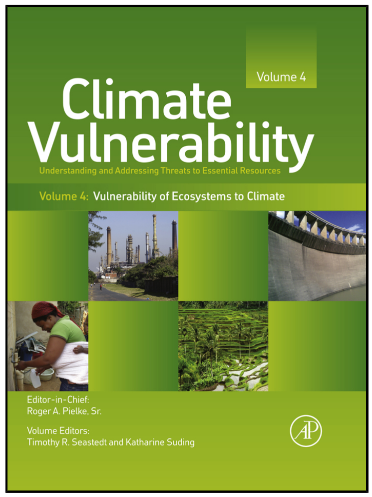

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4.05 Invasive Plants and Animal Species: Threats to Ecosystem Services

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4.05.1 Introduction

The ongoing global redistribution of species is one type of environmental change that we are experiencing. Climate, eutrophication, ocean acidification, and land use change are other types of change that are occurring. None of these changes are occurring in isolation. They all are interacting to produce a myriad of changes, including local, regional, and global changes in flora and fauna (Davis 2009; Perrings et al. 2010; Pereira et al. 2010). Ultimately, this means that the effects of different drivers of environmental change cannot be studied in isolation. They must be studied using an integrated approach (Walther et al. 2009). This approach can challenge or redefine traditional paradigms that may have developed when the drivers were studied separately.

For example, climate effects may blur the boundaries between native and non-native species (Thomas and Ohlemüller 2010). Some species could enter a new region with human assistance while others could enter via natural migration due to altered weather patterns. Both will be new species to the area and both could cause problems. Moreover, a native species that may not have caused problems in a region could become invasive. A good current example of this is the explosive spread by the native mountain pine beetle and the enormous amount of tree mortality it is currently causing in areas of the Rocky Mountains, where it was not found before (Robins 2008). Conversely, a prior native invasive species may become less of a problem. Esper et al. (2007) documented 123 outbreaks of the larch budmoth in Southern Switzerland during the period 832–2004 C.E., with a mean period between outbreaks of 9.3 years, with never a gap between outbreaks of more than two decades. Between 1981 and 2007, not a single outbreak was recorded and the authors hypothesized that increasing air temperatures in this region during this period may be at least partly responsible for the absence of outbreaks.

Non-native species, like native species can be beneficial, harmful, or negligible in their effects. When a non-native species causes harm of some sort, it is typically described as invasive. Thus, the term 'invasive species' normally refers to non-native species that are producing consequences that humans do not like and deem harmful. The harm caused by non-native species is usually described as one of three types: threats to human health, economic harm, and ecological harm. Of the three, most people would likely agree that organisms that threaten human health, primarily introduced pathogens that are the causes of emerging diseases throughout the world, should be our first priority, followed probably by species that cause or threaten to cause great economic harm. One type of economic harm is the damaging or compromising of ecosystem services. Since the replacement of ecosystem services exacts an economic cost for a society, harm to ecosystem services is considered a type of economic harm in this chapter. Ecological harm would consist of ecological effects that do not have an economic cost, or threaten human health, but may still be deemed undesirable. Also, some species may cause more than one type of harm, and/or the type of harm might change over time. For example a species causing just ecological harm at the outset may begin to produce economic harm as time proceeds.

Among invasive species, introduced pathogens almost certainly pose the greatest threats to human well being and that of other organisms, having shown over and over their ability to devastate crops (e.g., 19th century potato blight in Ireland), livestock (e.g., rinderpest outbreaks in Africa in the 19th and 20th century), humans (e.g., 1918 influenza pandemic), and other plant and animal species (e.g., ongoing chytrid fungus epidemic killing frogs worldwide). However, introduced plants and animals also can cause great economic harm, often by damaging ecosystem services. In this chapter, I review some of the harm done to ecosystem services by non-native invasive plants and animals, as well as threats of future harm, and how climate is mediating these effects, sometimes exacerbating them and sometimes mitigating them.

There are dozens of types of ecosystem services, but they are commonly grouped into four categories: provisioning, regulatory, supporting, and cultural (MA 2005). Examples of provisioning services provided by nature are food, timber, fiber, and

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water. Regulating services include pollination, flood control, water purification, and processes reducing threats of disease and harm from climate. Tourism, recreation, aesthetics, and spirituality constitute cultural services, while supporting services include processes such as nutrient cycling and soil formation, which support the other services (MA 2005). The impacts of invasive plants and animals on these ecosystem services, as well as examples of how climate sometimes interacts with species introductions to produce these effects, are described separately in the following sections.

4.05.2 Impacts of Invasive Species on Ecosystem Services

4.05.2.1 Provisioning Services

In several cases, introduced tree species have been found to reduce surface water availability, particularly when replacing native herbaceous species, due to increased water uptake and evapotranspiration. In parts of New Zealand uplands, native tussock grasslands help provide water supplies for humans living at lower elevations. Due to their morphology and physiology, the grasses are efficient at capturing water, whether rain, fog, or snow, and they exhibit low rates of transpiration. As a result, the grasses released 64–80% of the precipitation to surface waters that were then accessible to humans (Mark and Dickinson 2008). In 1980, *Pinus radiata* was introduced to provide a timber resource. Twenty five years later, surface runoff, which provides water resources for the local residents, had declined by 50% (Mark and Dickinson 2008). A similar phenomenon occurred in South Africa, where the encroachment of *Pinus*, *Eucalyptus*, and *Acacia* in native upland environments similarly reduced water yield (Van Wilgen et al. 2008). A global meta-analysis of water use by native and non-native invasive plants showed that while there was little difference in sap flow rates, at the ecosystem scale, invasive-dominated ecosystems were more likely to exhibit higher sap rates per unit ground area than native-dominated ecosystems (Cavaleri and Sack 2010). On the other hand, comparisons between evapotranspiration rates between ecosystems dominated by invasive versus native species found no consistent differences (Cavaleri and Sack 2010).

For many people in the world, fish is a primary part of their diet. Evidence has shown that the introduction of some non-native fish or other organisms influence the economic value of the native fish used for food by altering the age-dependent survival probabilities. If the non-native fish prefer the larger native fish, then selection favors smaller fish and earlier maturation in the prey population, a process known as 'stunting' (Lehtonen 2002), which then directly impacts this primary food resource for local people. The dramatic negative impact on the North American Great Lakes fisheries by the sea lamprey, *Petromyzon marinus*, is well known. During the 1940s and 1950s, populations of the native lake trout, white fish, and chub collapsed and with it the associated fishing industry. Prior to the introduction of the sea lamprey into lakes like Michigan and Superior, annual US and Canadian harvests of lake trout exceeded 15 million pounds. By the early 1960s, the total Great Lakes harvest had dropped to 300 000 pounds, with harvests in some of the lakes being virtually nil (Great Lakes Fisheries Commission 2000).

