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The findings and conclusions in this publication are those of the authors and should not be construed to represent any official USDA or U.S. Government determination or policy.

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Southeast and Caribbean

Albert E. Mayfield III, Humfredo Marcano-Vega, and Ariel E. Lugo

Introduction

Wide climatic variations characterize the Southeastern United States (Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, Texas, and Virginia) and Caribbean (Puerto Rico and the US Virgin Islands) region, including tropical, subtropical, warm-temperate, and temperate environments, as well as diverse ecosystems from coastal wetlands and dunes to piedmont savannahs and montane forests (Fig. A8.1). More than 85% of the forest land in the continental Southeast is privately owned, with the region experiencing rapid population growth (particularly around urban centers), as well as increased landscape and ownership fragmentation (Butler and Wear 2013). This population growth and urbanization, along with changing climate, are likely to put stressors on southeastern ecosystems in ways that may increase their invasion by, or decrease their resilience to, non-native invasive species (Duerr and Mistretta 2013; Miller et al. 2013a). The Atlantic, Gulf, and Caribbean Coasts in the region are home to numerous major commercial ports. The large quantity of shipments arriving from international ports daily serves as a constant potential pathway for new invasive pests and/or their propagules into the region.

Invasive species in the Southeast and Caribbean region include a wide variety of taxa and affect both terrestrial and aquatic systems. Wood-boring insect species, such as ambrosia beetles and their microbial associates and

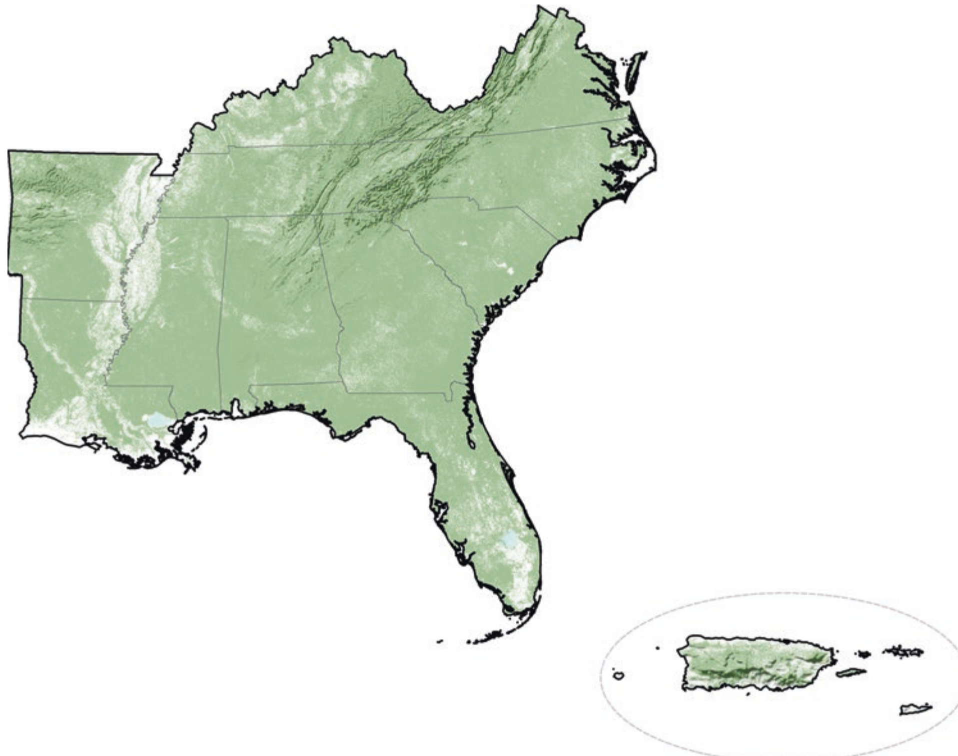

Fig. A8.1 The Southeast and Caribbean Region. (Figure courtesy Daniel Ryerson and Andy Graves, USDA Forest Service Southwestern Region, Forest Health Protection)

symbionts, are easily introduced in solid wood packing material; the warm, humid climate facilitates their establishment in southern forests (Marini et al. 2011). The climate is also very hospitable to the establishment of new invasive plants, which pose a threat not only through their own ecological effects but also via other organisms (e.g., insects and pathogens) they may harbor upon arrival through international plant trade (Miller et al. 2013a). In both the terrestrial and aquatic systems of the Southeast, numerous invasive animal species, including birds, reptiles, amphibians, and fish have, been introduced via the commercial pet trade, for which Florida is a major center of activity (Mazzotti and Harvey 2012; Padilla and Williams 2004; Ruiz-Carus et al. 2006; Russello et al. 2008; Townsend et al. 2003).

Compared to the continental Southeast, the ecosystems of the Caribbean islands of Puerto Rico and the US Virgin Islands, with their diverse sizes and respective levels of anthropogenic activities or land reserve statuses, experience unique environmental and economic effects of non-native invasive species. This chapter will highlight invasive species issues of importance in the region, with separate coverage for the continental Southeast and the Caribbean islands.

Continental Southeast

Invasive Plants

Invasive plants in natural and wildland-urban interface environments of the continental Southeast include over 470 non-native species of trees, shrubs, vines, grasses, and forbs (Miller et al. 2004). Many of these species cause environmental and economic effects that decrease the value of ecosystem services to humans in both terrestrial and aquatic environments. The region has invasive plant collaboration networks organized at local, State, and regional levels, including the Southeast Exotic Pest Plant Council (https://www.se-eppc.org), various cooperative weed management areas, and government agriculture and forestry organizations. These networks function to share information on the identification, control, and management of invasive plant species and to foster collaborative management and research efforts (Miller et al. 2013a). Invasive plants of substantial management importance in the Southeastern United States are numerous; detailed profiles on these plants have recently been covered elsewhere (Miller et al. 2013a, b). Here, key examples representing different plant taxonomic groups are introduced to illustrate the types of impacts and management needs associated with invasive plants in the region.

Cogongrass Cogongrass (*Imperata cylindrica*) is a dense, aggressive, colony-forming grass and is considered one of the top ten worst invasive weeds in the world (MacDonald 2004). In southeastern US forests, it outcompetes native plant species for nutrients, alters decomposition rates, reduces native plant species diversity, and limits light levels near the forest floor to the detriment of endemic flora (Brewer 2008; Brewer and Cralle 2003; Daneshgar and Jose 2009;

Daneshgar et al. 2008; Holly et al. 2009). Cogongrass infestations also increase fine fuel loads, fire intensity, and fire-related tree mortality in pine stands (Lippincott 2000; Platt and Gottschalk 2001) (Fig. A8.2a). Cogongrass forms a dense underground mat of rhizomes with abundant vegetative buds, making propagules difficult to eradicate and easy to spread via heavy equipment. Population genetic studies support historical records of at least two separate introductions of cogongrass into the Southeastern United States from Asia in the early 1900s (Lucardi et al. 2014). In the Southeast region, hundreds of thousands of hectares are infested with cogongrass in Florida, Alabama, and Mississippi (MacDonald 2004), and the species is spreading north and west from these epicenters (Center for Invasive Species and Ecosystem Health 2010). Federal- and State-funded control programs have been aimed at eradicating outlier populations and treating advancing fronts in South Carolina, Georgia, Alabama, Mississippi, Tennessee, Texas (Miller et al. 2004), North Carolina, Florida, and Louisiana. Control relies heavily on repeated applications of herbicides, but a number of cultural management techniques may be integrated to prevent new or treat existing infestations (MacDonald 2004; Miller et al. 2013b). Additional research and technology development on effective control and restoration strategies for cogongrass are ongoing.

Chinese Privet Chinese privet (*Ligustrum sinense*) is a shade-tolerant, evergreen shrub that invades bottomland forests, fencerows, and other habitats. Together with European privet (*Ligustrum vulgare*), it infests more than one million ha throughout the Southeastern United States (Miller et al. 2008). Chinese privet forms dense, mono-species thickets that reduce the abundance and diversity of native herbaceous and woody plants (Greene and Blossey 2012; Hanula et al.

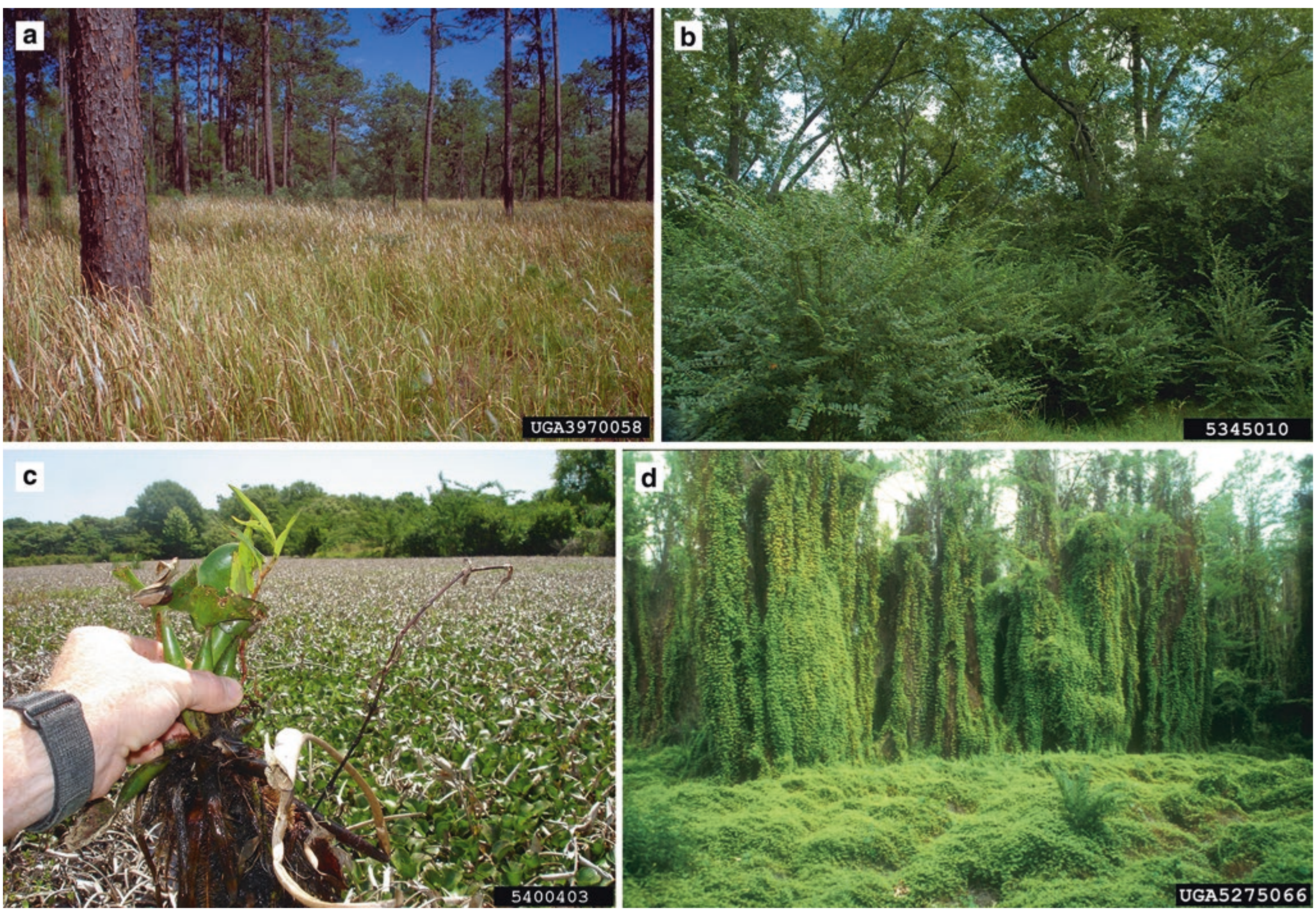

Fig. A8.2 Examples of invasive plant species in the continental Southeastern United States. (**a**) Cogongrass (*Imperata cylindrica)* forms monocultures that outcompete native plant species and increase fire intensity and fire-related tree mortality in southern pine stands. (**b**) Dense, single-species thickets of shade-tolerant Chinese privet (*Ligustrum sinense*) reduce the abundance and diversity of native plants, pollinators, and other taxa in forest understories. (**c**) Common water hyacinth (*Eichhornia crassipes*) has a free-floating, interlocking root system and can rapidly cover the surface of freshwater bodies within a few months of unchecked growth. (**d**) Old World climbing fern (*Lygodium microphyllum*) can cover trees and carry fires into the forest canopy. (Photo credits: (**a**) James R. Meeker, USDA Forest Service, bugwood.org; (**b**) David J. Moorhead, University of Georgia, bugwood.org; (**c**) Graves Lovell, Alabama Department of Conservation and Natural Resources, bugwood.org; (**d**) Tony Pernas, USDI National Park Service, bugwood.org)

2009) (Fig. A8.2b). Experimental removals of Chinese privet in the riparian forests of the southeastern Piedmont have produced positive responses in the abundance, diversity, and/or growth of numerous taxonomic groups, including native herbaceous plants, trees, pollinators, and earthworms (Hudson et al. 2013, 2014; Lobe et al. 2014). Although Chinese privet can be managed using chemical and mechanical treatments, they root sprout readily, and seeds are widely dispersed by birds and other animals, promoting site reinfestations and range expansion. A lace bug, *Leptoypha hospital*, shows promise as a biological control agent, which could provide more efficient long-term control in natural systems (Zhang et al. 2013).

