Sarnelle, S. Cooper, and R. D. Holt. 1999. Resolving ecological questions through meta-analysis: Goals, metrics, and models. *Ecology* 80: 1105–1117.

Owen-Smith, N. 1989. Megafaunal extinctions: The conservation message from 11,000 years B.P. *Conservation Biology*: 405–412.

Park, S. 2000. Bay of Plenty Maritime Wetlands Database. Environmental Report 2000/21, Environment Bay of Plenty.

———. 2004. Aspect of mangrove distribution and abundance in Tauranga Harbor. Environment Bay of Plenty, Environmental Publication 16.

Parkyn, S., F. Matheson, J. Cooke, and J. Quinn. 2002. Review of the environmental effects of agriculture on freshwaters. NIWA Client Report FGC02206, National Institute of Water and Atmospheric Research, Hamilton, New Zealand.

Partridge, T. R. 1987. *Spartina* in New Zealand. *New Zealand Journal of Botany* 25: 567–575.

Partridge, T.R., and J. B. Wilson. 1987a. Germination in relation to salinity in some plants of salt marshes in Otago, New Zealand. *New Zealand Journal of Botany* 25: 255–261.

———. 1987b. Salt tolerance of salt marsh plants of Otago, New Zealand. *New Zealand Journal of Botany* 25: 559–566.

———. 1988a. The use of field transplants in determining environmental tolerance in salt marshes of Otago, New Zealand. *New Zealand Journal of Botany* 26: 183–192.

———. 1988b. Vegetation pattern in salt marshes of Otago, New Zealand. *New Zealand Journal of Botany* 26: 497–510.

———. 1989. Methods of investigating vegetation/environment relations: A test using the salt marsh vegetation of Otago, New Zealand. *New Zealand Journal of Botany* 27: 35–47.

Paul, S., and R. Young. 2006. Experimental control of exotic spiny rush, *Juncus acutus*, from Badu saltmarsh in Sydney Olympic Park in coastal salt marshes: Recent techniques in protection, creation and rehabilitation. Sydney Olympic Park Wetland Education Program, Sydney.

Paviour-Smith, K. 1956. The biotic community of a salt meadow in New Zealand. *Transactions of the Royal Society New Zealand* 83: 525–554.

Pennings, S. C., M.-B. Grant, and M. D. Bertness. 2005. Plant zonation in low-latitude salt marshes: Disentangling the roles of flooding, salinity and competition. *Journal of Ecology* 93: 159–167.

Phillips, A. W. 1975. The establishment of *Spartina* in the Tamar estuary, Tasmania. *Papers and Proceedings of the Royal Society of Tasmania* 109: 65–765.

Pringle, A. W. 1982. Tidal immersion of the Tamar Estuary *Spartina* marsh. Tasmania, Australia. *Papers and Proceedings of the Royal Society of Tasmania* 116: 143–152.

Raffaelli, D. G., J. A. Raven, and L. J. Poole. 1998. Ecological impact of green macroalgal blooms. *Oceanography and Marine Biology: An Annual Review* 36: 97–125.

Rash, J. A. E., R. C. Williamson, and S. J. Taylor, eds. 1996. *How Green Is Your Mudflat? Proceedings of the Australian Conference on* Spartina *Control, Yarram, Victoria, 10–12 May, 1995*. Victoria: Department of Conservation and Natural Resources.

Reeves, P. N., and P. D. Champion. 2004. Effects of livestock grazing on wetlands: Literature review. NIWA Client Report HAM2004-059, National Institute of Water and Atmospheric Research, Hamilton, New Zealand.

Robinson, K. I. M., and P. Gibbs. 1982. *A Field Guide to the Common Shelled Mollusks of New South Wales Estuaries*. Perth: Coast and Wetlands Society.

Roman, C. T., R. W. Garvine, and J. W. Portnoy. 1995. Hydrologic modeling as a predictive basis for ecological restoration of salt marshes. *Environmental Management* 19: 559–566.

Roper, D. S., G. N. Mills, R. J. Wilcock, and M. A. Weatherhead. 1996. *Spartina* eradication using Gallant herbicide: An assessment of the environmental effects. NIWA Consultancy Report DOC 60205. Prepared for the Department of Conservation, Wellington, New Zealand.

Roper, D. S., S. F. Thrush, and D. G. Smith. 1988. The influence of runoff on intertidal mudflat benthic communities. *Marine Environmental Research* 26: 1–18.

Rouse, H., S. Nichol, and J. Goff. 2003. Introduction to the New Zealand coast. Pages 9–24 *in* J. R. Goff, S. L. Nichol, and H. L. Rouse (eds.), *The New Zealand Coast*. Palmerston North, New Zealand: Dunmore.