The introduction of the comb jelly (*Mnemiopsis leidyi*) into the Black Sea is estimated to have reduced the anchovy harvest by 10% between the mid-1980s and the early 1990s (Travis 1993). Impacts on local fish populations were among the highest concerns when zebra mussels, *Dreissena polymorpha*, were introduced into North America and began to spread. Clear impacts on fish have been recorded, but the impacts are not always consistent. Commonly, planktivores decline following the introduction of the mussels, which deplete plankton populations. Fish feeding on benthic resources often increase in numbers but, many fish are able to utilize both pelagic and benthic resources and are able to buffer the changes produced by the mussels (Higgins and Van der Zanden 2010). Data have shown that the size of the impact, and in some cases, even the direction of the impact, varies across ecosystems and fish species (Strayer 2009), making it impossible at this point to be able to make any clear predictions of impacts on fish populations in a lake when colonized by zebra mussels.

The South American plant, *Salvinia molesta*, is recognized as one of the most widespread invasive aquatic plant species in the world, (Room 1990). It is able to double in cover every 3–4 days and can quickly infest entire freshwater systems (Room and Thomas 1986). In a short time, *S. molesta* can threaten local water supplies, and by making waterways unnavigable, prevent access to fish, which constitute a primary part of the diet of local residents (Stone 2011). In Papua New Guinea, entire villages had to be abandoned. The residents moved once local waterways were infested with this species (Stone 2011). Similar harm has resulted from the introduction of water hyacinth, *Eichhornia crassipes*. In Lake Victoria, its introduction has been documented to reduce fish harvests, water supplies, and in general, obstruct navigation through the waterways (Kasulo 2000). Other invasive species have undermined local water quality. In Southeast Asia, the Golden Apple Snail, *Pomacea canaliculata*, has converted what were originally naturally water-purifying wetlands into algal-dominated systems (Carlsson et al. 2004). Introduced carp can have a similar effect, transforming mostly clear lakes and streams into turbid systems (Angeler et al. 2002).

The negative impacts on timber resources of introduced insects have been well documented. In some instances, evidence suggests that warming is exacerbating the negative effects of these forest insect pests by permitting additional generations (Gomi et al. 2007). The hemlock wooly adelgid, *Adelges tsugae*, has been a primary source of mortality for the eastern hemlock, *Tsuga canadensis*, in many eastern North American populations (McClure and Cheah 1999; Lovett et al. 2006). Other prominent invasive insects in North America which undermine timber resources and/or services such as shade and cooling in urban areas, include the Asian long-horned beetle (Haack et al. 1996; Becker 2000), the balsam woolly adelgid (Hain 1988), the gypsy moth (Davidson et al. 2001), and the emerald ash borer (McCullough and Katovich 2004). The Asian long-horned beetle is also a timber pest in Europe (Vilà et al. 2010). The emerald ash borer arrived in this author's neighborhood in 2009, which represented its first documentation in the state of Minnesota. The state of Minnesota is estimated to have 900 million ash trees, many of them growing in forests, but many also planted abundantly in cities to replace elms which had died due to the Dutch elm disease during the 20th century. Current estimates of the damage expected from the emerald ash borer, including loss of timber and costs to remove trees from yards, parks, and city streets, range from the tens to the hundreds of millions of dollars for Minnesota alone, and the species has spread throughout the entire upper Midwest, meaning regional costs will be much higher.

Introduced earthworms in Great Lakes forests of North America are known to have dramatically altered the physical and biogeochemical properties of the soil and litter layer, to the detriment of many native herbs (Frelich et al. 2006). They are also believed to be responsible for some of the declines in tree seedling success (Hale et al. 2006), which means they may constitute a threat to timber resources as well. Of course major threats to timber resources also come from native species. The southern pine beetle, *Dendroctonus frontalis*, is considered the most destructive insect pest of pines in the southern US. Price et al. (1992) estimated that the species had caused \$900 million of damage between 1960 and 1990. The mountain pine beetle, *Dendroctonus ponderosae*, may be killing more trees in North America at this time than any other species. Perhaps due to a warming climate, the species has spread northward into areas where it had not previously been found, or at least where it had never been a large problem, such as portions of British Columbia. This raises an interesting question: if a native North American species expands its range on its own, should it be considered a non-native species in its new range? For the ecosystems and species in the new range, it makes little difference how the species got there. There is no clear scientific answer to this question. Individuals will disagree as to whether such a species should be considered native or non-native, which illustrates the increasing ambiguity that is developing, as these different global change drivers interact with each other. Distinct categories that may have made sense when a driver is studied separately can begin to break down when studied in the context of another driver, for example, invasive species studied in the context of climate change.

Some non-native species compete for forage in pasturelands where they have been introduced, thereby reducing the productivity of domesticated grazers. Wilson (1995) estimated that foraging by rabbits in Australian pasturelands costs the country \$600 million annually. That introduced insects and weeds cause billions of dollars of damage to crops each year throughout the world is well known and will not be further discussed here. While insects (and pathogens) and weeds are normally the primary threats to crops, in some cases other animals cause significant harm as well, some of which are introduced. For example, the coypu (*Myocastor coypus*) causes extensive crop damage in parts of Europe (Bertolino and Genovesi 2007).

4.05.2.2 Regulating Services

Pollination of many crop and orchard species is a large ecological service provided by insects. While many studies of the effects of introduced insects on plant pollination have not focused on food plants, the results indicate the clear possibility that pollination extent and efficiency could be compromised by introduced species (Traveset and Richardson 2006). This can happen in several ways. Introductions of new flower visitors may reduce the visitation rates by native species, which may be more effective pollinators (Gross and Mackay 1998; Hansen et al. 2002; Dohzono et al. 2008). Introduced predators may reduce the size of native pollinator populations (Kelly et al. 2007). Also, introduced plants could compete with the native plants for pollinators (Brown and Mitchell 2001). Of course, it must be acknowledged that a primary pollinator of many crop and orchard species is the honey bee, *Apis mellifera*, an introduced species itself. The influence of non-native plants on native pollinator populations is an interesting question and some of the answers have been surprising. In some instances, non-native plants have been found to support native pollinators by providing nectar and pollen that may have been declining in the environment, due to a decline in some of the native plant species, or due to various factors, for example, land use change and climate change (Hughes et al. 2006; Kenta et al. 2007). On the other hand, non-native plants may compete with native plants for pollinators resulting in the decline of reproductive success of some of the native species (Morales and Traveset 2009).

