

Economic Impacts of Invasive Species in Forests

Past, Present, and Future

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Biological invasions by nonnative species are a by-product of economic activities, with the vast majority of nonnative species introduced by trade and transport of products and people. Although most introduced species are relatively innocuous, a few species ultimately cause irreversible economic and ecological impacts, such as the chestnut blight that functionally eradicated the American chestnut across eastern North America. Assessments of the economic costs and losses induced by nonnative forest pests are required for policy development and need to adequately account for all of the economic impacts induced by rare, highly damaging pests. To date, countrywide economic evaluations of forest-invasive species have proceeded by multiplying a unit value (price) by a physical quantity (volume of forest products damaged) to arrive at aggregate estimates of economic impacts. This approach is inadequate for policy development because (1) it ignores the dynamic impacts of biological invasions on the evolution of prices, quantities, and market behavior, and (2) it fails to account for the loss in the economic value of nonmarket ecosystem services, such as landscape aesthetics, outdoor recreation, and the knowledge that healthy forest ecosystems exist. A review of the literature leads one to anticipate that the greatest economic impacts of invasive species in forests are due to the loss of nonmarket values. We proposed that new methods for evaluating aggregate economic damages from forest-invasive species need to be developed that quantify market and nonmarket impacts at microscales that are then extended using spatially explicit models to provide aggregate estimates of impacts. Finally, policies that shift the burden of economic impacts from taxpayers and forest landowners onto parties responsible for introducing or spreading invasives, whether through the imposition of tariffs on products suspected of imposing unacceptable risks on native forest ecosystems or by requiring standards on the processing of trade products before they cross international boundaries, may be most effective at reducing their impacts.

Key words: forest pests; pathogens; invasive; nonnative; economic; contingent valuation; hedonics; ecosystem services; chestnut blight; management; trade; gypsy moth

Introduction

Biological invasions are among the most serious environmental problems currently facing society. Through millions of years of evolution,

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the world's biota has been largely separated into independently coevolved ecological communities, but globalization has broken down barriers to movement, resulting in an exchange of species among and within continents. While many invasions have been inconsequential, certain species have profoundly altered their invaded environments, affecting ecosystem processes and ultimately impacting human societies.

The biological (US Congress OTA 1993; Williamson 1996; National Research Council 2002; Lockwood *et al.* 2007) and economic (General Accounting Office 2000; Perings *et al.* 2002; Pimentel *et al.* 2000) aspects of biological invasions in general have been reviewed. There are also some general reviews covering ecological aspects of invasions in forests (Liebhold *et al.* 1995; Lovett *et al.* 2006), but we are not aware of a previous review focusing on the economic aspects of biological invasions in forest ecosystems.

In this review we discuss the economics of invasive species in forests. We start with a brief description of the biological invasion problem and its ecological effects on forests. Next we describe the economic underpinnings of invasions; globalization is the root cause of most invasions, and there are complex economic processes driving this global trend. Then we describe the various economic consequences of invasions, including impacts on nonmarket values of forests. Next we discuss the complications of estimating aggregate economic impacts based on a series of microeconomic studies. Finally, in the last section we discuss the implications of impacts of invading species on trade policies.

Ecological Threats Posed by Nonindigenous Forest Pests

Humans have transported species from one geographic region to another, both intentionally and unintentionally, for millennia. It's obvious that, in many cases, intentional intro-

ductions of nonnative species have provided tremendous social benefits. The production of wheat, which was domesticated from wild ancestors in southwest Asia, provides a good example. Today wheat is grown throughout the world and is one of the most important staple food crops (Prescott-Allen and Prescott-Allen 1990). Within the forestry sector, plantations of nonindigenous tree species have proven to be highly profitable. Monterey pine (*Pinus radiata*), which is native to a limited region in the fog belt of coastal California, accounts for approximately 90% of New Zealand's 1.8 million hectares of forest plantations (Anon 2007). On the other hand, some intentionally introduced species become invasive and can cause significant economic damages. This includes trees such as *Acacia mearnsii* (black wattle) in southern Africa and *Prunus serotina* (black cherry) in Europe (de Wit *et al.* 2001; Chabrierie *et al.* 2007). Other intentionally introduced plants that are having a detrimental impact on forest productivity include cogongrass (*Imperata cylindrica*), which is considered one of the ten worst weeds in the world, invading tropical and subtropical forested and nonforested areas in 73 countries. Cogongrass forms large, monotypic expanses, with Asia reporting over 35 million acres affected (Garrity *et al.* 1997). Another example is kudzu (*Pueraria montana*), which was intentionally introduced into the southeastern United States to prevent soil erosion and has inadvertently caused millions of dollars of losses to timber productivity (Forseth and Innis 2004).

The U.S. government defines an invasive species as "an alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health" (Federal Register, Presidential Documents 1999). Invasion is a process in which an organism moves from its native habitat (donor region) and ultimately becomes a pest in a new habitat (recipient region) (Vermeij 1996). This process occurs in discrete stages—arrival, establishment, and population spread—which ultimately can cause debilitating impacts to ecological and economic systems. At each stage

of the process, there is a low probability that a nonindigenous organism will progress to the next step. The sequence of transition probabilities has been referred to as the “tens rule,” based on the notion that roughly one out of every 10 organisms makes the transition from one stage to the next (Williamson and Fitter 1996). However, transition probabilities may be much higher for some groups of organisms (Jeschke and Strayer 2005).