Roy, P. S., and E. A. Crawford. 1984. Heavy metals in a contaminated estuary: Dispersion and accumulation trend. *Estuarine, Coastal Shelf Science* 19: 341–358.

Saenger, P., M. M. Specht, R. L. Specht, and V. J. Chapman. 1977. Mangal and coastal saltmarsh communities in Australasia. Pages 293–345 *in* V. J. Chapman (ed.), *Wet Coastal Ecosystems*. New York: Elsevier.

Saintilan, N., and R. J. Williams. 1999. Mangrove transgression into saltmarsh environments in south-east Australia. *Global Ecology and Biogeography* 8: 117–124.

Shaw, W. B., and R. B. Allen. 2003. Ecological impacts of sea couch and saltwater paspalum in Bay of Plenty estuaries. DOC Science Internal Series 113, Department of Conservation, Wellington, New Zealand.

Shaw, W. B., and D. S. Gosling. 1996. *Spartina* control in New Zealand: An overview. Pages 43–60 *in* J. Rash, R. C. Williamson, and S. J. Taylor (eds.), *How Green Is Your Mudflat? Proceedings of the Australasian Conference on* Spartina *Control, Yarram, Victoria, 10–12 May, 1995*. Victoria: Department of Conservation and Natural Resources.

———. 1997. *Spartina* ecology, control and eradication: Recent New Zealand experience. Pages 28–32 *in* K. Pattern (ed.), *Proceedings of the Second International* Spartina *Conference*. Olympia: Washington State University.

Short, A. D. 1988. Areas of Australia's coast prone to sea-level rise. Pages 98–104 *in* P. I. Pearman (ed.), *Greenhouse Planning for Climate Changes*. Clayton, South Victoria: Commonwealth Scientific and Industrial Research Organisation.

Silliman, B. R., and M. D. Bertness. 2004. Shoreline development drives invasion of *Phragmites australis* and the loss of New England salt marsh plant diversity. *Conservation Biology* 18: 1424–1434.

Silliman, B. R., and A. Bortolus. 2003. Underestimation of *Spartina alterniflora* production in Western Atlantic salt marshes. *Oikos* 143: 549–555.

Silliman, B. R., and S. Y. Newell. 2003. Fungal farming in a snail. *Proceedings of the National Academy of Sciences of the USA* 100: 15643–15648.

Silliman, B. R., and J. C. Zieman. 2001. Top-down control of *Spartina alterniflora* production by periwinkle grazing in a Virginia salt marsh. *Ecology* 82: 2830–2845.

Simas, T., J. P. Nunes, and J. G. Ferreira. 2001. Effects of global climate change on coastal salt marshes. *Ecological Modelling* 139: 1–15.

Statistics New Zealand. 2002. *The 2002 Census of Agriculture, Horticulture and Forestry: Provisional Results*. Wellington: Author.

Stuarman, A., I. Owens, and B. Fitzharrris. 2001. Precipitation processes and water storage. Pages 152–174 *in* A. Sturman and R. Spronken-Smith (eds.), *The Physical Environment: A New Zealand Perspective*. New York: Oxford University Press.

Swales, A., I. T. MacDonald, and M. O. Green. 2004. Influence of wave and sediment dynamic on cordgrass (*Spartina anglica*) growth and sediment accumulation on an exposed intertidal flat. *Estuaries* 27: 225–243.

Swales, A., R. Ovenden, T. MacDonald, K. Burt, and J. Hawken. 2002. Effect of Gallant herbicide for *Spartina* control: *Spartina* decomposition and sediment remobilization. NIWA Client Report DOC 00218, National Institute of Water and Atmospheric Research, Hamilton, New Zealand.

Swales, A., R. Ovenden, I. T. MacDonald, A. M. Lohrer, and K. L. Burt. 2005. Sediment remobilisation from decomposing cordgrass (*Spartina anglica*) patches on a wave-exposed intertidal flat. *New Zealand Journal of Marine and Freshwater Research* 39: 1305–1319.

Tasmanian Department of Primary Industries, Water and Environment. 2002. *Strategy for the Management of Rice Grass (*Spartina anglica*) in Tasmania, Australia*. Hobart: Author.

Taylor, R., and I. Smith, principal authors. 1997. *The State of New Zealand's Environment*. Wellington: Ministry for the Environment.

Thannheiser, D., and P. Holland. 1994. The plant communities of New Zealand salt meadows. *Global Ecology and Biogeography Letters* 4: 107–115.

Thomsen, D., I. D. Marsden, and A. D. Sparrow. 2005. A field experiment to assess the transplant success of salt marsh plants into tidal wetlands. *Wetlands Ecology and Management* 13: 489–497.