Impacts on the global carbon cycle, particularly processes that influence the effects of natural carbon sinks, represent another type of regulating service. Studies on North American prairies have shown that the introduction of new plant species can reduce the ability of the system to sequester carbon. In particular, studies showed that the grasslands stored less carbon when crested wheatgrass, *Agropyron cristatum*, was introduced and replaced many of the native grass species (Christian and Wilson 1999; Curtin et al. 2000). A similar finding was documented in Hawaii where the replacement of native forests with non-native grasslands resulted in a 93% reduction in above-ground biomass, and consequently a dramatic reduction in the above-ground carbon storage (Litton et al. 2006). On the other hand, Wardle et al. (2007) found that the introduction of invasive predators on islands promoted ecosystem carbon sequestration.

One of the most important threats to ecosystem services involves the harm to grazing animals by some introduced plants. Leafy spurge (Leistritz et al. 1993), yellow starthistle (Eagle et al. 2007), and knapweed (Ridenour and Callaway 2001) are estimated to cost landowners and ranchers well more than \$100 million per year (Duncan et al. 2004; and Eagle et al. 2007). Some introduced vertebrates can also diminish the value of pastureland.

Invasive seaweeds are known to produce population, community, and ecosystem effects, although long-term studies are still mostly lacking (Williams and Smith 2007). For example, while the invasive seaweeds may increase primary productivity, it is not yet known whether this may lead to increased abundances of species that might be utilized as food by local human residents (Williams 2007). In some cases, the opposite is believed more likely. Eutrophication is a common consequence of invasive seaweeds, which can trigger algal blooms, which in turn can compromise the productivity of other organisms that humans may use for food (Williams 2007). Introductions of other marine organisms have also been documented to undermine ecosystems services, including decline in food production and decline in property values (Williams and Grosholz 2008). Introductions into freshwater systems can have similar diverse consequences, from affecting particular species to affecting ecosystem processes (Ricciardi and MacIsaac 2011). These impacts include effects on nutrient cycling and primary productivity and significant alteration of the habitat (Ricciardi and MacIsaac 2011). In one instance, the introduction of a non-native planktivorous fish resulted in three native planktivores to shift their diets from zooplankton to benthic organisms near shore. This resulted in an increase in mercury concentrations in all three species (Eagles-Smith et al. 2008). As is the case in estuarine and marine systems, introductions of non-native freshwater plants can significantly affect ecosystem services, including altering water quality (Rommens et al. 2003)

There is considerable evidence that emergent diseases have several root causes, including the direct transporting of pathogens from one region to another, habitat loss or human movement into wild habitats (both of which can increase the contact of humans with novel pathogens), and the decline in biodiversity (Pongsiri et al. 2009), which sometimes can be caused by invasive species. Biodiversity decline can contribute to the spread of disease by the loss of predators of common animal hosts for the disease, as well as dramatic declines in the numbers of animal hosts. (Pongsiri et al. 2009). The previous sentence may appear self-contradictory, but if too many animal hosts are infected than the frequency of human contact with the pathogen will increase. On the other hand, if the number of typical animal host declines too much, then the pathogens need to adopt alternative hosts, for example, humans. Some introduced animals can increase the threat of human disease by creating more breeding sites for the hosts of the pathogens. For example, through their rooting foraging behavior, feral pigs in Hawaii create depressions that can fill with water, thereby providing breeding sites for disease-carrying mosquitoes (Atkinson 1995). Similarly, the tsetse fly (*Glossina* spp.) has been found to utilize stands of an invasive tree, *Lantana camara*, as a new habitat (Leak 1999).

Some introduced species threaten flood control or erosion efforts. The great shipworm (*Teredo navalis*) has damaged dikes and other flood protection installments around the Baltic and North Seas (Leppäkoski et al. 2002). In Europe, the coypu increases erosion along rivers through its herbivory and burrowing into the banks (Bertolino and Genovesi 2007). Other non-native species have increased erosion, including introduced vertebrates on some islands (North et al. 1994) and the replacement of native plant species by non-native plants, the latter which proved to be less effective in erosion control (Lacey et al. 1989). Many invasive plant species have altered fire regimes, some of which can come with very high economic costs. It is estimated that the increase in fire frequency in the Everglades and Australia due to the introduction of a non-native tree, *Melaleuca quinqeunervia*, will cost hundreds of millions of dollars (Serbesoff-King 2003).

4.05.2.3 Supporting Services

In the Great Lakes region of the US, non-native earthworms are believed to be affecting such fundamental ecosystem processes such as soil organic matter decomposition, carbon cycling, and nutrient cycling (Hendrix et al. 2008). Interestingly, a model of the earthworm effects on primary productivity suggested that the earthworm presence may increase primary productivity through the conservation of nutrients in the ecosystem (Barot et al. 2007). Considerable data exist showing how species traits can substantially affect various ecosystem processes in freshwater systems, including nutrient cycling, levels of dissolve oxygen, and water clarity (Vaughn 2010). This indicates that introductions of new species have the possibility of significantly altering some of these processes. One of the best examples of introduced species altering ecosystem processes involves the introduction of non-native mussels into North American freshwater systems. For example, introductions of zebra mussels, *Dreissena polymorpha*, usually cause dramatic declines in phytoplankton and small zooplankton, which substantially increase water clarity, as well as levels of soluble nitrogen and phosphorus, which in turn have sometimes resulted in significant increases in the growth and abundance of macrophytes (Caraco et al. 1997, 2000).

On islands, major sources of nutrient inputs come from seabirds during the breeding season, when they transport nutrients acquired from ocean food to the islands via their feces. A number of recent studies have shown that the introduction of non-native predators, such as foxes and rats, can dramatically reduce the transfer of nutrients from the ocean to the island, thereby significantly affecting soil fertility, which in turn influences the type of vegetation that dominates (Fukami et al. 2006; Wardle et al. 2007). Wardle et al. (2009) showed that the introduction of the predators and the consequent disruption of the nutrient transfer between ecosystems reduced leaf nitrogen levels and reduced the rate of litter decomposition.