Because nonindigenous organisms have not coevolved with native biota, they threaten the structure and functioning of native ecosystems (Wilcove *et al.* 1998; Simberloff 2000; Millennium Ecosystem Assessment 2005). Perhaps the most dramatic example can be found in the forests of eastern North America where, following European settlement, a sequence of nonindigenous pathogens and pests removed the American chestnut (*Castanea dentata*)—formerly a dominant or codominant tree species in forest communities across eastern North America—from the overstory. Beginning in 1825, a root pathogen (*Phytophthora cinnamomi*) killed chestnut trees in the southern portion of the tree’s natural range, and by the end of the century, *P. cinnamomi* had killed most chestnut trees in low, moist areas (Schlarbaum *et al.* 1997; Anagnostakis 2001). Thus, when chestnut blight (*Cryphonectria parasitica*), another nonnative pathogen, was first detected in 1904, the range of American chestnut was already restricted. Within a period of 50 years, *C. parasitica* swept over the entire range of American chestnut, killing virtually every overstory *C. dentata* in an area representing roughly 3.6 million hectares (Anagnostakis 2001).

The removal of chestnut over vast forest regions of eastern North America has left lasting effects on forest ecosystem structure and function, including decreased leaf-litter processing rates, decreased quality of litter inputs, decreased growth rates of aquatic invertebrates, and increased input of large woody debris into streams (Smock and MacGregor 1988; Hedman *et al.* 1996; Wallace *et al.* 2001). There was also a drastic change in the dominance

of tree species across its range (Mackey and Sivec 1973; Day and Monk 1974; Myers *et al.* 2004), as well as decreased abundance of cavity-nesting birds following the blight (Haney *et al.* 2001). American chestnut accounted for approximately 25 to 40% of the standing timber in some stands of the eastern deciduous forest. Economic costs associated with the demise of this species included losses in livelihood for those who collected the nuts for food, and the loss of a high-quality timber that was used extensively for construction and wood products (Myers *et al.* 2004). Microeconomic theory suggests that the reduction in timber inventories during the decades following the catastrophic decline of an important timber species will induce higher timber prices and a loss of profits for timber growers and wood-using firms (Holmes and Lee 1991). In addition, by diminishing or degrading the suite of nontimber benefits provided in forests formerly dominated by chestnuts, such as landscape aesthetics or wildlife habitat, a substantial loss in economic welfare is incurred. Because the change in ecosystem structure and function is fundamentally irreversible, the loss in economic welfare is potentially enormous.

Although striking, the loss of chestnut from the forests of eastern North America is not an isolated example of large-scale forest change caused by nonnative pests, pathogens, and plants. Invasions by nonnative species continue to drive widespread changes in forest composition and structure around the world. The pinewood nematode was accidentally introduced to Japan in the early 1900s, and over the last century has eliminated pines from vast forested areas in Japan and, more recently, in China (Kishi 1995). In Hawaii, an array of invasive plants is changing the structure of the rain forest (Asner *et al.* 2008). Emerald ash borer (*Agrilus planipennis*), discovered in Michigan in 2002, appears poised to remove an entire genus from natural and urban forests throughout the upper Midwest and eastern United States (Poland and McCullough 2006; U.S. Government Accountability Office 2006).

Although it is clear that nonnative organisms have had, and continue to have, substantial impacts on native forests, it is difficult to quantify long-term ecological impacts, both because forest ecosystems adjust and evolve in response to invasions and because most forest ecosystems around the world are subject to many anthropogenic disturbances. For example, palynological studies suggest that *C. dentata* increased in abundance in some areas following European settlement and prior to the arrival of chestnut blight (Fuller *et al.* 1998; Paillet 2002). Today eastern North American forests reflect a 250-year history of logging, clearing, burning, fire suppression, deer grazing, and natural reforestation, as well as the impacts of nonnative plants, insects, and pathogens (Foster *et al.* 1998; Fuller *et al.* 1998; Abrams 2003). These forests do not appear to be returning to the composition and structure of pre-European forests, but instead are much more homogeneous across climatic gradients (Foster *et al.* 1998). In fact, human disturbances over 100 years ago continue to leave a legacy of nonnative plant species (Von Holle and Motzkin 2007). In the face of deforestation and climate change, it is impossible to know what these forests would look like in the absence of chestnut blight and other nonnative pathogens, insects, and plants. Therefore, a comparison of the level of goods and services provided by forests with and without invasive species is confounded by the fact that a counterfactual “without” scenario cannot be easily defined. Transitory impacts, which occur while ecosystems and economies adjust to biological invasions, are likely to be more reliably evaluated than long-term impacts.

The stages of invasion have been described in terms of a series of ecological barriers or filters that may prevent an organism from proceeding to the next stage (Richardson *et al.* 2000; Colautti and MacIsaac 2004). Figure 1 illustrates the stages of a biological invasion, the management actions that can be applied at each stage, and the economic implications. The first stage in the invasion process is the arrival of an or-

ganism from a donor region. If an organism is undetected during transport and successfully escapes into a native ecosystem, establishment depends upon a suitable climate and proximity to food resources. When populations and ranges of nonnative organisms are small, eradication has the best chance of success due to the Allee effect (adverse effects of low population densities) (Williamson 1996; Liebhold and Bascompte 2003; Liebhold and Tobin 2008). However, if an organism becomes successfully established and is not eradicated in a timely manner, favorable conditions may allow it to spread throughout suitable ecosystems of the recipient country. Eventually the invasive species may saturate the entire geographical extent of suitable habitat. Even at this point, managers may continue to suppress populations in order to minimize damages, but society is also likely to adapt to the presence of the invasive species and its impacts.

Economic Underpinnings of Biological Invasions

One of the key complexities of biological invasions is the inherent interdependency of ecology and economics. While the economic consequences of ecosystem changes produced by invading species are often well publicized, the invasions themselves are ecological consequences of economic processes. In particular, they are externalities, or by-products, of economic activity. As a result, the potential costs of invasive species are generally not factored into decisions about exports, imports, and domestic transport of goods and people, all of which are pathways for the introduction and spread of invasive species. Thus, invasions act as a conduit by which one group of economic forces influences other economic processes.