Common Water Hyacinth Common water hyacinth (*Eichhornia crassipes*) is a free-floating, aquatic vascular plant native to South America that has been introduced into freshwater environments in more than 50 countries and five continents (Villamagna and Murphy 2010). It is listed as a noxious weed in four southeastern US States (USDA-NRCS 2017) and is particularly pervasive from eastern Texas through the southern Gulf Region and through the peninsula of Florida (EDDMapS 2016). The plant reproduces both sexually and asexually and forms dense interlocking mats that block light penetration into subsurface water (Fig. A8.2c). Population growth is exponential; a single plant can increase to 500,000 plants within 5 months, covering an acre of water surface and weighing 400 tons (Gettys 2014). Water hyacinth growth inhibits photosynthesis of phytoplankton and submerged vegetation, reduces dissolved oxygen concentrations, alters fish diets through effects on prey species, and severely restricts boating access, navigability, and recreational opportunities (Villamagna and Murphy 2010). Several insect biological control agents have been released to control water hyacinth in the Southeastern United States, and although these have not provided complete control in Florida (Gettys 2014), recent analyses indicate they have been very effective in Louisiana (Nesslage et al. 2016). The species is also managed by intensive chemical or mechanical techniques (Gettys 2014). Water hyacinth tolerates a range of pH, temperature, and nutrient conditions and thrives in certain polluted environments; as such, there is considerable interest in its use for phytoremediation of waters containing excessive nitrogen, heavy metals, and other aquatic pollutants (Alvarado et al. 2008; Fox et al. 2008).

Old World and Japanese Climbing Ferns Introduced as an ornamental plant, Old World climbing fern (*Lygodium microphyllum*) is a twining, vine-like fern that has spread rapidly in natural areas of South and Central Florida since the 1960s. It produces fronds up to 30 m long that spread along the ground, as well as up and over shrubs, mature trees, and other structures (Fig. A8.2d). These fronds form mats of vegetation over 1 m thick that smother, shade, weaken, or kill native vegetation, including native orchids, ferns, and other rare plants found only in unique South Florida ecosystems such as the Everglades. The fronds also act as ladder fuels that carry fire into tree canopies, thereby spreading fire in swamps and other wet areas that would otherwise function as fire boundaries. A partnership known as the Central Florida *Lygodium* Strategy is working to stop the northward spread of Old World climbing fern by monitoring and treating sentinel sites (Langeland et al. 2016; Munger 2005). A related species with greater frost tolerance, Japanese climbing fern (*Lygodium japonicum*) has a wider invasive range, including at least nine southeastern States from Texas to Florida and north to North Carolina. Japanese climbing fern is an economic and regulatory problem for the pine straw industry because the plant is spread via the baling and sale of needles raked from the understory of pine stands (Van Loan 2006). Control of both Old World and Japanese climbing ferns currently relies heavily on herbicides, but a biological control program for the former is being pursued using defoliating and galling insects from the native range (Langeland et al. 2016; Minogue et al. 2009).

Chinese Tallow Chinese tallow (*Triadica sebifera*) is a tree species that produces colorful foliage and white waxy seeds that has been cultivated in China for at least 15 centuries. Introduced to the United States by Benjamin Franklin in 1772, it is valued for numerous practical and economic uses, including seed oil and fat production, medicinal compounds, and biofuel, and as an ornamental. Chinese tallow was promoted commercially in Louisiana and Texas in the early 1900s and has naturalized rapidly throughout the Southeast through repeated introductions, ornamental plantings, and natural spread. Chinese tallow has been increasingly recognized as an invasive species since the 1990s, due to its large seed loads, rapid growth, aggressive response to disturbance, and habitat adaptability. In riparian and floodplain forests, Chinese tallow forms monoculture stands void of other woody vegetation and with less biodiversity. Chemical and mechanical control of Chinese tallow is short-lived and expensive, estimated to cost $200–400 million in Texas, Louisiana, and Mississippi over two decades. There are promising candidate insects for biological control of Chinese tallow, including two from Asia that are under evaluation and one that is naturalized in the invaded range (Duncan et al. 2016; Gao et al. 2016; Wheeler and Ding 2014).

Invasive Forest Insects and Plant Pathogens

There are numerous historic and contemporary examples of invasive insects and pathogens causing substantial tree mortality in the continental Southeast, across a wide range of host plant taxa. Invasive forest insects and pathogens are particularly challenging to manage due to their high rates of reproduction, the ease of spreading them through the transport of wood products and nursery stock, the impracticality (and potential environmental costs) of applying pesticides in forested areas, and their ability to persist in the environment long-term. Management issues and research needs regarding invasive forest insects and disease pathogens are addressed by a diversity of Federal, State and local governments, universities, and private entities throughout the Southeast and are represented by groups such as the Forest Health Committee of the Southern Group of State Foresters and the Southern Forest Insect Work Conference (https://www.sfiwc.org/). Although not comprehensive, below are some of the invasive insect and disease issues of management importance in the continental Southeast region of the United States.

Insects

Hemlock Woolly Adelgid The hemlock woolly adelgid (HWA) (*Adelges tsugae*) is an invasive insect pest threatening the health and sustainability of eastern hemlock (*Tsuga canadensis*) and Carolina hemlock (*Tsuga caroliniana*) in eastern North America. Although hemlock adelgids are native to Western North America, the population of HWA in the Eastern United States was introduced from Japan sometime prior to its initial detection in Virginia in the early 1950s and has spread through more than half the native range of eastern hemlock from Maine to Georgia (Havill et al. 2014). In the Southeastern United States, only a small portion of the hemlock range in Kentucky, Tennessee, and Alabama has not yet become infested (USDA Forest Service 2016). Hemlock mortality and decline caused by HWA has been particularly rapid and pronounced in the Southern Appalachian Mountains (Fig. A8.3c), where winter temperatures are rarely low enough to kill sufficient numbers of adelgids to affect insect population dynamics. Eastern hemlock is a foundation species in riparian areas of the Southern Appalachians, and its elimination by HWA results in numerous short- and long-term ecological effects that include microclimatic changes, altered hydrological and nutrient cycling regimes, and shifts in forest species composition (Vose et al. 2013). Systemic insecticides have been used successfully to temporarily protect trees in parks,

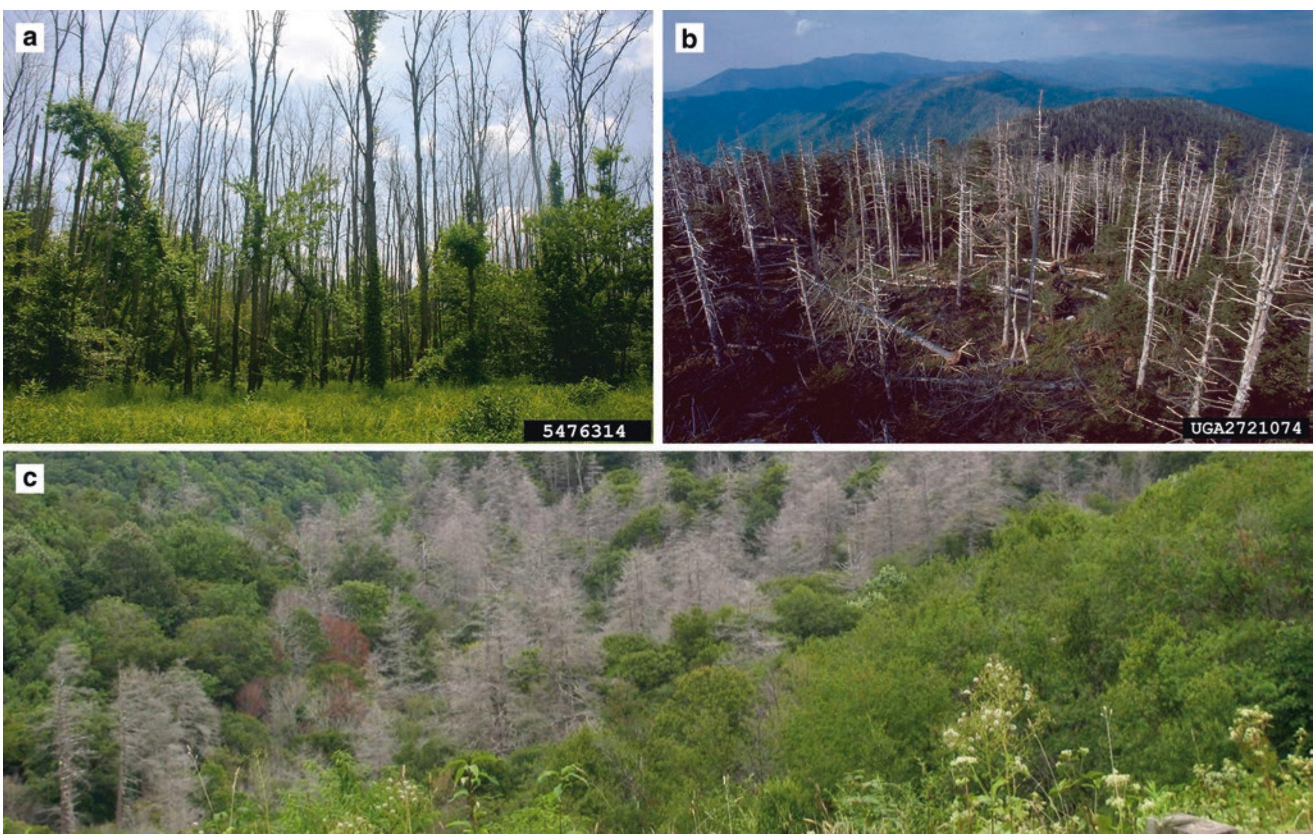

Fig. A8.3 Tree mortality caused by invasive forest insects in the Southern Appalachian Mountains. (**a**) Green ash (*Fraxinus pennsylvanica*) killed by the emerald ash borer (*Agrilus planipennis*) in Virginia. (**b**) Fraser fir (*Abies fraseri*) killed by the balsam woolly adelgid (*Adelges piceae*) in Tennessee. (**c**) Eastern hemlock (*Tsuga canadensis*) killed by the hemlock woolly adelgid (*Adelges tsugae*) in North Carolina. (Photo credits: (**a**) Christopher Asaro, Virginia Department of Forestry, Bugwood.org; (**b**) Ronald F. Billings, Texas A&M Forest Service, Bugwood.org; (**c**) David Casey, USDA Forest Service) recreational areas, and urbanized landscapes but must be applied on a single-tree basis and are not practical for indefinite use in forests. Several species of predators from the native range of HWA in Asia and the Pacific Northwest have been released and become established in the Southeastern United States; their efficacy is currently being evaluated (Onken and Reardon 2011). Research and management efforts in the areas of gene conservation (i.e., seed collection, banking, and orchard establishment, Jetton et al. 2013), host resistance (Oten et al. 2014), and silviculture and restoration (Jetton 2017; Piatek et al. 2016) are also currently being pursued, but additional research effort is likely needed in each of these areas to viably maintain hemlock in the Eastern United States.