4.05.2.4 Cultural Services

Non-native plants have always been popular among gardeners who enjoy the added diversity and aesthetics that they provide to the environment. Warming climates are already providing opportunities for many popular plant cultivars to be planted further north. In Europe, some garden species are being successfully planted and maintained as far as 1000 km north of their recently historical range (Van der Veken et al. 2008). At the same time, other non-native species may threaten other cultural services. Examples include non-native plants infesting waterways and interfering with boat traffic and tourism (Eiswerth et al. 2005). Impacts by some aquatic invasive species that reduce desired fish populations or impede boat travel or otherwise interfere with tourist activities will significantly reduce the public enjoyment of these environments, not to mention the economic losses experienced by communities for which tourism was a primary industry. Of course what is deemed environmental harm by one person may be deemed a cultural service by another. For example, while the Minnesota European environmentalists and nature lovers were decrying the introduction and spread of garlic mustard (*Alliaria petiolata*) into Minnesota forests, the local Hmong population was out harvesting the species as a food source, thereby retaining their cultural practice of harvesting wild plants for food.

4.05.3 Looking Ahead: Climate and Its Effects on Invasive Plants and Animals

Depending on the species, climate facilitates or inhibits the spreading of some invasive species (Bradley et al. 2010, Diez et al. 2012). Regional changes in precipitation and temperature regimes can cause expansions, contractions, or shifts in species' ranges depending on the direction of the changes in temperature and precipitation and on how the individual species respond to these changes. It is likely that the spread of certain types of species would be more likely to be more sensitive to any alterations in regional and local climate change and that certain types of environments may be particularly vulnerable to species introductions influenced by climate.

4.05.3.1 Species Sensitive to Enhanced Spread

The likelihood that a species will spread into and establish in a new area is a function of how many propagules are introduced into the new area and the likelihood of establishment by individual propagules (Leung et al. 2004), the latter of which is due to how well the traits of the species match up with the ecological conditions in the new environment (Davis 2009). The nature of the altered conditions in the new environment due to climate will open the door to some species more than others. For example, if precipitation in an area increases, this may make new resources available to plants, not only water, but also possibly nutrients, the uptake of which may have been water limited in the past. Plant species adapted to take advantage of high resource conditions will be favored in these environments, assuming the species are able to disperse, or be transported, into them. In both plants and animals, non-native species that either have great natural dispersal abilities or are regularly transported by humans in one way or another would be able to best keep pace with any alterations geographically in weather patterns.

4.05.3.2 Particularly Vulnerable Environments

Some regions of the world are more vulnerable to species introductions if weather patterns change. These include high altitude environments and high latitude regions. High altitude environments provide several important ecosystem services including water supplies and recreational opportunities (Körner 2004). To date, many high altitude sites have yet to experience high rates of species introductions, probably due to a combination of low levels of propagule pressure and habitat disturbance, both a result of low amounts of human activity in these environments. However, more than one thousand species of non-native plants have established in high altitude sites throughout the world and while local species richness may increase at some sites, there is growing concern that some of these plants may negatively affect the highly valued ecosystem services provided by these environments (Pauchard et al. 2009). For example, the replacement of herbaceous or shrub vegetation by trees might result in increased evapotranspiration and less water export down to lower elevations (Mark and Dickinson 2008). Moderating temperatures would inevitably make it easier for many non-native species to spread into higher altitude sites. Also, the spread of non-native plants is likely increasing due to the increased connectivity between high altitude sites, which is resulting from increased movements by humans between sites, often facilitated by the construction of roads and/or recreation areas. The establishment of some non-native plants at these higher elevations may also be facilitated by atmospheric nitrogen deposition (Price 2006), bringing in yet another environmental change driver to interact with invasive species and climate.

Arctic regions have been reported to have experienced dramatic temperature increases over the past few decades, exceeding recent historical trends by 1.4 °C (Kaufman et al. 2009), although other data questions the magnitude of this increase [Alexeev et al 2012]. If surface temperature increases over the Arctic, which then warm the soils, and melt the permafrost, snow and ice, this would transform the terrestrial and marine arctic environments and making them amenable to more species. For example, it is estimated that 219 shallow water mollusk species in the northern Bering Sea have the potential to spread into the North Atlantic due to warming arctic waters without any human assistance (Vermeij and Roopnarine 2008). Increase shrub growth has been documented (Sturm et al. 2001) and the spread of other plants northward, some facilitated by humans and some not, is assumed. Like plants everywhere, they will alter ecosystem processes such as nutrient cycling and carbon sequestration. Some of the changes we may view as enhancing ecosystem services; some we may not. Antarctica currently has only two native vascular plants, Antarctic hairgrass, *Deschampsia antarctica*, and Antarctic pearlwort, *Colobanthus quitensis* (Caryophyllaceae). If surface temperatures warmed and increasing human travel to Antarctica by scientists and tourists, new plant species, as well as animal species, would be arriving soon, if they have not done so already. The number of species in high latitude regions would almost certainly increase since the number of introductions will likely exceed the number of extinctions, perhaps substantially. It is also expected that hybridization will be common between some of the new and long-term residents (Vermeij and Roopnarine 2008).

Environments experiencing increased resource availability are particularly vulnerable to the establishment of new species (Davis et al. 2000). As described above, if climate permits increased water and/or nutrient availability in an environment, it would be more invasible to many plant species. Similarly, if the local and regional climate results in an increase in primary productivity, the increase of food availability would likely make the habitat more invasible for herbivores.

4.05.3.3 Environments Not Likely to Be Vulnerable to the Introduction of Invasive Species

No environment is immune to the introduction of new species. Evidence has shown that all environments are invasible to some extent (Davis 2009). While climate effects may change the invasibility of certain environments with respect to some species, all terrestrial, freshwater, and marine coastal environments would experience species introductions in the future. However, there is one group of environments that probably would be less affected by the interaction of species introductions and any long-term alterations in weather patterns than others. This is not because these environments are less invasible, but rather because they are already occupied mostly by non-native species. These are areas where human disturbance has been high for a long time, such as urban areas and agricultural lands. Undoubtedly, there would be some exchange of species due to climate, with some non-native species being replaced by other non-native species. However, these environments would not experience dramatic shifts in the relative abundance of long-term residents and newly arrived species, as would be the case in many forests, grasslands, wetlands, rivers and lakes, and coastal marine systems, which have currently experienced less intense human disturbance—hence typically are inhabited by proportionately fewer non-native species than are the highly anthropogenically disturbed environments.