Economic processes affect each of the four invasion stages shown in Figure 1, but none more so than arrival. Crosby (1993) documents how the beginnings of the current onslaught of species invasions can be traced back to the

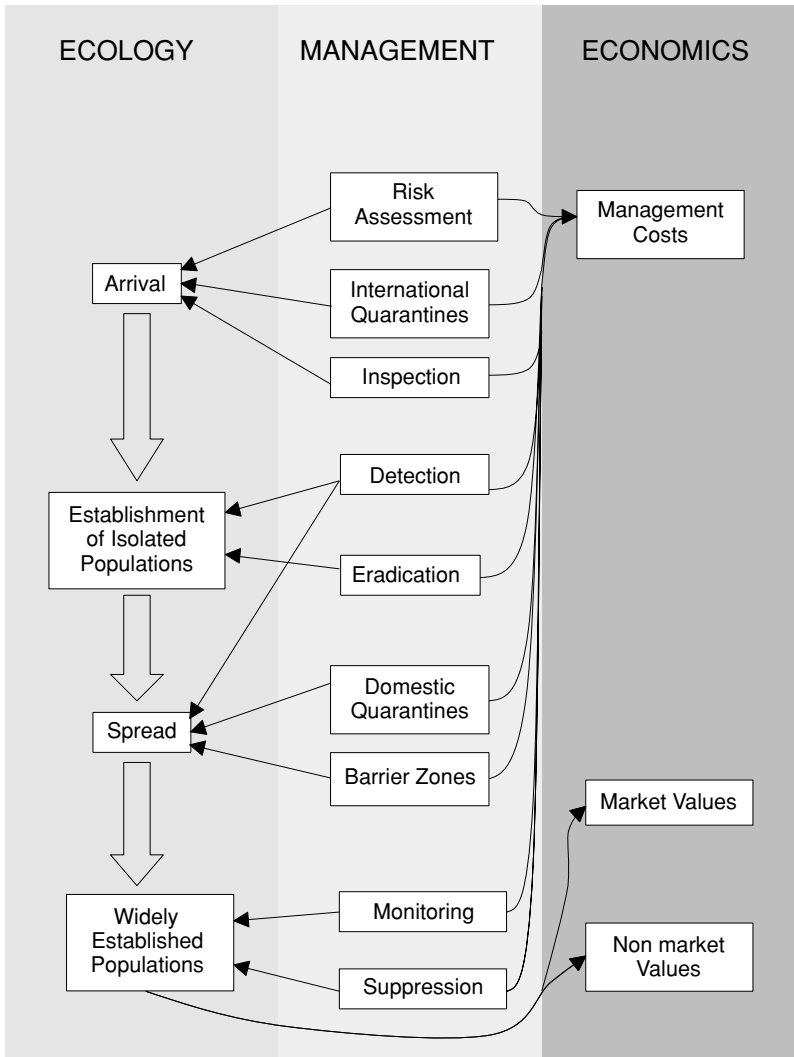


Figure 1. The stages of a biological invasion are linked to management actions that can be applied at each stage; each of these management actions has economic implications.

international expansion of colonial Europe. This era of imperialism was driven by a variety of economic processes, including the demand for raw materials to process within Europe's growing industrial centers, as well as the unsatiated demand for tillable land in Europe that led immigrants to seek better agricultural opportunities. These colonists brought with them (both intentionally and inadvertently), a variety of species, many of which established outside of Europe and forever altered native ecosystems.

As economies around the world have developed, international trade of virtually every commodity has increased and this trend has been the driving factor behind the steady increase in arrivals of nonindigenous species (Stanaway *et al.* 2001; Vilá and Pujadas 2001; Levine and D'Antonio 2003; Work *et al.* 2005; Caton *et al.* 2006; McCullough *et al.* 2006). The National Invasive Species Council (2005) has mapped the great diversity of human-mediated pathways by which species are introduced. Many invasive vertebrate and plant

species have been deliberately introduced, often through the pet and horticultural trades (Jeschke and Strayer 2005; Dehnen-Schmutz *et al.* 2007). Forest pests and pathogens are prone to hitchhike on other commodities or packing material (Mack *et al.* 2000). Solid wood packing material (crating, dunnage, and spoolage) as well as roundwood have been implicated as particularly important pathways (Brockerhoff *et al.* 2006; Haack 2006; Piel *et al.* 2008). Air passenger luggage is another significant invasion pathway (Liebhold *et al.* 2006; Tatem and Hay 2007).

While most arrivals of invasive species in modern times are almost exclusively anthropogenic, establishment and spread continue to be driven by the interplay of ecology and economics. A large body of literature indicates that establishment success is closely related to the frequency and size of arrivals, termed “propagule pressure” (Leung *et al.* 2004; Von Holle and Simberloff 2005; Lockwood *et al.* 2005). When species are repeatedly introduced, Allee effects and stochastic dynamics are much less likely to cause founding populations to go extinct. As a consequence, the same economic forces that widen invasion pathways and increase arrival rates can also be expected to enhance establishment. There is more mixed evidence that human disturbance of ecosystems may reduce their “biotic resistance” to the establishment of invasive species, particularly plants (Levine *et al.* 2004).

The spread of invading species is also affected by economic factors. As with arrivals, humans may either deliberately or inadvertently transport invasive species over much greater distances than their natural rate of diffusion (Shigesada and Kawasaki 1997). For example, the importance of human-mediated spread has been demonstrated for gypsy moth in the United States (Lippett *et al.* 2008) and the horse-chestnut leaf miner in Europe (Gilbert *et al.* 2004). The movement of firewood, for recreational and residential heat purposes, has been shown to play a critical role in mediating spread of wood-boring insects (Muirhead *et al.* 2006).

Movement of nursery stock greatly enhances the spread of a variety of insects and diseases (Poland and McCullough 2006).