Balsam Woolly Adelgid Like the hemlock woolly adelgid, the balsam woolly adelgid (*Adelges piceae*) kills a tree species with relatively limited regional abundance and distribution but with very high local ecological importance. Since its arrival in the high-elevation spruce-fir forests of the Southern Appalachians in the 1950s, balsam woolly adelgid has killed the vast majority of mature Fraser firs in these unique ecosystems (Fig. A8.3b). Drastic losses of the fir canopy and resulting changes in vegetative composition may pose risks for several rare plant and animal species, some of which are at risk of extinction. Although younger cohorts of Fraser fir have survived the initial balsam woolly adelgid epidemics, it is uncertain whether subsequent outbreaks will gradually reduce the genetic diversity of Fraser fir populations and/or its codominant status (with red spruce (*Picea rubens*)) in these forests. In addition to its ecological impact, balsam woolly adelgid is a major pest of the Fraser fir Christmas tree industry, with an annual cost of more than $1.5 million for insecticide treatments in North Carolina alone. Finding and propagating adelgid-resistant Fraser firs and conserving the genetic diversity of the species will likely be important to the conservation and restoration of these southeastern spruce-fir ecosystems (Potter et al. 2005).

Emerald Ash Borer The emerald ash borer (EAB) (*Agrilus planipennis*) is an invasive phloem-feeding beetle native to Asia that threatens to eliminate the ash (*Fraxinus* spp.) from North America. It was initially detected in Michigan in 2002, and by May 2020, it had spread to at least 36 US States and five Canadian provinces (Canadian Food Inspection Agency 2020; EAB Information Network 2020; Herms and McCullough 2014). In the Southeastern region, EAB has been spreading steadily since 2008 and by May 2020 had been detected in all southeastern States except Florida and Mississippi (EAB Information Network 2020). The EAB is the most destructive forest insect in US history and causes an estimated combined loss of $1.7 billion annually to governments and households and in property devaluation (Aukema et al. 2011). In forests, ash mortality due to EAB can (but does not necessarily) exceed 99% within 6 years (Fig. A8.3a) leaving an "orphaned cohort" of seedlings (Klooster et al. 2014; Knight et al. 2013). These changes alter stand light regimes, nutrient cycles, successional dynamics, susceptibility to invasive plants, and coarse woody debris volume (Gandhi and Herms 2010). There have been extensive investments made to improve early detection traps and lures, but even the best combinations are thought to only detect well-established populations. A unique system of bio-surveillance for EAB has been developed by monitoring nest entry by native *Cerceris* wasps, which prey on EAB and other buprestid species (Careless et al. 2014). Effective systemic insecticides are available for temporary protection of high-value landscape trees (Herms and McCullough 2014). Several introduced biological control agents (parasitoids from the native range in Asia) and native natural enemies are reducing EAB population growth rates and enhancing the survival of young ash trees (Duan et al. 2017). As with hemlock, research and management efforts in the areas of host gene conservation and development of host resistance are being pursued but need continued attention (Herms and McCullough 2014; Poland et al. 2015).

Gypsy Moth The European gypsy moth (*Lymantria dispar*) was introduced to Massachusetts in 1869. Outbreaks of this invasive defoliator in New England, Mid-Atlantic, and Northcentral US forests have been the focus of management and research attention for many decades. With the exception of Virginia and extreme northeastern North Carolina (NC-DACS 2016), the European gypsy moth has not yet established in most of the Southeastern United States, though it is expected to continue advancing into the region (Duerr and Mistretta 2013). The gypsy moth can completely defoliate forest canopies, resulting in a number of short-term ecological effects caused by increased light, reduced transpiration, changes in nitrogen and carbon dynamics, and reduced mast production. Oaks (*Quercus* spp.) and aspens (*Populus* spp.) are preferred gypsy moth hosts, but numerous species may be defoliated during outbreaks, including evergreen species such as pine (*Pinus* spp.) and hemlock (Lovett et al. 2006). The severity of tree mortality due to gypsy moth depends on a number of factors, such as tree species composition and susceptibility, the duration and frequency of outbreaks, tree canopy position, and preexisting tree health (Davidson et al. 1999). The spread of the gypsy moth into the Midwest and Southeast is being slowed through a multifaceted management program (including detection surveys, insecticide treatments, and mating disruption) implemented at the infestation's leading edge (Sharov et al. 2002; Tobin 2008). The extent to which an important fungal pathogen, *Entomophaga maimaiga*, helps suppress gypsy moth outbreaks varies with weather conditions, is difficult to predict, and may change with changing climate and movement of the gypsy moth into the Southeast region (Reilly et al. 2014).

Diseases

Laurel Wilt Laurel wilt is a vascular disease caused by a fungus, *Raffaelea lauricola*, that is carried by the redbay ambrosia beetle (*Xyleborus glabratus)* (Fraedrich et al. 2008; Hughes et al. 2015) (Fig. A8.4). This insect/pathogen complex was likely introduced from Asia in solid wood packing material, and the insect was first detected near Savannah, GA, in 2002 (Fraedrich et al. 2008). Since then, laurel wilt has spread rapidly throughout the southeastern Coastal Plain region and is found from North Carolina to Texas, as well as south into the Everglades (Georgia Forestry Commission 2016; Rodgers et al. 2014). The disease is deadly to redbay (*Persea borbonia*), swamp bay (*Persea palustris*), and silk bay (*Persea humilis*) and has killed millions of stems of these species, with the probability of mortality increasing rapidly as stem diameter increases (Mayfield and Brownie 2013; Shearman et al. 2015). The disease is also killing sassafras (*Sassafras albidum*) and avocado (*Persea americana*) and has been documented in populations of rare plant species including pondberry (*Lindera melissifolia*) and pondspice (*Litsea aestivalis*) (Hughes et al. 2015). Laurel wilt represents a serious potential threat to native California bay laurel (*Umbellularia californica*) populations and to avocado production on the West Coast, as well as to the wide diversity of lauraceous plants elsewhere in the Americas (Kendra et al. 2014; Mayfield et al. 2013). There are no effective management strategies in natural systems as yet. Prophylactic fungicide treatments are short-lived and cost-prohibitive (Mayfield et al. 2008), and current detection lures are not effective for manipulating even low beetle populations (Hanula et al.

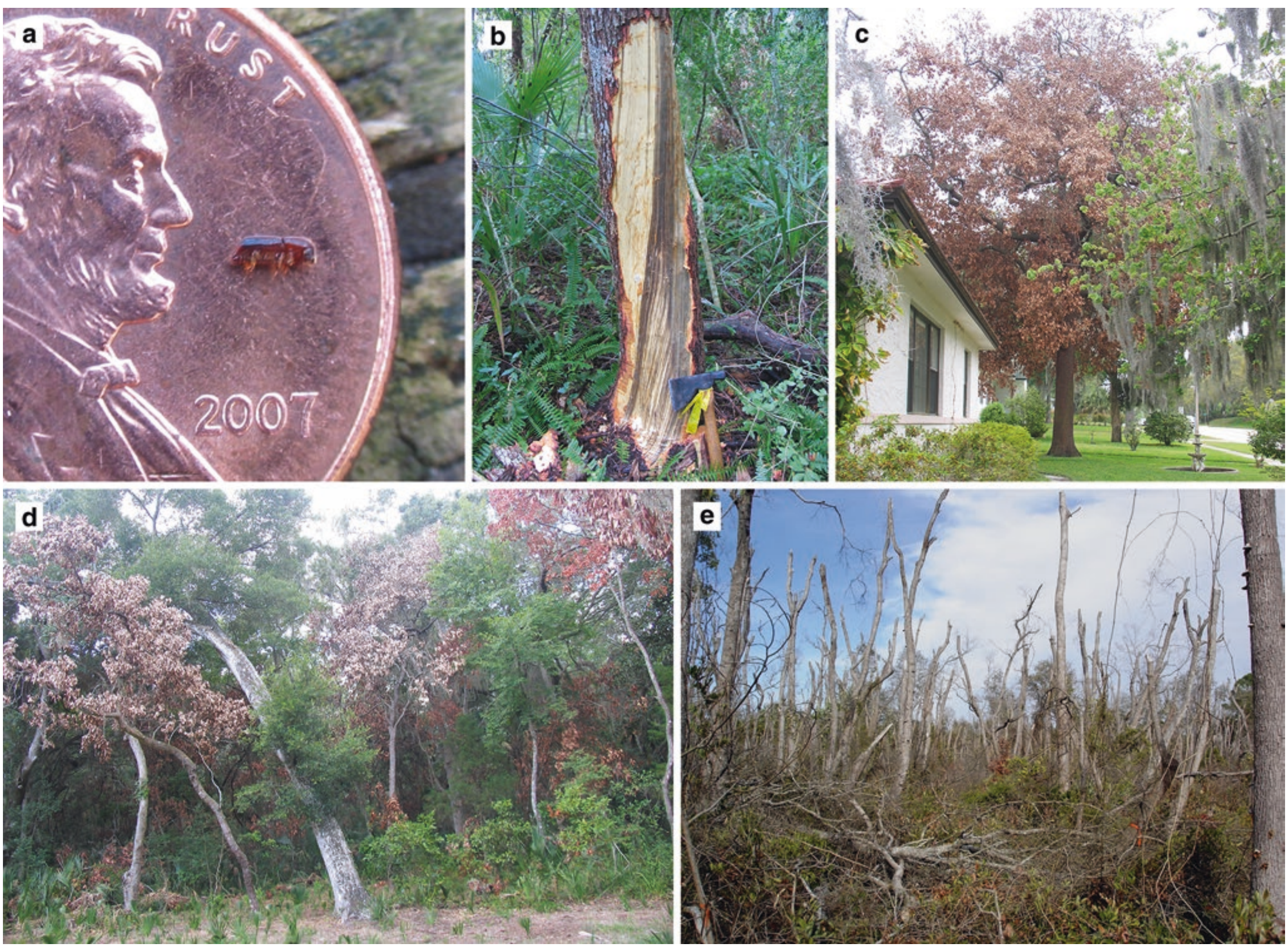

Fig. A8.4 Laurel wilt is a vascular disease of woody plants in the family Lauraceae in the Southeastern United States. It is caused by a fungal pathogen, *Raffaelea lauricola*, that is carried into healthy trees by the (**a**) redbay ambrosia beetle (*Xyleborus glabratus*). (**b**) Infected redbay (*Persea borbonia*) trees wilt rapidly and develop a dark discoloration in the outer sapwood. Millions of redbay trees have been killed in (**c**) neighborhoods, (**d**) mixed maritime forests, (**e**) bayhead swamps, and other southeastern United States ecosystems, and populations of other North American species including sassafras (*Sassafras albidum*) and avocado (*Persea americana*) are being negatively affected. (Photo credits: (**a–d**) Albert Mayfield, USDA Forest Service; (**e**) R. Scott Cameron, Advanced Forest Protection, Inc.)

2016). Efforts to find natural resistance to the disease among lingering redbay trees in post-epidemic forests show promise (Hughes et al. 2015), but additional research on management strategies for numerous other hosts is needed if substantial mortality of lauraceous plants elsewhere in the Americas is to be avoided or minimized.