4.05.3.4 Challenges in Distinguishing Harm from Change

There is an emerging consensus that invasive species and climate can no longer be studied as independent drivers (Davis 2009; Walther et al. 2009; Thomas and Ohlemüller 2010). Climate influences which species are able to live in particular areas and also what their ecological effects are. Native species that historically may have been beneficial, or at worst a periodic annoyance, could become invasive and very harmful, as is happening with the mountain pine beetle in North America. Some native species that may have been providing important ecosystem services may decline in numbers or even disappear from a region if weather patterns change. In this case, some non-native species may prove to be highly desirable if they are able to replace those ecosystem services and efforts may be undertaken to increase and facilitate their establishment (Walther et al. 2009). As emphasized by Thomas and Ohlemüller (2010), the traditional perspective that native species are 'good' and alien species are 'bad,' which dominated conservation and alien control programs is no longer viable.

The mere characterization of species as native and non-native is believed by many to becoming a dubious exercise, which will only increase in dubiousness as the years go by (Davis 2009; Thomas and Ohlemüller 2010; Davis et al. 2011), particularly since any alteration in weather patterns could shift the ranges of species resulting in new mixtures of species. Pejchar and Mooney (2009) emphasized that many invasive species have turned out to have positive impacts for local residents, and urged caution in making quick and knee-jerk assumptions about the effects of introduced species. Ultimately, determining harm is not a scientific decision, even if scientists are the ones making the decision. This represents a social decision, and one in which different stakeholders may honorably disagree. For example, herbalists may view introduced plants as promising new medicinal value and urban landscape architects may view non-native plants as being able to provide valuable ecosystem services to urban environments (del Tredici 2010). This important fact is recognized by the National Invasive Species Council in the US: 'Some non-native species are considered harmful, and therefore, invasive by some sectors of our society while others consider them beneficial. This discontinuity is reflective of the different value systems operating in our free society, and contributes to the complexity of defining the term *invasive species*' (NISC 2006).

I do not believe we can proceed with definitions of negative impacts from invasive species similar to the following: 'Negative impacts documented included, for example, predation, competition, change in fire regimes, displacement of native species, and modification of succession processes' (MgGeoch et al. 2010). Under such broad definitions, any non-native species could be declared invasive. If almost any ecological impact is enough to consider a species invasive, then almost any non-native species could be considered a threat to ecosystem services, since it takes little imagination to connect some ecological change with some potential threat to an ecosystem service. As a society, we need to be discriminating. We need to be slow to characterize any species as invasive, since that essentially obligates society to spend resources to fix 'the problem.' We need to be sure not to confuse harm with change. We need to focus on those species, both native and non-native, that are truly causing harm or are threatening to cause harm, to ecosystem services. The same holds for climate. Depending on the location and its effects, alterations in long-term weather patterns by itself is not necessarily harmful, and in fact may be regarded as beneficial by people in some places due to its effects, for example, by increasing food production due to increased growing seasons and/or precipitation. Similarly, the changes resulting from the interaction of climate and species introductions should not automatically be considered harmful.

4.05.3.5 Managing Our Priorities

We should stop judging species on their origins and instead focus more on their effects (Davis et al. 2011). Like native species, some non-native produce desirable effects, some are mostly benign, and some, the ones we call invasive species, cause problems, sometimes severe problems. It is of the utmost importance that we distinguish between severe problems and smaller problems, or even question whether we are calling simple change a problem. Any new species will cause some ecological effect. It may affect nutrient cycling, it might compete with other species, it might be prey on other species, but it will always have effects. This means that as long as we study hard enough, we will be able to document ecological effects of every single non-native species. Clearly, society does not have the resources to fund claims amounting to little more than personal preferences. Society needs to retain its resources so that they can be used to manage and mitigate the significant harm to ecosystem services produced by some non-native species.

Given limited social resources, scientists, land managers, and policy makers owe it to the public, who are usually providing the funding, to seriously question whether or not action should be taken to eradicate or manage a non-native species or try to mitigate its effects. Specifically, the following seven questions should be asked.

1. Do the net effects on ecosystem services by this species truly constitute harm, or can the effects be regarded simply as change?
2. Can we learn to live with this species?
3. If we decide we must manage or control it in some way, does an effective management approach actually exist?
4. Does this management approach make sense in the context of the spectrum of climate effects on the environment now and what is possible in the future?
5. If an effective management approach does exist, are we confident that the benefits we will receive from the management intervention will exceed the costs (with costs including not only dollars, but other inevitable effects on the environment and other species, including ourselves)?
6. Are we sure we would not prefer to spend these resources on some other environmental project or initiative?
7. Are we confident we are not throwing money down the drain?

4.05.4 Conclusion

Ultimately, our decisions regarding what to do with invasive species become economic. Of the thousands of non-native species that have entered and will continue to enter our countries in the future, we will only have the resources to manage a small proportion of them. We need to make smart decisions, which means we need to begin using the term invasive more sparingly. Like it or not, these new species are not going back. No matter what we call them, they are our new residents, except in extremely rare cases when an all out eradication campaign is undertaken and is successful, for example, the eradication of a deadly human pathogen from a country. Clearly, when the damage, or threatened damage, by plants and animals to ecosystem services is very high and clearly outweighs the costs and harm associated with whatever management measures will be undertaken, society needs to invest its resources to try to control these.

Although it sometimes seems that non-native species are considered as a kind of toxic or nuclear waste (e.g., with threats of an invasional meltdowns), or some kind of demons that threaten life as we have known it, we are talking about species—plants and animals, our fellow inhabitants of planet earth. As described by Peter del Tredici (2010) in his book 'Wild Urban Plants of the Northeast,'

> The confluence of climate change and urbanization—acting in concert with the global spread of species—has set the stage for spontaneous vegetation to play a major ecological role in the human dominated landscape (Ziska et al. 2004; Christopher 2008; George et al. 2009). Regardless of how humans feel about this brave new ecology, the plants described in this book are well adapted to the world we have created, and as such, are neither good nor bad—they are us.

In the end, we really have little choice. Climate influences and species introductions will continue into the foreseeable future no matter what humans decide to do or not do. We will continue to devote considerable resources to try to protect our ecosystem services from threats from particularly harmful invasive species. However, in many instances, rather than trying to manage the endless introductions, we would do better to try to manage our attitudes toward them. Although we do have some ability to shape the direction and combined impacts of climate and species introductions, ultimately, like other species, we are going to have to adapt and accommodate these changes into our lives, economies, and cultures.

References

Alexeev, V., I. Esau, I. Polyakov, S. Byam, and S. Sorokina, 2012: Vertical structure of recent Arctic warming from observed data and reanalysis products. *Clim. Change*, **111,** 215–239.