Policy or management can be applied at each stage of Figure 1 to reduce the probability of successful invasion or to reduce the associated damages (Leung *et al.* 2002; Lodge *et al.* 2006; Liebhold and Tobin 2008; Holmes *et al.* 2008a). International quarantine and phytosanitary policies such as ISPM15 are designed to prevent invasive species from entering the transport network, by mandating thermal treatment, fumigation, or inspection of products in exporting countries (Mumford 2002; Jabara *et al.* 2008). Inspection programs in ports of arrival may detect and eliminate organisms that have survived transport. When an invasive species first becomes established, governments may seek to eradicate it; for example, the U.S. Department of Agriculture APHIS (Agriculture and Plant Health Inspection Service) monitors and treats any areas where Asian gypsy moth and Asian long-horned beetle are detected in the United States. However, because it is often difficult to detect isolated populations until they have become well established, eradication may no longer be possible by the time that public resources are mobilized (Carey 1996; Sharov and Liebhold 1998; Lodge *et al.* 2006; U.S. Government Accountability Office 2006). Once an invasive species starts to spread, domestic quarantine and other barrier-zone programs may be implemented to postpone its impacts in new areas. For example, a U.S. Forest Service program has effectively slowed the spread of European gypsy moth (Sharov *et al.* 2002). At the same time, the U.S. Forest Service operates a cooperative suppression program, providing cost-share and technical support for aerial spraying to suppress gypsy moth populations in states and localities where there are already widely established populations.

Public initiative to manage invasive species is essential because forest health protection is a “public good,” characterized at least to some degree by nonexcludability (everyone benefits from prevention, eradication, or containment

of an invasive species) and nonrivalry (the benefits to one person are not reduced by benefits to another) (Sumner 2003). As previously noted, biological invasions are externalities of trade and transport of people and goods—that is, they are side effects of economic activities that are expected to increase welfare (Perrings *et al.* 2002). For example, both import-export firms and the consumers in the importing country derive benefits from trade in nonindigenous ornamental plants and commodities transported in wood packing. However, each shipment is also associated with a small risk of biological invasion. Economic theory suggests that if the exporters and importers had to pay for those risks, they would effectively “internalize” the externalities, resulting in an optimal level of risk, given the costs and benefits involved. For example, economists have explored the potential for tariffs and fees in invasive-species policy (Costello and McAusland 2003; McAusland and Costello 2004; Knowler and Barbier 2005). Optimally, these would reflect the risks and potential costs of invasive species arriving via various trade pathways, defined by commodity, country of origin, packaging, mode of transport, and phytosanitary measures, and would likely raise the price and lower the quantity of traded goods. Of course, this is easier to conceptualize than to implement, because information on these risks and costs is lacking.

One clear example of the balance between economic causes and consequences of pest invasions involves plantations of nonindigenous tree species. Such plantations represent the majority of forest land area in many countries and indeed the world (Zobel *et al.* 1987). An important reason why plantations of nonindigenous tree species exhibit exceptional productivity is the absence of pests and diseases from their native range (Nair 2001; Wingfield 2003). Maintaining this exclusion of nonindigenous tree species from their pests necessitates strict phytosanitary procedures. Unfortunately, quarantine activities in many countries have not been sufficient to prevent pest invasions and many pests of nonindigenous tree species have

managed to “catch-up” with their hosts (Withers 2001; Wingfield 2003). An excellent example of this is provided by *Eucalyptus* spp., which are native to Australia but widely planted in South America, Africa, Europe, and Asia. Several major pest species from Australia (e.g., *Thaumastocoris peregrinus*, *Gonoptherous* spp., *Phorecantha* spp., *Leptocybe invasa*, and *Colletogloeopsis zuluense*) have invaded all major regions in the world where eucalyptus is grown. The problem may be traced back to the operational difficulty of charging the cost of invasions (reduced timber yield and reduced provision of non-market ecosystem services) into the economic sector that causes invasions (e.g., shipping industries). These pest species cause substantial reduction in volume yield, and ultimately the economic return from *Eucalyptus* plantations is less than had these pests been successfully excluded.

Several economic, legal, and political factors converge to make effective management quite challenging. First, invasions can progress through the “weakest link,” for example, the port with the least effective surveillance, the homeowner who fails to notice diseased trees, or the fuelwood vendor who ignores a domestic quarantine (Perrings *et al.* 2002; Holmes *et al.* 2008a). For example, the deliberate importation and accidental release of European gypsy moth in just one site eventually led to its establishment as one of the most costly invasive defoliators affecting U.S. forests. Second, measures to prevent the arrival of potential invasive species are constrained by international trade policy, which allows inspections and quarantine but not tariffs differentiated by invasion risk. Third, policy makers are handicapped by limited information about which species will become invasive and about the size of newly established and spreading populations (Moffit and Osteen 2006; D’Evelyn *et al.* 2008). This may discourage investment in measures (with large known costs) to reduce arrival rates (with uncertain benefits that are occasionally very large but usually negligible) (Horan *et al.* 2002; Finnoff *et al.* 2007). It is self-evident that the

costs of preventing the arrival or of eradicating the first established populations of particular invasive species would have been much lower than the costs of attempting eradication after establishment and spread. However, as Perrings (2007) notes, this self-evident truth “is like calculating the benefit-cost ratio of the purchase of a winning lottery ticket. The *ex ante* calculation involves uncertainty about the invasiveness of the species, the invasibility of the system being protected, the effectiveness of the control program, and the responses of those whose life and livelihoods are affected by invasive species and their control” (p. 150). Finally, many invasive forest pests and pathogens affect nonmarket values, such as some ecosystem services, landscape aesthetics, and biodiversity. These generally attract less funding and a less coordinated response than invasions that directly threaten agriculture or human health (Lodge *et al.* 2006).