Thousand Cankers Disease Thousand cankers disease of black walnut (*Juglans nigra*) and other *Juglans* spp. is caused by the joint action of the walnut twig beetle (*Pityophthorus juglandis*) and a fungal pathogen, *Geosmithia morbida*, which it carries. Both the insect vector and the pathogen are thought to be native to the Southwestern United States and Northern Mexico, but they have been discovered far outside their historically known ranges in recent decades, causing dieback and mortality of *Juglans* spp. in the Western United States (Rugman-Jones et al. 2015). Thousand cankers disease was first detected on black walnut in the Southeastern United States in 2010–2012, in portions of Tennessee, North Carolina, and Virginia. Black walnut is one of the most economically valuable tree species in the United States, where standing volume is estimated to be worth ~$500 billion. Numerous States have responded to the spread of thousand cankers disease with quarantines and regulations limiting the movement of walnut material (Daniels et al. 2016). The disease has also been detected in Italy, illustrating the risk of spreading the causal organisms internationally via shipment of unprocessed wood. To date, the impact caused by thousand cankers disease in the Eastern United States has not been well quantified, and the degree to which the disease will be a problem in the native range of black walnut is uncertain.

Butternut Canker Butternut (*Juglans cinerea*) has experienced high rates of mortality (70–90%) throughout its range due to butternut canker, a disease caused by a rapidly spreading, invasive fungal pathogen, *Ophiognomonia clavigignenti-juglandacearum* (Broders et al. 2015). Butternut is not a major component of southeastern US forests (less than 0.5%), but it is a locally valued hardwood for carving, furniture, and various uses of the nuts. Dramatic losses due to butternut canker have resulted in butternut being listed as a Species of Special Concern in Kentucky and as Threatened in Tennessee (Duerr and Mistretta 2013). Research on the development of disease-resistant butternut has shown promise using single-tree selection and breeding, but there is a need for continued butternut conservation and genetic characterization of both the host and the pathogen (Michler et al. 2006; Ostry et al. 2003).

Chestnut Blight The chestnut blight fungus (*Cryphonectria parasitica*) along with the ink disease pathogen (*Phytophthora cinnamomi*) functionally eliminated the American chestnut (*Castanea dentata*) from the Appalachian Mountains in the early 1900s (Anagnostakis 1987). Research aimed at reintroducing blight-resistant American chestnut has employed a backcross breeding program with Chinese chestnut (*Castanea mollissima*) (Hebard 2005) and, more recently, the use of transgenic techniques (Maynard et al. 2008; Newhouse et al. 2014). Field tests of chestnut seedlings produced through the backcross breeding program are being evaluated in the Southern Appalachians (Clark et al. 2016).

Dogwood Anthracnose Another invasive pathogen, *Discula destructiva*, has spread from the Northeastern United States into the Southern Appalachians (Hiers and Evans 1997; Jenkins and White 2002) and has caused precipitous declines in the abundance of flowering dogwood (*Cornus florida*), with potential ecological effects that include disruption of the soil calcium cycle, decreased soil fertility, and decreased species richness (Holzmueller et al. 2010). Research suggests that prescribed fire may help to perpetuate dogwood in forests affected by dogwood anthracnose (Holzmueller et al. 2006, 2009).

Invasive Vertebrates

Burmese Python Burmese pythons (*Python molurus bivittatus*), giant constrictors native to Southeast Asia, were recognized as established in southern Florida by 2000. Over the following decade, they increased dramatically in abundance and distribution, including widespread colonization of Everglades National Park (Dorcas et al. 2012). Both indirect and recent experimental evidence now link this invasive apex predator to precipitous declines in populations of numerous mammal species; Burmese pythons also jeopardize threatened and endangered species such as the Key Largo woodrat (*Neotoma floridana smalli*) and wood stork (*Mycteria americana*) (Dorcas et al. 2012; McCleery et al. 2015). Radio tracking in the Everglades reveals that Burmese pythons have large home ranges that may exceed 2000 ha, use sloughs and coastal habitats as core areas, and frequently use tree islands (Hart et al. 2015). Despite their large size, monitoring and estimating Burmese python population sizes are difficult due to the pythons' cryptic nature and aquatic habitats (Piaggio et al. 2014). New detection techniques using environmental DNA (eDNA) may be useful for monitoring Burmese python occurrence and spread (Piaggio et al. 2014). Government-sanctioned bounty hunts and other attempts to kill pythons have helped raise awareness of the issue, but the thousands that have been documented as killed in Florida over the last 16 years likely represent only a small fraction of the population. The Florida Fish and Wildlife Conservation Commission partners with other agencies to implement management strategies that include licensed hunts, use of citizen science reporting, and prevention of new introductions (Florida FWCC 2016), but it is likely that eradicating Burmese pythons from the region is now impossible.

Feral Swine Feral swine, or wild pigs (*Sus scrofa*) (Fig. A8.5a), have repeatedly been introduced around the globe for centuries and are believed to have been introduced into what is now the Southeastern United States by Spanish explorers in the 1500s (Mayer and Brisbin 2008). Historically, US populations of feral swine were limited to the Southeast, Hawaii, and California, but their range expanded dramatically between 1982 and 2012, from 17 to 38 States (Bevins et al. 2014). This range expansion is thought to have been expedited by several factors, including human transport and release, elimination of predators, and increased human alteration of landscapes for agriculture. While some effects of feral swine in the Southeast are viewed as beneficial (e.g., they serve as part of the diet of the endangered Florida panther (*Puma concolor coryi*) and are hunted for sport), they are widely considered a nuisance species due to rooting/feeding damage to crops, native vegetation, and soil-related habitats (Fig. A8.5b) and are associated with declines in numerous rare, threatened, or endangered plant and animal species populations (Bevins et al. 2014). Successful control or reduction of feral swine populations outside of the Southeastern region (e.g., California, Oregon, and Kansas) has involved sustained, integrated use of multiple management strategies such as hunting, trapping, toxicants, and exclusion devices, but substantial proportions of the populations may need to be eliminated annually to restrict negative effects in areas where they have long been established (Bevins et al. 2014; Timmons et al. 2012).

Nutria Nutria (*Myocastor coypus*) are large semiaquatic rodents that are native to South America but that have been introduced worldwide for fur farming and in some cases to control nuisance aquatic vegetation. They were introduced into Louisiana in the 1930s and now occur in all southeastern States with the possible exception of South Carolina (Carter and Leonard 2002). In wetlands, nutria herbivory destroys aboveground vegetation and root mats, retarding marsh development and contributing to their conversion to open water (Evers et al. 1998; Ford and Grace 1998). Nutria also undermine water control structures via burrowing, cause damage to agricultural crops, and harbor parasites of humans, livestock, and other wildlife (Carter and Leonard 2002). Control programs to reduce nutria damage include eradication attempts as well as managing for low populations, such as in Louisiana where nutria are not only considered ecological pests but are also valued for fur and meat by hunters and trappers (Jojola et al. 2005). In 2002, the Louisiana Department of Wildlife and Fisheries began the Coastwide Nutria Control Program, in which participants are provided an incentive payment for each nutria tail delivered. This program has consistently resulted in more than 300,000 nutria harvested annually and is correlated with a reduction in damaged wetland area from 32,400–40,500 ha annually down to 1600–2500 ha annually (Louisiana Department of Wildlife and Fisheries 2016).

Caribbean Islands

The archipelagos of Puerto Rico and the United States Virgin Islands (USVI) are case studies for the level of species introductions and their social and ecological consequences on local, insular, and regional scales that span the whole insular Caribbean. Table A8.1 shows that the number of introduced species in the region is substantial. Introduction of species has increased the total number of flora and fauna species in the Caribbean. Part of the reason is historical, as the islands have experienced a human presence for thousands of years, including over 500 years of European presence in the region.

Fig. A8.5 (**a**) Feral swine (*Sus scrofa*) have been present in the Southeastern United States for hundreds of years, but their range has expanded dramatically in the last three decades largely due to human land-use changes. (**b**) Their rooting and feeding activity can cause substantial soil disturbance and erosion, influencing plant succession and species composition. They are also hunted for sport and comprise part of the diet of large predators such as the Florida panther. (Photo credits: (**a**) Karan A. Rawlins, National Park Service; (**b**) University of Georgia, bugwood.org)

Table A8.1 Estimates of the number of introduced taxa to the Caribbean islands

Taxon	Native (Caribbean)	Introduced (Caribbean)	Native (PR, USVI)	Introduced (PR, USVI)
Families of seed plants	183	25	163	
Genera of seed plants	1474	560	792	
Species of seed plants	10,401	1799	2108	1032
Herbs				302
Trees				271
Shrubs				167
Vines				129
Grasses				119
Succulents or aquatic				44
Crustaceans		2		
Earthworms		1		
Insect		90		
Jellyfishes		1		
Mites		8		
Mollusks		17		
Solifugids		1		
Tunicates		1		
Amphibians		8		
Birds		20		
Fish		37		
Mammals		20		
Reptiles		15		
Fungi		2		
Diseases		2		

The first three rows for the Caribbean and native seed plants in Puerto Rico and the US Virgin Islands (PR, USVI) are from Acevedo-Rodríguez and Strong (2012). Data on introduced plants and their life forms in PR VI are from Rojas-Sandoval and Acevedo Rodríguez (2015). The rest of the data are from Kairo et al. (2003), who caution that the data are incomplete

Empty cells mean the information is not available in the publications cited or is incomplete. However, information for some of the seed plant taxa is available: http://botany.si.edu/antilles/WestIndies/query.cfm or http://www.sil.si.edu/smithsoniancontributions/Botany/. The numbers of Kairo et al. (2003) add to 225 species (121 invertebrates, 100 vertebrates, and 4 fungi and diseases)

The importance of global commerce to the economy of the islands is another reason for so many species of plants and animals to have been introduced to the region (Ewel et al. 2013). More significant from an ecological viewpoint is the fact that a large number of the introduced species have been naturalized in the Caribbean islands. Naturalized introduced species are now part of the natural milieu of insular landscapes, so much so that many of the insular forests are novel forests and many people cannot differentiate between a native endemic and an invasive introduced species. This mixing of species at ecological and social processes leads to proposals that incorporate introduced species into national or municipal symbols.

The geographic theory known as the forest transition model (Rudel 1998) helps explain why so many introduced species naturalize in the Caribbean and why they become part of the natural forest succession of the region. Fig. A8.6 depicts Forest Inventory and Analysis (FIA) data on the historical changes of forest cover in Puerto Rico. It shows a 400-year deforestation process that accelerated in its last 150 years, followed by an equally rapid natural reforestation over the last 60 years or so. The islands of the Caribbean were deforested to accommodate both agricultural activity and fuelwood extraction to power their economies. With the advent of fossil fuels, fuelwood extractions declined and, with the decline of monoculture crops, so did the pressure on the land as people abandoned the land and moved to cities. The process was quicker in Puerto Rico, which today is ahead of other islands in the temporal progression of the forest transition.

Land abandonment left degraded soils and a deforested landscape behind. This gave introduced species an opportunity to compete, establish, and form what are now recognized as novel forests and ecosystems, i.e., forests and ecosystems with novel combinations of plants and animals. The invasive nature of many introduced tree species was an ecologically sound attribute that allowed the restoration of forest conditions and favorable habitats for the regeneration of native species (flora and fauna) that had been limited to isolated refugia during the deforestation period. Most evidence from Puerto Rico and the USVI shows that introduced and native species can cohabitate and complement each other (Lugo et al. 2012). However, current population models of the native Puerto Rico applecactus (*Harrisia portoricensis* Britton) predict that it may become extinct within areas colonized by the non-native guinea grass (*Megathyrsus maximus* (Jacq.) B.K. Simon & S.W.L. Jacobs), suggesting a distinctive circumstance of displacement in Puerto Rico (Rojas-Sandoval et al. 2016a). Here, we highlight examples of introduced species that are considered invasive and problematic to a variety of human activities. Though we are careful to point out whether perceptions are based on objective information or on the geographic origin of the species, the reality of our examples is usually mixed. A species can be problematic under certain social or ecological conditions and not in others and depending on the context of human judgments regarding health, safety, and wealth. Thus, we suggest that, rather than focusing on the non-native species as the source of problems, attention be given to the functions and interactions among species and humans with regard to their intermixed effects on native populations and ecosystems.