Angeler, D. G., M. Álvarez-Cobelas, Sánchez-Carillo, and M. A. Rodrigo, 2002: Assessment of exotic fish impacts on water quality and zooplankton in a degraded semi-arid floodplain wetland. *Aquat. Sci.*, **64,** 76–86.

Archer, S. R., and K. I. Predick, 2008: Climate change and ecosystems of the southwestern United States. *Rangelands*, **30,** 23–28.

Atkinson, C., 1995: Wildlife disease and conservation in Hawaii: pathogenicity of avian malaria (*Plasmodium relictum*) in experimentally infected liwi (*Vestiaria coccinea*). *Parasitology*, **111,** S59–S69.

Barot, S., A. Ugolini, and F. B. Brikci, 2007: Nutrient cycling efficiency explains the long-term effects of ecosystem engineers on primary production. *Funct. Ecol.*, **21,** 1–10.

Becker, H., 2000: Asian longhorned beetle. *Agri. Res.*, **48,** 18–21.

Bertolino, S., and P. Genovesi, 2007: Aquatic alien mammals introduced into Italy: impacts and control strategies. *Biological Invaders in Inland Waters: Profiles, Distribution, and Threats*. F. Fherardi, Ed., Springer, Dordrecht, Netherlands.

Bradley, B. A., D. M. Blumenthal, D. S. Wilcove, and L. H. Ziska, 2010: Predicting plant invasions in an era of global change. *Trends Ecol. Evol.*, **25,** 310–318.

Brown, B. J., and R. J. Mitchell, 2001: Competition for pollination by invasive plants: effects of pollen from *Lythrum salicaria* (Lythraceae) on seed set in native *Lythrum alatum*. *Oecologia*, **129,** 43–49.

Caraco, N. F., J. J. Cole, S. E. G. Findlay, D. T. Fischer, G. G. Lampman, M. L. Pace, and D. L. Strayer, 2000: Dissolved oxygen declines in the Hudson River associated with the invasion of the zebra mussel (*Dreissena polymorpha*). *Environ. Sci. Technol.*, **34,** 1204–1210.

Caraco, N. F., J. J. Cole, P. A. Raymond, D. L. Strayer, M. L. Pace, S. E. G. Findlay, and D. T. Fischer, 1997: Zebra mussel invasion in a large, turbid river: phytoplankton response to increased grazing. *Ecology*, **78,** 588–601.

Carlsson, N. O. L., C. Bronmark, and L. Hansson, 2004: Invading herbivory: the golden apple snail alters ecosystem functioning in Asian wetlands. *Ecology*, **85,** 1575–1580.

Cavaleri, M. A., and L. Sack, 2010: Comparative water use of native and invasive plants at multiple scales: a global meta-analysis. *Ecology*, **91,** 2705–2715.

Christian, J. M., and S. D. Wilson, 1999: Long-term ecosystem impacts of an introduced grass in the northern great plains. *Ecology*, **80,** 2397–2407.

Christopher, T., 2008: Can weeds help solve the climate crisis? *N. Y. Times Mag.*, 29 June 2008.

Curtin, D., F. Selles, H. Wang, R. P. Zentner, and C. A. Campbell, 2000: Restoring organic matter in a cultivated, semiarid soil using crested wheatgrass. *Can. J. Soil Sci.*, **80,** 429–435.

Davidson, C. D., J. E. Johnson, K. W. Gottschalk, and R. L. Amateis, 2001: Prediction of stand susceptibility and gypsy moth defoliation in coastal plain mixed pine-hardwoods. *Can. J. For. Res.*, **31,** 1914–1921.

Davis, M. A., 2009: *Invasion Biology*. Oxford University Press, Oxford.

Davis, M. A., M. K. Chew, R. J. Hobbs, and Coauthors, 2011: Don't judge species on their origins. *Nature*, **474,** 153–154.

Davis, M. A., J. P. Grime, and K. Thompson, 2000: Fluctuating resources in plant communities: a general theory of invasibility. *J. Ecol.*, **88,** 528–536.

del Tredici, P., 2010: *Wild Urban Plants of the Northeast*. Comstock Pub. Associates, Ithaca, NY.

Diez, J. M., and Coauthor, 2012: Will extreme climatic events facilitate biological invasions? *Front. Ecol. Environ.*, **10,** 249–257.

Dohzono, I., Y. K. Kunitake, J. Yokoyama, and K. Goka, 2008: Alien bumble bee affects native plant reproduction through interactions with native bumble bees. *Ecology*, **89,** 3082–3092.

Duncan, C. A., and Coauthors, 2004: Assessing the economic, environmental, and societal losses from invasive plants on rangeland and wildlands. *Weed Technol.*, **18,** 1411–1416.

Eagle, A. J., M. E. Eiswerth, W. S. Johnson, S. E. Schoenig, and G. Cornelis van Kooten, 2007: Costs and losses imposed on California ranchers by yellow star-thistle. *Range Ecol. Manag.*, **60,** 369–377.

Eagles-Smith, C. A., T. H. Suchanek, A. E. Colwell, N. L. Anderson, and P. B. Myle, 2008: Changes in fish diets and food web mercury bioaccumulation induced by an invasive planktivorous fish. *Ecol. Appl.*, **18,** A213–A226.

Eiswerth, M. E., T. D. Darden, W. S. Johnson, J. Agapoff, and T. R. Harris, 2005: Input-output modeling, outdoor recreation, and the economic impacts of weeds. *Weed Sci.*, **53,** 130–137.

Esper, J., U. Buntgen, D. C. Frank, D. Nievergelt, and A. Liebhold, 2007: 1200 years of regular outbreaks in alpine insects. *Proc. R. Soc. B*, **274,** 671–679.

Frelich, L. E., C. M. Hale, S. Scheu, A. R. Holdsworth, L. Heneghan, P. J. Bohlen, and P. B. Reich, 2006: Earthworm invasion into previously earthworm-free temperate and boreal forests. *Biol. Invasions*, **8,** 1235–1245.

Fukami, T., and Coauthors, 2006: Above- and below-ground impacts of introduced predators in seabird-dominateed island ecosystems. *Ecol. Lett.*, **9,** 1299–1307.

George, K., L. H. Ziska, J. A. Bunce, and B. Quebedeaux, 2009: Elevated atmospheric CO_2 concentration and temperature across an urban-rural transect. *Atmos. Environ.*, **41,** 7654–7665.

Gomi, T., M. Nagasaka, T. Fukuda, and H. Hagihara, 2007: Shifting of the life cycle and life-history traits of the fall webworm in relation to climate change. *Entomol. Exp. Appl.*, **125,** 179–184.