Economic Consequences of Biological Invasions in Forests

Since the time Europeans discovered North America, more than 368 nonindigenous phytophagous insects have become established in forests, woodlots, parks, and orchards (Mattson *et al.* 1994). A characteristic of biological invasions is that most nonindigenous pests are innocuous, and only a few become serious pests (Williamson and Fitter 1996). From the ensemble of nonindigenous forest insects that have become established in North America, a limited number—such as the European gypsy moth, emerald ash borer, and hemlock woolly adelgid—are causing severe economic losses. The number of nonindigenous tree pathogens that have become established in North America is more difficult to gauge, as many pathogens are either innocuous or insidious and, in general, no effort is expended to document invasions by microorganisms unless they have measurable impacts. Nonetheless, a similar pattern emerges for tree pathogens that have become

major pests; only a few tree diseases—such as chestnut blight, Dutch elm disease, and sudden oak death—have had substantial economic consequences. Therefore, if the past incidence of economic damages induced by forest pests is viewed as a random drawing from some underlying distribution of economic damage potential, then accounting for the economic losses caused by a few catastrophic nonindigenous forest pests might provide a reasonable lower-bound approximation to the level of aggregate economic damages that might be expected in the future.

Although it appears to be both pragmatic and reasonable to focus attention on the economic impacts of the most damaging nonindigenous forest pests, accounting for the economic impacts of the top few forest-invasive species is not a simple matter. We stress that estimates of economic damages from nonindigenous forest pests need to be consistent with that component of microeconomic theory known as welfare economics, which emphasizes the impact of changes in prices, quantities, and environmental quality on the economic well-being of consumers and producers. Although a number of studies have been conducted in the context of microeconomic theory to evaluate the economic impacts of nonindigenous forest species, the question remains as to how these few studies might be scaled-up to provide estimates of aggregate economic damage.

Nonindigenous forest invaders cause damages to a suite of forest ecosystem goods and services. While impacts on timber values and timber markets are substantial, we suspect that damages to nonmarket economic values are at least as important. A preliminary estimate of aggregate economic damages would keep separate accounts for the various categories of damages that are incurred, keeping in mind that dollar values cannot always be summed if the underlying theoretical constructs are disparate. For example, reductions in revenues cannot be summed with reductions in profits because the economic measures differ, even though both impacts are measured

in dollars. Likewise, reductions in property values should not be summed with costs of hazard tree removals on private property because this might lead to double-counting of impacts.

Changes in Economic Well-Being

Within the domain of neoclassical economics, changes in the economic well-being of producers and consumers in response to changes in prices, quantities, or environmental quality are topics in welfare economics (e.g., Varian 1984). Microeconomic analysis describes the behavior of producers and consumers at the individual level and in terms of aggregates such as markets. A fundamental principle of microeconomic theory is that economic agents attempt to do the best they can, as they define it, given constraints on their access to resources. When producers or consumers are faced with a change, such as damage to forest assets from a nonindigenous pest, they will generally seek means for minimizing their losses. The tendency for people to search for creative ways to mitigate losses creates nuances in economic analysis that need to be recognized when evaluating the economic impacts of nonindigenous forest pests. Although these nuances typically demand greater rigor in estimating economic impacts, we argue that the theoretical foundation underlying such approaches makes them most suitable for policy analysis.

A particularly vexing issue in the economic analysis of nonindigenous forest species concerns the distinction between the value of capital accounts and changes in economic welfare. Microeconomic theory, which provides the foundation for economic welfare analysis, is primarily concerned with understanding trade-offs among economic variables and how decisionmakers respond to changes in economic and environmental conditions (this forms the basis for marginal analysis). From an economic welfare perspective, economic values for Nature are determined by the trade-offs made by

individual economic agents, such as how much one would be willing to sacrifice to preserve a natural resource in its current condition relative to some altered condition. Although previous studies have sought to estimate the economic value of nature without reference to alternative states of nature (Costanza *et al.* 1997), this approach has been criticized for failing to consider economic trade-offs as the basis for economic valuation (Bockstael *et al.* 2000). Among the nuances of individual economic behavior that enrich, and complicate, economic analysis of forest damages is the tendency for people to seek substitutes for resources that have been or will imminently be damaged. For timber, this could involve the substitution of alternative species by timber growers or by wood-using firms, or the substitution of salvaged timber for green timber in the marketplace. Likewise, in a non-market context, residential landowners could substitute alternative species for species damaged by forest pests. Substitution of one good or service for another adds an important dynamic to economic analysis that needs to be recognized when evaluating the economic value of ecosystems.

To illustrate this point, consider the following juxtaposition between the capital value of standing chestnut timber and the suggested economic behavior for mitigating losses. Based on reports from governmental agencies in Pennsylvania, West Virginia, and North Carolina, the value of standing chestnut timber at risk of mortality from chestnut blight in those states was estimated to be about \$82.5 million in 1912 (Anagnostakis 1987). In current (2007) dollars, this amount represents roughly \$1.7 billion in standing timber value. Although this is a substantial sum of money, it may very well overstate the market value of timber lost to the disease in those states if forest owners followed the advice offered by the assistant director of the New York Botanical Gardens: "Utilization is the big issue. See that you are advised of the progress of the disease, appropriating money for this if necessary, and market your timber as it approaches. Be businesslike and accept the

inevitable in time to make the best of it” (Dr. W.A. Murrill, *New York Times*, 1912). Businesslike behavior, such as preemptive logging of chestnut stocks, mitigates economic losses by liquidating some of the accounting value before trees are killed (Keever 1953). Further, state foresters in the affected areas advised landowners who were not able to harvest live trees to salvage the value they could from trees killed by the blight (Anagnostakis 1987). This action would further mitigate the loss of accounting value. Economic behavior drives a wedge between accounting values and the change in economic welfare.

In North America, two countrywide studies have been conducted that provide estimates of the aggregate economic impacts of forest-invasive species. Both studies conflate accounting value and economic loss. The most widely cited study providing estimates of aggregate economic damages due to nonindigenous forest pests is Pimentel *et al.* (2000), who suggest that economic damages arising from nonindigenous forest pests in the United States are on the order of \$4.2 billion annually. This estimate is based on two critical assumptions: (1) forest pests reduce overall timber productivity by 9% per year, and (2) nonindigenous forest insects and diseases account for 30% of the damage caused by all forest pests. We suspect that the estimate of aggregate damage is biased for several reasons. First, the authors used accounting methods that do not account for economic behavior and are not consistent with microeconomic theory. Second, impacts are measured in final product markets, not markets for timber inputs. Third, the authors do not account for nonmarket economic values.