Interspecific Interactions with Non-native Invertebrates

Herbivory The invasive South American *Harrisia* cactus mealybug (*Hypogeococcus pungens* Granara de Willink) is severely affecting columnar cacti in Puerto Rico. Native to

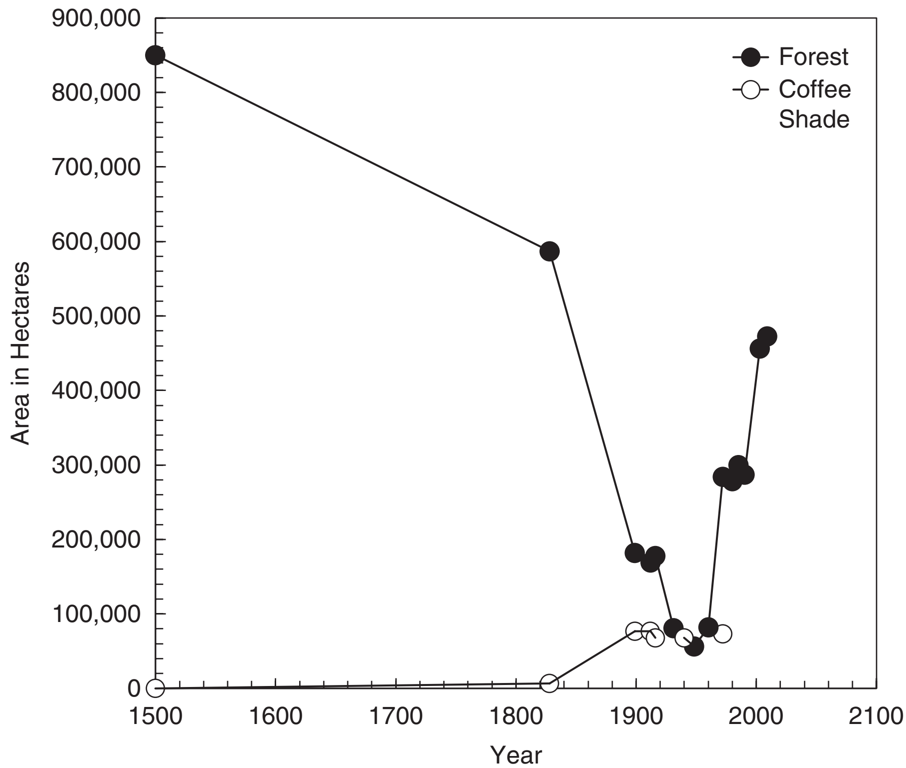

Fig. A8.6 Historical changes of forest cover and coffee shade in Puerto Rico. (Sources of data: Birdsey and Weaver 1982, 1987; Brandeis and Turner 2013a; Brandeis et al. 2007; Franco et al. 1997)

northern Argentina and Chile, westernmost Brazil, Paraguay, and southern Perú, this mealybug was originally intercepted in Puerto Rico on the imported ornamental little hogweed (*Portulaca oleracea* L.) in 2000 and was discovered feeding on the island's cactus during 2005 (Segarra-Carmona et al. 2010). Cacti are important components of dry forests in Puerto Rico and provide food sources for native animals such as endemic bats, birds, moths, and other pollinators (US Department of Agriculture APHIS PPQ 2009). Observed herbivory on cactus hosts includes malformations exposed as gall-like structures, usually associated with arrested growth, prevention of flowering, and the eventual death of some individuals (Zimmerman et al. 2010). The mealybug has substantially extended its distribution and is quickly spreading in the dry forest life zone of the southern coast of Puerto Rico, causing extensive impairment on the tree cactus (*Pilosocereus royenii* (L.) Byles & Rowley). Other members of the Cactaceae family exhibiting severe impairment include the native pitahaya (*Leptocereus quadricostatus* (Bello) Britton & Rose) (Fig. A8.7) and Turks' cap (*Melocactus intortus* (Mill.) Urb.) and the introduced lady of the night cactus (*Cereus hexagonus* (L.) Mill.) found under cultivation. Minor effects have been observed on the native Spanish stenocereus (*Stenocereus fimbriatus* (Lam.) Lourteig) (Segarra-Carmona et al. 2010). The host range of the *Harrisia* cactus mealybug also includes species in families dissimilar to the Cactaceae. Early detection programs and regular inspection of nurseries have been suggested by scholars as a way to limit its spread (Zimmermann et al. 2010). Wind dispersal is considered one of the foremost pathways for the spread of the *Harrisia* cactus mealybug. Thus, its potential expansion from mainland Puerto Rico to other islands of the Puerto Rican Archipelago where endangered cacti are found (i.e., guinea grass) has been highlighted as a concern (Campbell 2015; Zimmerman et al. 2010). The *Harrisia* mealybug is also known to be causing the death of native cacti on the USVI (Campbell 2015).

Chemical control of the *Harrisia* cactus mealybug has been suggested as an appropriate measure specifically for infested plants within nurseries and commercial cultivations. This measure is not feasible for protecting cacti populations in the wild, however, where biological control is recommended instead (Segarra-Carmona et al. 2010). The coccinellid *Decadiomus seini* sp. nov. Segarra & Otero is a new species whose larvae and adults have been discovered feeding on the *Harrisia* cactus mealybug in the dry forests of Puerto Rico (Segarra-Carmona and Otero 2014). In addition, *Gyranusoidea pseudococci* Brèthes, a parasitoid widely distributed in Florida, and the parasitoid wasps of the genus *Leptomastidea* have been influential natural enemies of the *Harrisia* cactus mealybug, thus offering further potential for biological control in Puerto Rico and the Caribbean (US Department of Agriculture APHIS PPQ 2009). Other introduced insects, such as the pink hibiscus mealybug (*Maconellicoccus hirsutus* (Green)), have been successfully controlled in the USVI through the release of nonindigenous wasps in the genus *Leptomastix* (Daley et al. 2012; Meyerdirk

Fig. A8.7 Herbivory of the introduced South American *Harrisia* cactus mealybug (*Hypogeococcus pungens*) on the native pitahaya (*Leptocereus quadricostatus*) induces the proliferation of miniature cacti. Observed herbivory on other native cacti also includes malformations such as gall-like structures; it is usually associated with arrested growth, prevention of flowering, and the eventual death of some individuals. (Photo credit: Humfredo Marcano-Vega, USDA Forest Service Southern Research Station)

et al. 2001). The pink hibiscus mealybug is known to attack diverse crops and ornamental plants, some of which are part of the secondary forests of Puerto Rico and the USVI (Brandeis and Turner 2013a, b; Marcano-Vega et al. 2015; Meyerdirk et al. 2001).

The effects of introduced species such as the *Harrisia* cactus mealybug on landscape elements of the subtropical dry forest (sensu Holdridge; Ewel and Whitmore 1973) are an opportunity to include human ecology principles such as human perceptions (e.g., emotional value) in the study of species invasions within the Caribbean context and abroad (González Bernáldez 1985; Sax et al. 2007). The destructive agave snout weevil (*Scyphophorus acupunctatus* Gyllenhaal), native to the Southwestern United States and Mexico, is present in Puerto Rico and the USVI. Its occurrence has increased the risk of extinction of the USVI endemic and endangered century plant (*Agave eggersiana* Trel.) in St. Croix and of the Puerto Rico Bank endemic corita (*A. missionum* Trel.) in St. John. Chamorro et al. (2016) recently reported the weevil in St. Croix, providing tools for its identification at any stage and recommending a monitoring program throughout the already restricted range of century plant. This initiative is considered to be crucial for prevention because the agave snout weevil's effect is currently described by specialists as virtually impossible to reverse, and landscape management experts have suggested the application of a systemic insecticide around the base of healthy plants (DLC Resources, Inc. 2015). On the other hand, the College of Agricultural Sciences of the University of Puerto Rico at Mayagüez formed the Grupo Antillano de Especies Invasoras (GAEI), which maintains an atlas of invasive insect species. This group shows the *Tabebuia* thrips (*Holopothrips tabebuia* Cabrera & Segarra) as an invasive insect known to be widespread around all mainland Puerto Rico (GAEI 2016; González 2016). Although the genus *Holopothrips* is known to be of neotropical origin, the species' native range is currently uncertain. The *Tabebuia* thrips were found in Puerto Rico in 2006 and are now known to affect the esthetic appearance of leaflets (inward curling forming gall-like structures) and growth of native and introduced tree species in the *Tabebuia* and the *Crescentia* genus in urban and secondary forests (GAEI 2016; González 2016). Currently there is no known mortality of trees as a consequence of the *Tabebuia* thrips, but long-term monitoring is recommended along with the preventive use of systemic insecticides and pruning of affected trees (González 2016). Alternative biocontrol is offered by the insect *Montandoniola moraguesi* Puton (Hemiptera: Anthocoridae), which has been reported as a predator of the *Tabebuia* thrips in Puerto Rico and Florida (GAEI 2016; González 2016).

Interspecific Competition and Mutualism Dispersion of the introduced honeybee (*Apis mellifera* L.) to relatively undisturbed forests at high elevations in northeast Puerto Rico has resulted in novel interactions among species. The observation of swarms of the honeybee indicates that the bees are able to exploit the local forest's floral resources and compete for cavity nesting sites with the endangered Puerto Rican parrot (*Amazona vittata* Boddaert) and other native species (Lugo et al. 2012; Snyder et al. 1987). On the other hand, wild populations of honeybees have also been observed as second in efficiency at depositing pollen on stigmas of flowers of the native terrestrial bromeliad pina cortadora (*Pitcairnia angustifolia* Aiton), resulting in a more complex plant-pollinator relationship than what may be expected from the bromeliad's flower morphology (Fumero-Cabán and Meléndez-Ackerman 2007). Opposing results were found for the native Turk's cap cactus, whose flowers are also visited by honeybees but with negligible effects on the reproductive success of the plant (Fagua and Ackerman 2011). The novel development of partial dependence of native species upon non-natives is developing into a common interspecific interaction in tropical islands (Ewel et al. 2013), but more research is needed to determine the effectiveness and/or consequences of acquired relationships.

Belowground Terrestrial Interactions Additional studies are suggested to better understand the effects of non-native earthworms on tropical belowground ecosystems (González et al. 2006). Studies show how earthworm density decreased while earthworm diversity increased along successional stages of wet forest regeneration after the abandonment of pastures, where *Pontoscolex corethrurus* Müller appeared as the dominant non-native earthworm occurring in all successional stages in Puerto Rico (Zou and González 1997). This earthworm is also found in relatively undisturbed cloud forests, as well as lower altitude tabonuco (*Dacryodes excelsa*) forests in Puerto Rico (Hendrix et al. 1999), but their ecosystem effects remain generally unknown (González et al. 2006). Even so, studies from Puerto Rico show how conversion of forests can increase soil carbon dioxide (CO_2) emissions as litter decomposition and soil respiration can significantly increase under the presence of introduced earthworms such as *P. corethrurus* (Liu and Zou 2002).

Interactions with Non-native Plants

Puerto Rico's secondary forests represent a legacy of anthropogenic activities with new combinations of both native and introduced tree species (Abelleira-Martínez 2010; Fonseca Da Silva 2014; Lugo 2004, 2009). Studies characterizing tree species composition in secondary stands of Puerto Rico highlight the critical function of the introduced African tulip tree (*Spathodea campanulata* P. Beauv.) (Fig. A8.8) as a novel forest facilitator, along with remnant native forests or trees functioning as seed sources for native species recovery within the subtropical moist and wet forest life zones (Abelleira-Martínez 2010; Brandeis and Turner 2013a). The importance of mature native forests as local seed pools is critical because novel forests exhibit fewer endemic species than native forests (Lugo and Helmer 2004). Both novel and native forests, however, share similar structural characteristics (Fonseca Da Silva 2014; Lugo and Helmer 2004). Nevertheless, when comparing novel forest stands with native forests of similar age in analogous Holdridge life zones with similar topographies and elevation, novel forest stands demonstrate lower rates of aboveground biomass, basal area, and soil carbon accumulation (Lugo and Helmer 2004).