Great Lakes Fisheries Commission 2000: Fact Sheet 3. Sea Lamprey: A Great Lakes Invader.

Gross, C. L., and D. Mackay, 1998: Honeybees reduce fitness in the pioneer shrub *Melastoma affine* (Melastomataceae). *Biol. Conserv.*, **86,** 169–178.

Haack, R. A., J. F. Cavey, E. R. Hoebeke, and K. R. Law, 1996: *Anoplophora glabripennis*: a new tree-infesting exotic cerambycid invades New York. *Newslett. Mich. Entomol. Soc.*, **41** (2–3), 1–3.

Hain, F. P., 1988: The balsam woolly adelgid in North America. *Dynamics of Forest Insect Populations: Patterns, Causes, Implications*. A. A. Berryman, Ed., Plenum, New York, 87–109.

Hale, C. M., L. E. Frelich, and P. B. Reich, 2006: Changes in hardwood forest understory plant communities in response to European earthworm invasions. *Ecology*, **87,** 1637–1649.

Hansen, D. M., J. M. Olesen, and C. G. Jones, 2002: Trees, birds and bees in Mauritius: exploitative competition between introduced honey bees and endemic nectarivorous birds? *J. Biogeogr.*, **29,** 721–734.

Hendrix, P. F., M. A. Callaham Jr., J. M. Drake, C.-Y. Huang, S. W. James, B. A. Snyder, and W. Zhang, 2008: Pandora's box contained bait: the global problem of introduced earthworms. *Annu. Rev. Ecol. Syst.*, **39,** 593–613.

Higgins, S. N., and M. J. Van der Zanden, 2010: What a difference a species makes: a meta-analysis of dreissenid mussel impacts on freshwater ecosystems. *Ecol. Monogr.*, **80,** 179–196.

Hughes, R. F., S. R. Archer, G. P. Asner, C. A. Wessman, C. McMurtry, J. Nelson, and R. J. Ansky, 2006: Changes in aboveground primary production and carbon and nitrogen pools accompanying woody plant encroachment in a temperate savanna. *Global Change Biol.*, **12,** 1733–1747.

Kasulo, V., 2000: The impact of invasive species in African lakes. *The Economics of Biological Invasions*. C. Perrings, M. Williamson, and S. Dalmazzone, Eds., Edward Elgar, Cheltenham.

Kaufman, D. S., and Coauthors, 2009: Recent warming reverses long-term arctic cooling. *Science*, **325,** 1236–1239.

Kelly, D., J. J. Ladley, and A. W. Robertson, 2007: Is the pollen limited mistle *Peraxilla tetrapetala* (Loranthaceae) also seed limited? *Austral Ecol.*, **32,** 850–857.

Kenta, T., I. Naoki, N. Teruyoshi, Goka Ko, and T. Hiura, 2007: Comercialized European bumblee can cause pollination disturbance: an experiment on seven native plant species in Japan. *Biol. Conserv.*, **134,** 298–309.

Kondoh, M., 2006: Contact experience, alien-native interactions, and their community consequences: a theoretical consideration on the role of adaptation in biological invasion. *Conceptual Ecology and Invasions Biology: Reciprocal Approaches to Nature*. M. C. Cadotte, S. M. McMahon, and T. Fukami, Eds., Springer, Dordrecht, The Netherlands, 225–242.

Körner, C., 2004: Mountain biodiversity, its causes and function. *Ambio*, **13,** 11–17.

Lacey, J. R., C. B. Marlow, and J. R. Lane, 1989: Influence of spotted knapweed (*Centaurea maculosa*) on surface runoff and sediment yield. *Weed Technol.*, **3,** 627–631.

Leak S. G. A., 1993: *Tsetse Biology and Ecology: Their Role in the Epidemiology and Control of Trypanosmosis*. CAB International.

Leak, S. G. A., 1999: *Tsetse Biology and Ecology: Their Role in the Epidemiology and Control of Trypanosomiasis*. CABI Publishing, Wallingford, UK.

Leistritz, F. L., A. B. Dean, N. M. Wallace, and J. A. Leitch, 1993: Economic impact of leafy spurge on grazingland and wildland in North Dakota. *Great Plains Res.*, **3,** 21–37.

Lehtonen, H., 2002: Alien freshwater fishes of Europe. *Invasive Aquatic Species in Europe*. E. Leppakoski, S. Gollasch, and S. Olenin, Eds., Kluwer Academic Publishers, Dordrecht, 153–161.

Leppäkoski, E., S. Olenin, and S. Gollasch, 2002: The Baltic Sea: a field laboratory for invasion biology. *Invasive Aquatic Species in Europe. Distribution, Impacts and Management*. E. Leppakoski, S. Gollasch, and S. Olenin, Eds., Kluwer Academic Publishers, Dordrecht, The Netherlands, 253–259.

Leung, B., J. M. Drake, and D. M. Lodge, 2004: Predicting invasions: propagule pressure and the gravity of Allee effects. *Ecology*, **85,** 1651–1660.

Litton, A. F., M. Creighton, D. R. Sandquist, and S. Cordell, 2006: The effects of non-native grass invasion on aboveground carbon pools and tree population structure in a dry tropical forest of Hawaii. *For. Ecol. Manag.*, **231,** 105–113.

Lovett, G. M., C. D. Canham, M. A. Arthur, K. C. Weathers, and R. D. Fitzhugh, 2006: Forest ecosystem responses to exotic pests and pathogens in eastern North America. *Bioscience*, **56,** 395–405.

MA (Millennium Ecosystem Assessment), 2005: *Ecosystems and Human Well-being*. Island Press, Washington DC.

McClure, M. S., and C. A. S.-J. Cheah, 1999: Reshaping the ecology of invading populations of hemlock woolly Adelgid, *Adelges tsugae* (Homoptera: Adelgidae) in eastern North America. *Biol. Invasions*, **1,** 247–254.

McCullough, D., and S. Katovich, 2004: *Pest Alert—Emerald Ash Borer*. USDA Forest Service.

MgGeoch, M. A., S. H. M. Butchart, D. Spear, E. Marais, E. J. Kleynhans, A. Symes, J. Chanson, and J. Hoffmann, 2010: Global indicators of biological invasion: species numbers, biodiversity impact and policy responses. *Divers. Distrib.*, **16,** 95–108.

Mark, A. F., and K. J. M. Dickinson, 2008: Maximizing water yield with indigenous non-forest vegetation: a New Zealand perspective. *Front. Ecol. Environ.*, **6,** 25–34.