Turner, L., D. Tracey, J. Tilden, and W. C. Dennison. 2004. *Where River Meets Sea: Exploring Australia's Estuaries*. Brisbane: CRC for Coastal Zone Estuary and Waterway Management.

Turner, S. J., and J. E. Hewitt. 1997. Effect of Gallant for *Spartina* control. *Department of Conservation, Wellington, Conservation Advisory Science Notes* 158: 16.

Valiela, I., and J. L. Bowen. 2002. Nitrogen sources to watersheds and estuaries: Role of land cover mosaics and losses within watersheds. *Environmental Pollution* 118: 239–248.

Valiela, I., and M. L. Cole. 2002. Comparative evidence that salt marshes and mangroves may protect seagrass meadows from land-derived nitrogen loads. *Ecosystems* 5: 92–102.

Vanderaee, M. P. 1988. Changes in saltmarsh vegetation as an early indicator of sea-level rise. Pages 147–160 *in* G. I. Pearman (ed.), *Greenhouse Planning for Climate Changes*. Clayton, South Victoria: Commonwealth Scientific and Industrial Research Organisation.

van de Staaij, J. W. M., G. M. Lenssen, M. Stroetenga, and J. Rozema. 1993. The combined effects of elevated CO_2 levels and UV-B radiation on growth characteristics of *Elymus athericus* (= *E. pycnanathus*). *Plant Ecology* 104/105: 433–439.

Vestergaard, P. 1997. Possible impact of sea-level rise on some habitat types at the Baltic coast of Denmark. *Journal of Coastal Conservation* 103–112.

Wardle, P. 1991. *Vegetation of New Zealand*. Cambridge: Cambridge University Press.

Webb, C. R., and R. C. Russell. 2006. Assessing and managing the nuisance biting and public health risks of mosquitoes associated with constructed and rehabilitated estuarine wetlands. *In Coastal Saltmarshes: Recent Techniques in Protection, Creation and Rehabilitation*. Sydney: Sydney Olympic Park Wetland Education Training Program.

Webster, K. 1997. Macrofauna and marsh plant associations in the Avon-Heathcote estuary, Christchurch. Unpublished master's thesis, University of Canterbury, Christchurch, New Zealand.

West, R. J., C. A. Thorogood, T. R. Walford, and R. J. Williams. 1985. An estuarine inventory for New South Wales, Australia. *Fisheries Bulletin 2, Department of Agriculture New South Wales*: 1–140.

Williams, R. 2006. Evaluation of methods to map saltmarsh in NSW: achieving detection at the local scale. *In Coastal Saltmarshes: Recent Techniques in Protection, Creation and Rehabilitation*. Sydney: Sydney Olympic Park Wetland Education Training Program.

Williams, R. J., and E. A. Watford. 1996. An inventory of impediments to tidal flow in NSW estuarine fisheries habitats. *Wetlands (Australia)* 15: 44–54.

Williams, R. J., F. A. Watford, M. A. Taylor, and M. L. Button. 1998. New South Wales coastal aquatic estate. *Wetlands (Australia)* 18: 25–48.

Williamson, B., G. Lewis, G. N. Mills, and B. Vant. 2003. Contaminants on the coast. Pages 237–264 *in* J. R. Goff, S. L. Nichol, and H. L. Rouse (eds.), *The New Zealand Coast*. Palmerston North, New Zealand: Dunmore.

Williamson, R. B., R. J. Wilcock, B. E. Wise, and S. E. Pickmere. 1999. Effect of burrowing by the crab *Helice crassa* on chemistry of intertidal muddy sediments. *Environmental Toxicology and Chemistry* 18: 2078–2086.

Wilson, J. B., W. M. King, M. T. Sykes, and T. R. Partridge. 1996. Vegetation zonation as related to the salt tolerance of species of brackish riverbanks. *Canadian Journal of Botany* 74: 1079–1085.

Wraight, M. J. 1964. Modification of grasslands by grazing animals. *Proceedings of the New Zealand Ecological Society* 11: 27–33.

Young, B. M., and L. E. Harveya. 1996. A spatial analysis of the relationship between mangrove (*Avicennia marina* var. *australasica*) physiognomy and sediment accretion in the Hauraki Plains, New Zealand. *Estuarine, Coastal and Shelf Science* 42: 231–246.

Zedler, J. B., E. Paling, and A. J. McComb. 1990. Differential salinity response to help explain the replacement of native *Juncus kraussii* by *Typha orientalis* in Western Australian salt marshes. *Australian Journal of Ecology* 15: 57–72.