In another circumstance, the introduction of Caribbean pine (*Pinus caribaea* Morelet) required the additional introduction of ectomycorrhizal (EM) inoculum to Puerto Rico during the 1950s, for plantations to be successfully established (Briscoe 1959). After reporting the non-native EM fungus *Pisolithus tinctorius* (Pers.) Coker & Couch fruiting under Caribbean pine, populations of both species can be found today outside plantations in disturbed areas (Rivera et al. 2015). It remains uncertain if Caribbean pine will function in the long term as a facilitator or inhibitor of native species during forest regeneration, but some native tree species have been able to reach the canopy layer of small unmanaged plantations after 18 years of growth (Lugo 1992).

Studies show that, after hurricanes, the rapid revitalization of native species in protected old-growth forests within wet areas keeps shade-intolerant introduced species as relatively rare within these areas (Thompson et al. 2007). In contrast, the shade-tolerant introduced rose apple (*Syzygium*

Fig. A8.8 The introduced African tulip tree (*Spathodea campanulata*), with its large orange-red flowers, is widely naturalized in the humid forests of Puerto Rico functioning as a novel forest facilitator along with remnant native forests. (Photo credit: Luis O. Ortiz-López, USDA Forest Service Southern Research Station) *jambos* (L.) Alston) has shown the ability to establish populations and persist in mature native forests over a wide range of soil and topographic conditions (Brown et al. 2006; Thompson et al. 2007). But the invasive rust fungus *Puccinia psidii* Winter, known to infect diverse hosts in the Myrtaceae family, has been affecting rose apple in Puerto Rico and acting as an unintentional biological control across the island (Ross-Davis et al. 2013). All the same, the expected increase in frequency of tropical storms and hurricanes in the Caribbean region due to climate change (UNEP 2008) could open opportunities for the colonization of naturalized tree species in old forest reserves (Thompson et al. 2007). We only know of cases of coexistence of native and non-native plant species, but the monitoring of interspecific relationships needs to endure. Continuous studies comparing the ecological performance of native and non-native *Ardisia* (Myrsinaceae) species to identify traits that promote invasiveness in Puerto Rico serve as examples of the importance of monitoring (Muñoz and Ackerman 2011).

Additionally, the introduced autogamous monk orchid (*Oeceoclades maculata* (Lindl.) Lindl), which has shown the capacity to establish in old remnant forests in Puerto Rico, encourages the establishment of long-term studies to determine its influence on native orchids (Cohen and Ackerman 2009). Studies regarding another introduced orchid that has naturalized in Puerto Rico and the USVI—the Philippine ground orchid (*Spathoglottis plicata* Bl.)—have shown that it shares the native florivorous weevil, *Stethobaris polita* Chevrolat, with the native orchid, flor de pasmo (*Bletia patula* Graham) (Recart et al. 2013). A positive relationship has been found between the density of the introduced orchid and the number of weevils and floral damage on the native orchid (Recart et al. 2013). This indirect negative effect on flor de pasmo is expected to continue as distribution models predict that both species of orchids will strongly overlap, having the effect of raising the abundance of the native florivorous weevil as an orchid specialist (Recart et al. 2013).

Furthermore, the non-native red imported fire ant (*Solenopsis invicta* Buren) is attracted to extrafloral nectaries of *S. plicata* and inhibits the native weevil behavior on this non-native orchid. These acquired plant-animal interactions diminish biotic resistance to the continuous spread of *S. plicata* and are proposed as examples of apparent competition between the native and non-native orchid mediated by the native weevil and of invasional meltdowns through facilitation of the non-native ant's invasion by the introduced orchid (Ackerman et al. 2014; Recart et al. 2013). Partial support for the invasional meltdown hypothesis has been provided by showing that red fire ants have a positive effect on the population growth rate of *S. plicata* (Falcón et al. 2016).

In the USVI, the non-native tan tan tree (*Leucaena leucocephala* (Lam.) de Wit) has been perceived as harmful to the local environment because it appears to displace native vegetation while recognizing at the same time that it does not invade closed canopy forests due to its shade intolerance (Daley et al. 2012). An alternative view of tan tan considers the species to be beneficial for Caribbean forest regeneration after chronic anthropogenic disturbance. Tan tan has the ability to regenerate in areas dominated by African grasses and maintained by fires, where native saplings have shown slow growth and high mortality. Tan tan forms mono-dominant stands in the subtropical dry forest life zone in which these conditions prevail (Wolfe and Van Bloem 2012). When anthropogenic disturbance abates, tan tan stands in Puerto Rico evolve toward novel species combinations as native species begin to establish in these sites (Molina Colón et al. 2011). However, as with other novel forests in the different ecological life zones of Puerto Rico and the USVI, a transition to primary forest composition has not been documented (Abelleira Martínez 2010; Aide et al. 2000; Atkinson and Marín-Spiotta 2015; Brandeis et al. 2009; Fonseca Da Silva 2014; Lugo and Helmer 2004; Marcano-Vega et al. 2002, Molina Colón and Lugo 2006). An alternative situation is represented by the cajeput tree (*Melaleuca quinquenervia* (Cav.) S.F. Blake), introduced as an ornamental and forming persisting mono-dominant stands covering small areas (0.06–1.4 ha) within palustrine and herbaceous wetlands in coastal Puerto Rico (Pratt et al. 2005). Sites where the cajeput tree is growing have signs of fire, have been partially drained, and were either farmed in the past or show signs of historical development (Pratt et al. 2005). While the cajeput tree is classified as intolerant of shade (Geary and Woodall 1990), there is no native tree species in Puerto Rico capable of withstanding this combination of flood and fire (Lugo 2004).

Other tree species like neem (*Azadirachta indica* A. Juss.), sweet lime (*Triphasia trifolia* (Burm. f.) P. Wilson), and genip (*Melicoccus bijugatus* Jacq.) have been designated as forest invaders, implying that they have the capacity to establish in forested areas in the USVI without depending on disturbances (Daley et al. 2012). However, forest stands sampled in the USVI show that the presence or dominance of species like sweet lime and genip is the result of secondary forest development after agricultural lands or timber plantations were abandoned (Atkinson and Marín-Spiotta 2015; Brandeis and Oswalt 2007; Brandeis and Turner 2013b). Island-wide forest inventories done in the USVI in 2004, 2009, and 2014 reveal that the number of individuals recorded for sweet lime has increased, while genip appears to be fluctuating (Brandeis and Oswalt 2007; Brandeis and Turner 2013b; Marcano-Vega and Williamson 2017; Miles 2018). Neem has not been found in forested sites in the USVI, and its habitat of establishment has been described as pastureland and cropland (Más and Lugo-Torres 2013). Ongoing studies are needed to determine the interspecific relationships between non-native and native flora and fauna, especially when both native and non-native birds and bats eat their fruits and disperse their seeds (Atkinson and Marín-Spiotta 2015; Daley et al. 2012). Other species introduced as ornamentals in the USVI like the Madagascar rubbervine (*Cryptostegia madagascariensis* Bojer ex Decne.) and the coral vine (*Antigonon leptopus* Hook. & Arn.) (Daley et al. 2012), also present in Puerto Rico (Acevedo-Rodriguez 2005), are similarly in need of field assessments to reveal more specific information regarding their influence on native biodiversity. Madagascar rubbervine has been confirmed as an invasive threatening endemic species in Northeast Brazil (da Silva et al. 2008) and coral vine as having the potential of displacing native species where it invades (CABI 2018), stressing the need for continuous monitoring in Puerto Rico and the USVI.

More than half of the non-native plant species present in Puerto Rico and the USVI have been intentionally introduced for horticulture (Rojas-Sandoval and Acevedo-Rodríguez 2015). Discussions of the wide range of timber and non-timber forest products offered by introduced plant species need to be an essential part of deliberations regarding their management to sensibly balance the conservation of native biodiversity and forest ecosystem services on the insular Caribbean (Benedetti and Negrón Flores 2012; Chamberlain et al. 2018; Davis et al. 2011; Marcano-Vega 2017; Marcano-Vega et al. 2015). Plant families exemplifying the highest numbers of naturalized species considered as invasive (i.e., Poaceae, Fabaceae) in Puerto Rico and the USVI represent intentional introductions of individuals used as nitrogen fixers, fodder, and ornamentals in agroforestry systems and to support cattle activities (Rojas-Sandoval and Acevedo-Rodríguez 2015). On Buck Island, St. Croix, USVI, there have been recommendations for perpetual treatments to remove non-native plant species (Clark et al. 2007), although most of the targeted species, originally cultivated for their offering of various non-timber products (e.g., genip, tamarind (*Tamarindus indica* L.)), have been naturalized for decades and are not known to replace native dry forest species (Atkinson and Marín-Spiotta 2015; Benedetti and Negrón-Flores 2012; Brandeis and Oswalt 2007; Brandeis and Turner 2013b; Little and Wadsworth 1964; Molina Colón et al. 2011). Other targeted species like tan tan have been shown to coexist with native species in novel forest types where tree planting is an option to accelerate natural regeneration or to foster particular species combinations (Atkinson and Marín-Spiotta 2015; Molina-Colón et al. 2011). The tree yellow trumpetbush (*Tecoma stans* (L.) Juss. ex Kunth) is the official flower that has been depicted on the USVI seal since 1991 (Government of the United States Virgin Islands, Office of Management and Budget 2013) and is included in the list for targeted extirpation to provide recruitment areas for native plant species (Clark et al. 2007). As far as we know, yellow trumpetbush is native to the Caribbean including St. Croix (Acevedo-Rodríguez and Strong 2012) and has not been shown to replace native forest plants (Brandeis and Oswalt 2007; Brandeis and Turner 2013b; Little and Wadsworth 1964).

Among non-native plants in aquatic habitats, the seagrass *Halophila stipulacea* (Forssk.) Asch. has spread from the Indian Ocean to the Mediterranean Sea and throughout the Caribbean, most likely assisted by commercial and recreational boat traffic (Willette et al. 2014). Since its arrival, it has intermixed or replaced native seagrasses (Rogers et al. 2014; Willette et al. 2014). *H. stipulacea* has been reported in St. Thomas and St. John, USVI, and Culebra, PR. Additional research has been recommended to ponder what can be considered as the positive or negative effects of *H. stipulacea* on native seagrass ecosystems and if any management action should be pursued (Rogers et al. 2014). Preliminary data from Willette and Ambrose (2012) showed that seagrass beds dominated by *H. stipulacea* had larger individual fish, fewer juvenile fish, and more fish species than seagrass beds dominated by the native manatee grass (*Syringodium filiforme* Kütz). Results also suggest that *H. stipulacea* has had a negative effect on manatee grass by displacement but sustained an equal or higher number of trophic groups than the native seagrass (Willette and Ambrose 2012). Rogers et al. (2014) advocated for more data on herbivory rates, selective feeding, and relative nutritional values of the native and introduced seagrass species to understand the role and effects of *H. stipulacea* beds in habitats for parrotfish, green sea turtles, sea urchins, and other herbivores in the Caribbean. Other suggested research refers to genetic investigations to determine the similarity of *H. stipulacea* in the Caribbean with the seagrass populations in the Indian Ocean and Mediterranean Sea and on the interaction of the introduced seagrass with native Caribbean seagrass (Willette et al. 2014).