Morales, L. C., and A. Traveset, 2009: A meta-analysis of impacts of alien vs native plants on pollinator visitation and reproductive success of co-flowering native plants. *Ecol. Lett.*, **12,** 716–728.

NISC (National Invasive Species Council), 2006: Invasive Species Definition Clarification and Guidance White Paper. Approved by the Invasive Species Advisory Committee, 27 April 2006.

North, S. G., D. J. Bullock, and M. E. Dulloo, 1994: Changes in the vegetation and reptile populations on Round Island Mauritius following eradication of rabbits. *Biol. Conserv.*, **67,** 21–28.

Pauchard, A., and Coauthors, 2009: Ain't no mountain high enough: plant invasions reaching new elevations. *Front. Ecol. Environ.*, **7,** 479–486.

Pejchar, L., and H. A. Mooney, 2009: Invasive species, ecosystem services and human well-being. *Trends Ecol. Evol.*, **24,** 497–504.

Pereira, H. M., and Coauthors, 2010: Scenarios for global biodiversity in the 21st Century. *Science*, **330,** 1496–1501.

Perrings, C., H. Mooney, and M. Williamson, 2010: *Bioinvasions & Globalization: Ecology, Economics, Management, and Policy*. Oxford University Press, Oxford.

Pongsiri, M. J., and Coauthors, 2009: Biodiversity loss affects global disease ecology. *Bioscience*, **59,** 945–954.

Price, M. E., 2006: *Global change in mountain regions*. Sapiens Publishing, Duncow, UK.

Price, T. S., C. Doggett, J. L. Pye, and T. P. Holmes, Eds., 1992: *A History of Southern Pine Beetle Outbreaks in the Southeastern United States*. Sponsored by the Southern Forest Insect Work Conference. The Georgia Forestry Commission, Macon, GA, 65.

Ricciardi, A., and H. J. MacIsaac, 2011: Impacts of biological invasions on freshwater systems. *Fifty Years of Invasion Biology*. D. M. Richardson, Ed., Wiley Blackwell, Oxford, 211–224.

Ridenour, W. M., and R. M. Callaway, 2001: The relative importance of allelopathy in interference: the effects of an invasive weed on a native bunchgrass. *Oecologia*, **126,** 444–450.

Robins, J., 2008: Bark beetles kill millions of acres of trees in west. *NYT*, 17 November 2008.

Rommens, W., and Coauthors, 2003: The impact of water hyacinth (*Eichhornia crassipes*) in a eutrophic subtropical impoundment (Lake Chivero, Zimbabwe). I. Water quality. *Arch. Hydrobiol.*, **158,** 373–388.

Room, P. M., 1990: Ecology of a simple plant-herbivore system: biological control of *Salvinia*. *Trends Ecol. Evol.*, **5,** 74–79.

Room, P. M., and P. A. Thomas, 1986: Population growth of the floating weed *Salvinia molesta* Mitchell in the field: effects of weather, insect damage, fertilizers and age. *Aquat. Biol.*, **24,** 213–232.

Serbesoff-King, K., 2003: Melaleuca in Florida: a literature review on the taxonomy, distribution, biology, ecology, economic importance and control measures. *J. Aquat. Plant Manag.*, **41,** 98–112.

Stone, L., 2011: Waltz of the weevil. *Nature*, **470,** 47–48.

Strayer, D. L., 2009: Twenty years of zebra mussels: lessons from the mollusk that made headlines. *Front. Ecol. Environ.*, **7,** 135–141.

Sturm, M., C. Racine, and K. Tape, 2001: Increasing shrub abundance in the Arctic. *Nature*, **411,** 546–547.

Thomas, C. D., and R. Ohlemüller, 2010: Climate change and species' distributions: an alien future? *Bioinvasions & Globalization: Ecology, Economics, Management, and Policy*. C. Perrings, H. Mooney, and M. Williamson, Eds., Oxford University Press. Oxford, 19–29.

Traveset, A., and D. M. Richardson, 2006: Biological invasions as disruptors of plant reproductive mutualisms. *Trends Ecol. Evol.*, **21,** 208–216.

Travis, J., 1993: Invader threatens Black, Azov Seas. *Science*, **262,** 1366–1367.

Van der Veken, S., M. Hermy, M. Vellend, A. Knapen, and K. Verheyen, 2008: Garden plants get a head start on climate change. *Front. Ecol. Environ.*, **6,** 212–216.

Van Wilgen, B. W., B. Reyers, D. C. Le Maitre, D. M. Richardson, and L. Schonegevel, 2008: A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *J. Environ. Manage.*, **89,** 336–349.

Vaughn, C. C., 2010: Biodiversity losses and ecosystem function in freshwaters: emerging conclusions and research directions. *Bioscience*, **60,** 25–35.

Vermeij, G. J., and P. D. Roopnarine, 2008: The coming arctic invasion. *Science*, **321,** 780–781.

Vilà, M., and Coauthors, 2010: How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. *Front. Ecol. Environ.*, **8,** 135–144.

Walther, G.-R., and Coauthors, 2009: Alien species in a warmer world: risks and opportunities. *Trends Ecol. Evol.*, **24,** 686–693.

Wardle, D. A., P. J. Bellingham, K. I. Bonner, and C. P. H. Mulder, 2009: Indirect effects of invasive predators on litter decomposition and nutrient resorption on seabird-dominated islands. *Ecology*, **90,** 452–464.

Wardle, D. A., P. J. Bellingham, C. P. H. Mulder, and K. Fukami, 2007: Promotion of ecosystem carbon sequestration by invasive predators. *Biol. Lett.*, **3,** 479–482.

Williams, S. L., 2007: Introduced species in seagrass ecosystems: status and concerns. *J. Exp. Mar. Biol. Ecol.*, **350,** 89–110.

Williams, S. L., and J. E. Smith, 2007: A global review of the distribution, taxonomy, and impacts of introduced seaweeds. *Annu. Rev. Ecol. Syst.*, **38,** 327–359.

Williams, S. L., and E. D. Grosholz, 2008: The invasive species challenge in estuarine and coastal environments: marrying management and science. *Estuar. Coasts*, **31,** 3–20.

Wilson, G., 1995: *The Economic of Rabbits on Agriculture Production in Australia: A Preliminary Assessment.* ACIL Economics and Policy, Canberra.

Ziska, L. H., D. E. Gebhard, D. A. Frenz, S. Faulkner, B. D. Singer, and J. G. Straka, 2004: Cities of harbingers of climate change: common ragweed, urbanization, and public health. *J. Allerg. Clin. Immunol.*, **111,** 290–295.