Colautti *et al.* (2006) evaluated economic losses due to nonindigenous forest insect and diseases in Canadian forests. They employed the price-times-quantity method by assembling a list of 16 nonindigenous invasive species affecting Canadian ecosystems, and obtaining estimates of the proportional loss of resource production associated with each pest. As noted by the authors, the productivity loss estimates

used in their empirical model are highly biased, as they were based on small-scale, short-term studies. They attempted to offset this bias by ranking, from smallest to largest, the loss proportions associated with each pest. By treating the median, quartile, and half-quartile as estimates of the maximum, midrange, and minimum levels of productivity loss, they obtained alternative, but not necessarily unbiased, estimates for reduced yields. Applying this technique to forest ecosystems, the authors estimate the loss in value to forest products resulting from four nonindigenous insects (Asian longhorn beetle, balsam woolly adelgid, brown spruce longhorn beetle, and gypsy moth) and three nonindigenous pathogens (Dutch elm disease, scleroderris canker, and white pine blister rust) to be \$9.6 billion (ranging from \$7.7 billion to \$20 billion). Because their estimates do not account for “businesslike behavior” taken to mitigate potential losses, because values are measured in final product markets, and because they did not quantify nonmarket economic impacts, we conclude that their estimated impacts are susceptible to multiple sources of bias.

Turner *et al.* (2004) estimated the costs and benefits of biosecurity and forest health research to the forest growing industry and urban forest estate in New Zealand. Because the timber supply and demand relationships needed to estimate economic welfare impacts have not been estimated for the New Zealand forest sector, they performed a linear aggregation based upon estimates of yield decline and average stumpage prices. Although this method for estimating timber-sector impacts is more defensible than applying values from final product markets, it does not represent a change in either timber producer welfare (the area above the supply curve and below price) or wood-products consumer welfare (the area below timber demand and above price). Thus, it is not consistent with microeconomic theory. The authors also included accounting measures for costs associated with eradication and control programs, household expenditures to control forest pests, and expenditures to replace trees in

the urban forest. Projecting impacts to the year 2070, they concluded that the net present value of economic impacts associated with a new forest pest range from \$3.8 billion to \$20.3 billion, depending on the discount rate used.

In contrast to the accounting studies reported by Pimentel *et al.* (2000), Turner *et al.* (2004), and Colautti *et al.* (2006), Holmes (1991) demonstrated how forest-sector market models can be used to measure the changes in economic welfare due to catastrophic pest outbreaks using timber supply-and-demand curves. This approach is consistent with microeconomic theory and captures the behavioral responses made by timber producers and wood-using consumers as economic conditions adjust to changes in forest health. The changes in economic welfare measured with the forest-sector market approach are consistent with Just and Hueth (1979), who show how the changes in profits within a vertically integrated industry (such as the forest-products sector) can be measured using supply-and-demand curves for raw inputs. The forest-sector market-model approach to measuring changes in economic welfare was also used by the USDA Forest Service (Holmes and Lee 1991) to evaluate changes in the economic welfare of timber producers and wood-products consumers if non-indigenous pests were introduced from timber trade with the Soviet Far East. Potential net changes in producer and consumer welfare over the period 1990 to 2040 due to insect defoliators were estimated to range from \$35 billion to \$58 billion using a 4% discount rate.

Nonmarket Economic Losses

A potentially enormous source of bias in aggregate estimates of the economic losses due to nonindigenous forest pests is the failure to account for nonmarket values. These values encompass a suite of ecosystem services provided by trees, groves, stands, and forests (such as water filtration, flood mitigation, wildlife habitat, carbon sequestration, and landscape aesthetics)

that are not directly traded in markets. Hundreds of nonmarket valuation studies seeking to quantify virtually all dimensions of forest goods and services have been conducted over the past several decades (Sills and Abt 2003). However, only a limited number of these have focused attention specifically on the economic impacts of forest insects and diseases. The results to date suggest that the residential impacts of forest pests are large and might dominate other categories of market and nonmarket impacts in developed countries (Leuschner *et al.* 1996; Reinhardt *et al.* 2003; Turner *et al.* 2004; Nowak *et al.* 2006).

Forest landscapes are highly valued when there is a confluence of an advantageous spatial location and the presence of biological attributes that satisfy human needs. Forests in favorable spatial locations that provide desirable biological attributes are the landscapes most likely to be modified for human needs. The relative scarcity of favored private forest locations, relative to more abundant, remote and/or less biologically desirable forest areas, conveys economic value. Consequently, we would expect that forested residential landscapes, in general, are more highly valued—per unit area—than the landscapes that are traditionally categorized as forest (e.g., private and public timberlands and nonindustrial private forest lands). Therefore, invasion of these modified forest landscapes is likely to impose the greatest losses in nonmarket values. At the same time, these modified landscapes are historically more likely to be invaded, possibly precisely because they have been modified and therefore have lower biotic resistance, and more certainly because of proximity to transportation hubs and the resultant propagule pressure.

The nonmarket costs of invasive species have been studied in an economic welfare-theoretic context using both direct questions about willingness to pay (stated preference methods) and evidence from behavior in related markets (revealed preference methods). The most commonly used approach is the contingent valuation method (CVM). In this survey method,

respondents are asked their willingness to pay for a change in conditions in a constructed (hypothetical) market. CVM has been used to understand the full cost of invasive species—including aesthetics and other unpriced ecosystem services—in residential areas as well as public lands. For example, several CVM surveys have elicited willingness to pay for better control of gypsy moth in residential landscapes (Jakus 1992; Miller and Lindsay 1993; MacDonald *et al.* 1997). Leuschner *et al.* (1996) relied on this literature (specifically University of Maryland, 1988) for their conclusion that residential costs were much higher than any other component of costs.