Interactions with Non-Native Vertebrates

Mammals Borroto-Páez et al. (2012) published a table of mammals introduced to the West Indies. Populations of introduced rats, mongoose, cats, hogs, goats, donkeys, deer, dogs, and monkeys are generally discussed in the literature as threatening to local native populations of plants and animals. Populations of the brown rat or Norway rat (*Rattus norvegicus* Berk.) and the black rat or ship rat (*Rattus rattus* L.) are the result of human settlement in Puerto Rico and the USVI dating back to the sixteenth and eighteenth centuries (Campbell 1991; Clark et al. 2007; Daley et al. 2012; Joglar 2005). Their threat to the native biodiversity in the Caribbean was established as rat abundance was found to be negatively correlated with significant declines in avian diversity within small islands and cays (0.83–5500 ha) near Puerto Rico and the USVI (Campbell 1991). However, black rats are not regarded as a threat today because they have been integrated into the local food web (Ewel et al. 2013; Willig and Gannon 1996).

The mongoose (*Herpestes auropunctatus* Hodgson) is an example of an intentional introduction in many Caribbean islands. They were brought in for the purpose of controlling rats in sugarcane plantations during the nineteenth century (Seaman and Randall 1962). The adjustments of this species' interspecific relationships (i.e., prey switching) with the native biodiversity produced results counter to the human intentions. Initial accounts portray the mongoose as a severe predator of the black rat but afterward as a heavy predator of native ground-nesting birds, amphibians, and reptiles (Henderson 1991; Pimentel 1955). Moreover, there was an extraordinary recovery of the black rat population after predator-prey adjustments (Seaman and Randall 1962). Henderson (1991) describes at least seven extinctions of West Indian native amphibians and reptiles that were primarily attributable to the introduction of the mongoose, especially on the Lesser Antilles and smaller islands. Numerous accounts of the predation of native fauna (especially amphibians and reptiles) by the mongoose have been registered, as has its emergence as the major vector of rabies and leptospirosis in the area (Borroto-Páez et al. 2012; Pimentel 1955, 2007). Pimentel et al. (2005) estimated that the mongoose causes around $50 million in damages per year in Puerto Rico and Hawaii due to public health effects and the predation of poultry, native birds, reptiles, and amphibians.

On the other hand, the successful translocation and establishment of the critically endangered St. Croix ground lizard (*Ameiva polops* Cope) to Buck Island is an example of a fruitful interagency collaboration, where translocation strategy, life history of the species, habitat quality, and monitoring were identified as indispensable factors for success (Fitzgerald et al. 2015). The strategy included eliminating the mongoose and black rat from Buck Island before translocation.

A study done in Puerto Rico by Guzmán-Colón and Roloff (2014) suggests that the coastal forests of Puerto Rico support a greater abundance of mongoose than wet forests and that the abundance is associated with human activity (e.g., forest trails, garbage). Further studies are planned to confirm initial baseline data and inform mongoose management plans (Guzmán-Colón and Roloff 2014). During the turtle breeding season, the US Fish and Wildlife Service conducts mongoose control near sea turtle nesting sites at the Sandy Point National Wildlife Refuge on St. Croix (Barun et al. 2011), where localized trapping is seen as an alternative providing protection for sensitive insular species facing multiple effects of land-use history. Coblentz and Coblentz (1985) have documented that hawksbill turtles (*Eretmochelys imbricata* L.) nesting in beaches of the USVI, in which they trapped mongooses, experienced no known loss of eggs or young during the period of their study, compared to as much as 23% recorded previously. The Virgin Islands National Park has an ongoing program for controlling mongoose using box traps on the beaches of St. John, USVI (Barun et al. 2011). In Puerto Rico, intense egg predation of the Puerto Rican freshwater turtle (*Trachemys stejnegeri stejnegeri* Schmidt) by the mongoose has been found on the Humacao Nature Reserve of Puerto Rico (León and Joglar 2005), a wetland area that was drained at the beginning of the twentieth century for growing sugarcane and cattle grazing (RNH 2015). Eradication of the mongoose has been suggested to have a direct positive effect on the recruitment of the early life stages of the turtle (Joglar et al. 2007). In addition, the USDA Forest Service and USDA Animal and Plant Health Inspection Service (APHIS) spend around $10,000 per year on mongoose control to protect the critically endangered Puerto Rican parrot (Barun et al. 2011). The Puerto Rican parrot was a common endemic bird into the nineteenth century, with an estimated population of around one million individuals. In 1975, this number declined drastically to 13 birds due to dramatic habitat loss, hunting, and the pet trade (Oberle 2000). All of these instances exemplify present remedies to the product of past human actions.

Mona Island is another example of the effects of introduced mammals in the Archipelago of Puerto Rico. Mona is a 22 mi^2 reserve managed by the Department of Natural and Environmental Resources of Puerto Rico (DNER). It is located approximately 68 km west of the main island and is the single home of the threatened Mona Island iguana (*Cyclura cornuta stejnegeri* Barbour & Noble) (Brandeis et al. 2012; Garcia et al. 2007). Major threats to the iguana come from habitat changes related primarily to litter accumulation of the introduced Australian pine (*Casuarina equisetifolia* L.) at nesting sites and to predation on juveniles by introduced mammals such as feral cats (*Felis silvestris catus* Schreber), as well as on eggs by wild hogs (*Sus scrofa* L.) (Garcia et al. 2007; Wiewandt and García 2000). Wild hog predation on Mona Island iguana eggs has been documented as reaching 100% in dry years (Wiewandt and García 2000). This positions climate change as a potential catalyst of predation rates, as most climate projections for the Caribbean suggest a higher frequency of droughts within the region (UNEP 2008). A survival rate of 0.22 has been recorded during the first 20 weeks of life for the Mona Island iguana hatchlings, which can include effects of potential native predators such as two snakes, *Alsophis* spp. Fitzinger and *Epicrates* spp. Wagler, and two raptors, *Falco* spp. L. (Pérez-Buitrago and Sabat 2007). Introduced feral goats (*Capra* *hircus aegagrus* Erxleben) are also a threat to the habitat of the Mona Island iguana, coupled with secondary growth areas regenerated after a past disturbance regime that included burning and vegetation clearance (Wiewandt and García 2000). Goats frequently feed on at least four critically endangered plant species in the Cactaceae, Portulacaceae, Orchidaceae, and Rhamnaceae families, some with an unknown reproductive ecology (e.g., Taylor's jujube (*Ziziphus taylorii* (Britton) M.C. Johnst.)) (Meléndez-Ackerman et al. 2008). A study done in Mona Island revealed that the regeneration of understory vegetation following ungulate exclusion was low, suggesting synergistic effects between herbivore-plant interactions and hurricane disturbance on forest conditions (Rojas-Sandoval et al. 2016b). Even so, goats are by now an integral food web component on Mona Island, and no records of goat-driven extinctions have been described five centuries after introduction (Ewel et al. 2013; Lugo et al. 2012). Long-term monitoring efforts on the ecology of natural plant populations and on the effects of ungulate exclusion are encouraged and should be supported (Ewel et al. 2013; Meléndez-Ackerman et al. 2008; Rojas-Sandoval et al. 2016b).

Wild hogs in conjunction with beach erosion are also documented as major threats to the hawksbill sea turtle on Mona Island; fencing beaches to prevent hog predation has corresponded with an increase in the number of hawksbill nests in the last decade (Joglar et al. 2007). The DNER works with other entities to pursue several intervention strategies to manage introduced mammals on Mona Island. These include moving the hunting season to a period not overlapping with the Mona Island iguana nesting and incubation seasons, creating additional nesting sites and fencing them in a way that prevents feral hogs from entering while allowing the free passage of iguanas, opening clearings in Australian pine stands, and fencing remote nest sites (Wiewandt and García 2000). Additional strategies include an ongoing cat eradication program through trapping and hunting, coupled with a headstarting program of young iguanas that consists of keeping hatchlings in captivity until they reach a size less vulnerable to predators such as feral cats and monitoring the headstarted iguanas through radio tracking and direct observation (Garcia et al. 2007).

Traps, fencing, and radio tracking are also planned as intervention strategies in the Virgin Islands National Park where feral species of introduced mammals include hogs, goats, and cats, all of which are a threat to the native biodiversity (Borroto-Páez et al. 2012; Burde and Feldhamer 2005; US Department of the Interior National Park Service Southeast Region 2003). The feral goat population in St. John USVI, an island of approximately 62 km^2, has been estimated at 600–1000 individuals (Burde and Feldhamer 2005). Feral cats are reported as predators or potential predators of numerous reptiles, amphibians, and insects on USVI (Borroto-Páez et al. 2012), but without correlating quantitative data on their effects on native prey populations. Demographic studies of native prey should be encouraged to determine population status and to elucidate key underlying factors explaining dynamics. In Puerto Rico, the predation of bats by feral cats has been described as a factor facilitated by the island's landscape as a mosaic of urban and natural areas (Rodríguez-Durán et al. 2010). In the Virgin Islands National Park, 25 threatened and endangered plant species have been listed as potentially being affected by wild hogs. Pre- and post-reduction hog surveys along with long-term monitoring programs of native vegetation have been included in a sustained reduction plan (US Department of the Interior National Park Service Southeast Region 2003). Other introduced mammals reported in the USVI are the white-tailed deer (*Odocoileus virginianus* Zimmermann), feral donkey (*Equus asinus* L.), and feral dog (*Canis lupus familiaris* L.) (Borroto-Páez et al. 2012). The National Park Service's view of the donkey (Fig. A8.9) as an invasive spe-

Fig. A8.9 Although viewed as an invasive mammal, feral donkeys (*Equus asinus*) represent a valued part of life and history to local St. Johnians. (Photo credit: Humfredo Marcano-Vega, USDA Forest Service Southern Research Station) cies met with an alternative view from local St. Johnians, who see a species representing a valued part of life and history (Fortwangler 2009). Wild populations of deer and goats are also present in Culebra Island, PR, where the whitetail deer was introduced in 1966, and intervention strategies for controlling these species have been proposed (US Department of the Interior FWS Southeast Region 2012).

Among the non-native mammals in Puerto Rico are wild populations of monkeys, introduced for research purposes (e.g., virus vaccine program, free-ranging ecology). Rough estimates suggest a population of 500 monkeys in southwest Puerto Rico (Jensen et al. 2004). The introduced Asian rhesus monkey (*Macaca mulatta* Zimmermann), the African patas monkey (*Erythrocebus patas* Schreber), and the Central and South American squirrel monkey (*Saimiri sciureus* L.) thrive in Puerto Rico (Grana-Raffucci 2006–2012; Jensen et al. 2004). A survey of rhesus monkeys (n = 57) found that 72% had evidence of B virus infection (Jensen et al. 2004). Nonetheless, risks of transmission to humans are low because an average of only 2–3% of seropositive monkeys shed the virus. An additional public health concern, however, is the potential for monkeys to be exposed to rabies through their contact with the mongoose, which is the major vector of the virus in Puerto Rico (Engeman et al. 2010). The frequency of human encounters and disease exposure is expected to increase as monkeys continue to expand their range and population (Jensen et al. 2004). Populations of rhesus and patas monkeys have extended their range into productive agricultural lands, resulting in substantial crop losses to commercial farmers. Economic losses by commercial farmers have increased from $1.13 to $1.46 million USD from the years 2002 to 2006, and control measures for monkey populations are viewed as highly challenging (Engeman et al. 2010). Research advancements on wildlife contraceptive agents or immunocontraceptives may offer new humane, nonlethal alternatives to manage introduced mammal populations in Puerto Rico and the USVI if deemed necessary (The Humane Society of the United States 2016; US Department of Agriculture APHIS NWRC 2013).

Amphibians and Reptiles Six introduced species of amphibians are described as threatening to native amphibians in Puerto Rico as competitors, predators, and vectors of pathogens and disease (Burrowes and Joglar 2005; Joglar 2005). Coupled with habitat loss, the introduced cane toad (*Rhinella marina* L.) is considered a major factor in explaining the threatened and critically endangered status of the Puerto Rican crested toad (*Peltophryne lemur* Cope) (Joglar et al. 2007). In an effort to increase the Puerto Rican crested toad population, captive-born toadlets are being released, some into artificial ponds. Ongoing research plans include a dietary assessment and characterization of the health threats prevalent in the cane toad individuals cohabiting with Puerto Rican crested toads (Joglar et al. 2007). The fact that *Salmonella* Lignieres has been isolated from the liver of cane toad individuals in Puerto Rico has also been highlighted as a possible public health concern to people on the island (Burrowes and Joglar 2005). The cane toad is also an introduced species in the USVI, where it is estimated to be competing with the native white-lipped frog (*Leptodactylus albilabris* Günther) for limited freshwater resources (Platenberg 2007). The non-native Cuban treefrog (*Osteopilus septentrionalis* Duméril & Bibron) is also believed to be competing with the white-lipped frog in the USVI for ephemeral freshwater, and there is circumstantial evidence of its negative effects on populations of native frogs and lizards (Platenberg 2007).

The green iguana (*Iguana iguana* L.) (Fig. A8.10) was introduced to Puerto Rico in the 1970s and has naturalized and dispersed widely, especially around the main island

Fig. A8.10 The introduced green iguana (*Iguana iguana*) is primarily described as an herbivore, but gut content analysis in Puerto Rico has indicated that it is an opportunistic omnivore (e.g., crabs (*Uca* spp.)). Its naturalization has raised concern for the conservation of endangered and threatened native species and economic losses on nature reserves, gardens, agricultural lands, and airports. Nevertheless, interdisciplinary research shows ways in which the green iguana is providing new opportunities for ecotourism destinations in Puerto Rico. (Photo credit: Humfredo Marcano-Vega, USDA Forest Service Southern Research Station)

(López-Torres et al. 2012). Concerns about the presence of green iguanas in Puerto Rico are mostly related to the habitats of listed endangered or threatened native species, nature reserves, gardens, agricultural lands, and airports (Falcón et al. 2012, 2013; García-Quijano et al. 2011; López-Torres et al. 2012). The reproductive season and nesting sites of the green iguana and the leatherback turtle (*Dermochelys coriacea* Vandelli) overlap. Concerns are mainly due to the digging habits of the iguana and its potential for destroying leatherback eggs, as well as the iguanas' potential for attracting predators that can feed on leatherback eggs (e.g., dogs, mongoose). The digging and herbivory activities of the green iguana in Las Cabezas de San Juan Nature Reserve are resulting in annual investments of up to $10,000 USD for road repair and young tree losses in the plant nursery (López-Torres et al. 2012). The green iguana activities have also affected crops such as yams (*Dioscorea* spp. L.), yautías (*Xanthosoma* spp. Schott), pumpkins (*Cucurbita* spp. L.), and melons (*Cucumis* sp. L.), particularly in southern Puerto Rico (López-Torres et al. 2012). Additionally, although the green iguana is primarily described as an herbivore, gut content analysis in Puerto Rico indicates that it is an opportunistic omnivore (e.g., crabs (*Uca* spp.)), revealing the need for further research regarding community-level function and effects (Govender et al. 2012).

Concerns about economic losses due to green iguana incursions in aircraft traffic areas have also been highlighted for their potential to diminish public trust in the air transport industry. A study over a 2-month period (October–November 2001) in the Luis Muñoz Marín International Airport (SJU) of San Juan, PR, found that operations in airstrip areas had to be stopped on six occasions (Engeman et al. 2005). With an average of 233,000 operations annually at SJU (Engeman et al. 2005), results of the study can be transduced to a rate of 15.4 occasions per 100,000 movements. Events at SJU resulted in the interruption of operations, though they do not seem to have represented actual strikes; rather, they recorded the frequency with which observed green iguanas were judged to be a hazard for a collision with an aircraft (Engeman et al. 2005). In comparison, a rate of 10.8 strikes per 100,000 movements was reported in the United States during the year of the study (US Department of Transportation Federal Aviation Administration and US Department of Agriculture APHIS 2014). Current control measures for the green iguana in Puerto Rico are costing the US Federal Aviation Administration and the Ports Authority $98,000 annually (Falcón et al. 2012).

Ninety-six percent of reports of civil aircraft damage resulting from wildlife strikes in the United States are due to birds. Of a total of 143,800 strikes reported from 1990 to 2013, only 189 (0.1%) involved reptiles, from which 9 refer to strikes with green iguanas (US Department of Transportation Federal Aviation Administration and US Department of Agriculture APHIS 2014). Four of those reports involved a negative effect on a flight, but none gave an account of damage to aircraft or economic losses. This stresses the need for ongoing studies of wildlife hazard assessments with more comparable data to derive implications and potential alternative actions. Considering that it is estimated that most airports account for less than 20% of actual strikes, efforts should be directed to make reporting an essential aspect of contributing to wildlife management (Engeman et al. 2005). Researchers have recommended that a management plan be developed, as the DNER has begun a population control program for the green iguana (López-Torres et al. 2012).

Ethnographic interviews of residents of the San Juan Bay Estuary (SJBE) in Puerto Rico reveal that most people found the systematic killing of green iguanas as a control measure objectionable (García-Quijano et al. 2011). Interest in the green iguana is generating increased participation in local and overseas tourism to coastal and riparian areas, fostering iguana-watching experiences through kayaking and boat tours, as well as iguana-themed souvenirs on the island (García-Quijano et al. 2011). These circumstances provide new opportunities for SJBE residents and for new interest in estuarine ecosystems as ecotourism destinations in Puerto Rico. An adult green iguana census, also by García-Quijano et al. (2011), showed a positive spatial association between human settlements and iguana populations and that iguanas were significantly aggregated toward the edges of mangrove forests. These results illustrate how green iguanas can represent high defoliation rates on localized mangrove stands near forest edges exposed to the effects of human activities. Even so, the majority of the mangrove forests studied were not being affected by the iguanas. The authors emphasized how a conceptual framework for understanding ecosystem function and species introductions needs to incorporate the sociocultural context of host ecosystems (García-Quijano et al. 2011). The green iguana is also present in the USVI, especially around resorts and restaurants on St. Thomas, but neither its origin nor its ecosystem effects have been identified (Platenberg 2007). The expansion of the green iguana across the Caribbean Basin raises conservation concerns, as modeling techniques predict climatic suitability for new invasions in, essentially, all of the insular Caribbean (Falcón et al. 2012). Falcón et al. (2013) offered a review of control efforts and response to management of the green iguana.

Other introduced reptiles in Puerto Rico include the caiman (*Caiman crocodilus* L.) and the red-eared slider turtle (*Trachemys scripta elegans* Wied-Neuwied). The caiman was apparently introduced in the 1950s–1960s and is now naturalized, especially around the northern coast of Puerto Rico (Joglar 2005). There are no ongoing studies on the effects of caimans on the local biodiversity. The introduced red-eared slider turtle is reported to be widespread in the wetlands of Puerto Rico, raising concerns about interspecific competition and hybridization with native turtle populations. Scholars have underscored the need for local agencies on the island to enforce trade laws (Joglar et al. 2007), with research efforts focused on monitoring local populations of the Puerto Rican freshwater turtle through mark-recapture studies and investigation of its reproductive activity and on the identification of predators and non-native turtles within study areas (León and Joglar 2005). The Puerto Rican freshwater turtle population is estimated as being close to threatened due to its restricted distribution and the effects of habitat destruction, human consumption, introduced turtle populations, and (as discussed earlier) intense egg predation by the mongoose (León and Joglar 2005).

Additional introduced species of reptiles and amphibians that are presumed to have an effect in the USVI include the corn snake (*Pantherophis guttatus* L.), the red-footed tortoise (*Geochelone carbonaria* Spix), the house gecko (*Hemidactylus mabouia* Gray), the red-eared slider, the common ground lizard (*Ameiva exsul* Cope), the Puerto Rican coqui (*Eleutherodactylus coqui* Thomas), and the Lesser Antillean whistling frog (*Eleutherodactylus johnstonei* Barbour) (Platenberg 2007). However, their effects on native populations are, for the most part, undetermined, and long-term monitoring programs coupled with habitat protection could be supported to safeguard native herpetofaunal species of conservation concern (Platenberg and Boulon 2006). The corn snake is present in St. Thomas, where it has been highlighted as a potential threat to the endangered USVI tree boa (*Epicrates monensis granti* Stull) (Platenberg 2007).

Birds Members of the Psittacidae (parrots and parakeets), Ploceidae (weaver finches), and Estrildidae (waxbills and allies) families are included among the most numerous introduced bird species in Puerto Rico (Delannoy Juliá 2005). The canary-winged or white-winged parakeet (*Brotogeris versicolurus* Statius Muller) and the monk parakeet (*Myiopsitta monachus* Boddaert) figure among the most successfully established and widespread psittacines in Puerto Rico (Oberle 2000; Raffaele 1989). Studies about interactions of these species with the local biodiversity are not available. Other bird species described as introduced, such as the pin-tailed whydah (*Vidua macroura* Pallas) and the shiny cowbird (*Molothrus bonariensis* Gmelin), are parasitic on other bird species' nests. While the pin-tailed whydah was introduced from Africa, the shiny cowbird is a species originally from South America and Trinidad and Tobago; it expanded its range during the twentieth century, taking advantage of habitat changes due to agricultural activities throughout the Caribbean (Oberle 2000). Shiny cowbirds deposit their eggs in other bird species' nests and exploit parental care, affecting its principal host, the endemic yellow-shouldered blackbird (*Agelaius xanthomus* P. L. Sclater) in Puerto Rico. It is reported that almost all yellow-shouldered blackbird nests have been parasitized by the shiny cowbird since its introduction in 1955 (Delannoy Juliá 2005; Oberle 2000; Raffaele 1989). A trapping program to remove cowbirds from the blackbird's breeding range has been established and is considered a control measure that will probably have to continue indefinitely to avoid the blackbird's eradication from mainland Puerto Rico (Oberle 2000). Although it is documented that cowbirds had reached Mona Island by 1971 (Post and Wiley 1977), they are now absent, and a non-threatened subspecies of the blackbird is present on Mona and Monito Islands (Oberle 2000; Raffaele 1989). The arrival dates of the shiny cowbird in the Lesser Antilles are described as suggesting natural "island hopping" from Trinidad and Tobago (Cruz et al. 1995), while its range movements in North America support the hypothesis of a dispersal pattern unaided by human introductions (Post et al. 1993).

Fishes The Indo-Pacific red lionfish (*Pterois volitans* L.) has spread through the Western Atlantic and Caribbean since 1985, apparently after its release or escape from marine aquaria into Florida coastal waters (Albins and Hixon 2013). As a competitor and generalized predator of native reef fishes, and with its spread into critical nursery ecosystems for native juvenile reef fishes (i.e., mangroves, seagrass beds), the lionfish is considered a potential threat to the recruitment of reef fishes, some of which are the prey of important fishery species or are important food species such as groupers and snappers (Albins and Hixon 2013; Diller et al. 2014; Morris and Akins 2009). A field experiment has shown that the lionfish alone was able to reduce prey fish by 90% after 2 months, but authors indicate that more data are still needed before projecting into the future (Albins and Hixon 2013). Studies in the Bahamas have shown that predation of red lionfish on algae grazers such as parrotfishes, surgeonfishes, and damselfishes has the potential to shift coral reefs to an algal-dominated community, causing declines in biodiversity (Lesser and Slattery 2011). The high growth, reproductive, and consumption rates of the lionfish coupled with its apparent resistance to predation has made biological control a theme of debate (Albins and Hixon 2013; Diller et al. 2014). Yet, recent experimental studies by Diller et al. (2014) have suggested that native predators are capable of consuming healthy, tethered lionfish and that their inexperience preying on lionfish can be overcome by conditioning. In addition, the gourmet food industry is taking advantage of the lionfish's spread, and a cookbook has been written, providing information on ways to catch and prepare the fish (Akins 2012; CBS Interactive 2015).

The nature of Caribbean islands as case studies of the social and ecological consequences of introduced species demands supporting long-term monitoring, education, and management programs at the regional level.

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