The contribution of landscape attributes to private-property values can be studied using an economic welfare-theoretic method known as the hedonic property-value method. This method has been used to estimate the value that trees contribute to the sale values of homes from three perspectives: (1) yard trees contribute to property values, (2) forest preserves near residential neighborhoods convey value, and (3) trees in the general forest matrix surrounding residential areas convey value. These studies indicate that trees contribute, roughly, from 2% to 5% to the private-property value of private residences (Morales 1980; Anderson and Cordell 1988; Garrod and Willis 1992; Dombrow *et al.* 2000; Tyrvaainen and Mietinen 2000). Consequently, we would expect that nonindigenous forest pests that cause a visible loss in forest health (Sheppard and Picard 2006), or that ultimately cause tree mortality, would induce a loss of property values in residential areas.

Although evidence of the impact of non-indigenous forest pests on residential property values is limited, the study by Holmes *et al.* (2006)—also reported in Huggett *et al.* (2008)—indicates that private-property value losses due to infestations of the hemlock woolly adelgid (*Adelges tsugae*) in residential areas are large and that spillovers from properties with damaged trees to neighboring properties are evident. Values reported in those studies indicate that

losses to residential property values can be as much as \$20,000 per acre of dead and dying hemlock. Impacts of this magnitude are much larger than comparable losses in timber value on well-stocked, old-growth eastern hemlock stands.

In addition to the economic losses borne by private-property owners in residential areas, forest pests can cause substantial nonmarket economic losses on public forest lands (Walsh *et al.* 1990; Haefele *et al.* 1991). Household willingness to pay to avoid nonmarket economic losses, as measured by CVM studies, is additive across the relevant population of consumers because forest protection is a public good (Holmes *et al.* 2008a). Before aggregating nonmarket values, however, two questions must be answered. First, who cares about the change in nonmarket goods and services (what is the extent of the market) (Smith 1993)? Second, are substitute goods and services available that would cause aggregation to be nonlinear (Bockstael *et al.* 2000)? Given these considerations, even conservative estimates of aggregate willingness to pay can be large and should not be ignored. For example, Moore (2008) used a CVM study of households in North Carolina to evaluate willingness to pay by residents of that state for a 3-year program to control the hemlock woolly adelgid in the Great Smoky Mountain National Park and neighboring U.S. Forest Service lands. He found that household willingness to pay is about \$50/year. Summing this value across all households in North Carolina results in aggregate value of protection programs that exceed \$100 million per year for the 3-year program. This value could be considered a lower-bound estimate of the total value of hemlock protection programs because it is likely that households in other states that either use or are familiar with these public lands would also have a positive willingness to pay. The values reported by Moore (2008) are consistent with other CVM studies of forest protection programs reviewed by Kramer *et al.* (2003), who concluded that the total non-market economic value derived from protecting

the health of public forests is largely due to the knowledge that healthy forest ecosystems exist (existence value) and will be maintained for future generations (bequest value).

Scaling-up Microeconomic Values

Although a number of theoretically consistent microeconomic models of nonindigenous forest-pest impacts have been reported in the literature, and should form the foundation for economic analysis of nonindigenous forest pests, a major difficulty associated with the use of microeconomic studies to estimate aggregate economic impacts is the problem of extrapolation. This is true whether we are trying to estimate the aggregate economic impacts due to a single nonindigenous forest pest or whether aggregate estimates are sought for the entire suite of nonindigenous forest pests. The aggregation problem faces two major challenges. First, existing microeconomic studies reported in the literature are a sample of some underlying population. Extrapolation of estimates based on microstudies must then have a clear idea of the nature of the population from which the sample is drawn, and the spatial extent of the impacts should be circumscribed. Second, the scaling from microeconomic to aggregate economic impacts may not be linear, and interactions, either in the marketplace or for nonmarket values, may be critical.

The aggregation of microeconomic studies across relevant geographic areas will be simplified by focusing attention on the transient ecological and economic impacts that occur while these systems adjust to a biological invasion. The idea that economic variables operating on slow-time-scales can be treated as constants, allowing one to focus on fast-time-scale economic dynamics, was presented in the seminal paper by Simon and Ando (1961). They recognized that aggregation in dynamic systems can be greatly simplified by decomposing all of the interactions in the economy into a small number of sectors, within which interac-

tions can be studied (on the fast-time-scale) as if the slow interactions between sectors did not exist. For forest protection, this approach suggests that separate accounts can be established for sectors such as timber markets, residential landscapes, and public forests. Within each sector, the fast-time-scale dynamics can be studied and the interactions (such as substitution patterns) can be identified. Further, by limiting the time period over which intrasectoral dynamics and interactions are studied to, say, 10 years, the longer-term intra- and intersectoral interactions can be held constant.

Once the economic system has been decomposed into the relevant sectors for analyzing the economic impacts of nonindigenous forest pests, aggregation of the fast-time-scale variables will necessitate the development of models for each sector. The goal of the models is to link the available microeconomic studies to the population from which they were drawn. In the case of nonindigenous forest species, aggregation will generally be across geographic areas representing the distribution of host species and across the population of people who are affected by the specific changes in forest health.

In the case of timber damages from a nonindigenous forest pest, extrapolating from a single producer to the level of regional damages could proceed using forest-sector market models, as reviewed above. Scaling-up of nonmarket economic values could be accomplished using benefit-transfer models (Rosenburg and Loomis 2003). The basic idea of a benefit transfer is to extrapolate the results of a case study to other sites of policy interest. Perhaps the best known method of benefit transfer is meta-analysis, in which statistical methods are used to relate economic values reported in a set of case studies to a set of explanatory variables that vary across the case studies (Smith and Huang 1995). Statistically significant explanatory variables that can be linked to those same variables at other geographic locations can then be used to transfer values from the case studies to other geographic areas with similar characteristics.

We expect that too few microeconomic studies have been reported to conduct explicit meta-analysis of economic damages from forest-invasive species. In this case, we suggest that spatial analysis, using Geographic Information System (GIS) tools, might be productively used to extrapolate economic damage estimates obtained from microeconomic studies to the mesoscale (Eade and Moran 1996; Bateman *et al.* 2002; Holmes and Smith 2007). In particular, the population from which documented case studies have been sampled might be identified in a two-step process: (1) list a set of salient characteristics that are associated with case-study sites, and (2) use GIS tools to identify locations with similar characteristics at the landscape scale. By linking economic values obtained at the study site with the set of locations identified at the landscape scale, values can be transferred from study sites to other sites of policy interest.

We anticipate that, having extrapolated economic damage estimates from case studies to areas of policy relevance, a distribution of economic impacts will be observed, and that the original case studies might well be among the set of worst-case damages. This is because economic case studies often focus on bad- or worst-case scenarios in order to detect significant economic impacts due to the availability of data and the power of statistical tests. A relatively simple, and conservative, approach to aggregation within the specified economic sectors would be to sum the damage estimates at the bad- and worst-case geographic locations across the landscape.

Finally, damages estimated within specified economic sectors at the landscape scale could be summed across the ensemble of most-damaging forest pests to arrive at a conservative estimate of aggregate economic impact. We warn that economic sectors should be kept separate and not aggregated if the underlying economic logic used to describe economic impacts differs across sectors. Nonetheless, this procedure could provide estimates of the economic losses incurred within sectors and iden-

tify the pests that are most damaging within the sectors.

We speculate that this approach to obtaining conservative, lower-bound estimates within prescribed economic sectors would be eminently reasonable if economic damages are distributed following a heavy-tailed distribution. This speculation is based on a particularly intriguing result reported in previous studies showing that the size distribution of some well-known forest disturbances such as wildfires (Malamud *et al.* 2005; Holmes *et al.* 2008b) and bark-beetle epidemics (Gamarra and He 2008) follow power laws. Power laws are unusual in that they are self-similar across scales, and might represent the behavior of complex adaptive systems (Chave and Levin 2003). This property gives power-law distributions heavy tails and suggests that most of the damage from forest disturbances occurs in a few, rare events. In general, power laws have the property that the magnitude of the sum of addends is equal in magnitude to the largest of many addends (Mandelbrot 1997). If damages from nonindigenous forest pests follow power-law distributions, then the problem of scaling-up from case studies to aggregate estimates may be simplified by focusing attention on the few cases where the largest impacts are anticipated—because it is these cases that would dominate the sum of the economic impacts.

Finally, to continue our speculation, if forest damages follow heavy-tail distributions, then the past may not provide a reliable guide for the future. Imagine for a moment an economic team given the mandate to estimate the potential economic losses from nonindigenous forest pests in 1904—the year before chestnut blight was first identified in the United States on chestnut trees in the New York Zoological Garden (Anagnostakis 1987), and 2 years before white pine blister rust was discovered on pine seedlings imported from Europe (Maloy 1997). Although gypsy moth was present in eastern U.S. forests at that time, there was considerable optimism that it could be controlled (Popham and Hall 1958). If the analysts had

summed up all of the economic damages to that point caused by nonindigenous forest pests and divided by the number of years of damage to arrive at an estimate of the average annual impact, they would not have come close to predicting the catastrophic levels of forest damage observed during the subsequent decades. Likewise, it may be presumptuous for economic analysts today to assume that past damages from nonindigenous forest pests represent the worst kind of pest behavior by portraying average damages from historical events as typical.

Concluding Remarks

The economics of forest-pest invasions are particularly complex because (1) invasions have both important economic causes and consequences; (2) once established, the ecological and economic impacts of forest-invasive species may be irreversible; and (3) knowledge is limited regarding which nonnative forest pests will ultimately cause catastrophic damage. Government policies that address the pest-invasion problem must balance the economic factors causing invasions (e.g., trade) with the economic impacts of invasions. At present, the economic costs and losses associated with forest-pest invasions are borne by taxpayers and private forest landowners. In theory, policies that shift the costs and losses induced by invasive pests onto the responsible economic sectors (e.g., by imposing tariffs) should result in an optimal reduction in risk, given the balance of costs and benefits. Unfortunately, substantial political and sociological barriers prevent the internalization of these spillover effects and other approaches may be necessary. The Precautionary Principle, which emerged in the 1992 Conference of Rio on Environmental Development, provides a rationale for applying cost-benefit analysis and discretionary judgment in policy development where scientific knowledge is lacking regarding the impacts of economic activities on the environment. Although the Precautionary Principle remains controversial, its

application may be justified in cases where it is supported by the weight of scientific evidence (Foster *et al.* 2000).

Whether for the development of international trade policy or on-the-ground for forest health protection programs, reliable estimates of the economic impacts from invasive forest pests are needed to inform policy making at all levels of government. Due to a paucity of data and underlying microeconomic analysis that could provide a basis for estimating macrolevel parameters, few attempts have been made to estimate countrywide economic damages from nonindigenous forest pests. Although existing estimates are useful in providing a rough idea of the magnitude of economic threats, new approaches are needed.

In this chapter, we argue that economic analysis needs to be consistent with economic theory. People respond to factors that threaten their economic well-being, and economic systems adjust to changes in ecosystem dynamics. In some instances, people take actions to mitigate damages, whether that means applying preventive treatments to protect trees, preemptive harvesting of timber, salvaging dead timber, removing hazard trees, or substituting alternative species in the production process or on the landscape. These types of actions often involve transfers between various market participants. In other cases, forest damages cannot be prevented and losses to aesthetic values, recreational opportunities, or wildlife habitat are incurred.

Previous countrywide analyses of the economic impacts of forest-invasive species have focused primarily on productivity losses in the wood-products sector of the economy. These studies are biased because of the methods used to estimate impacts to the forest-products sectors and because impacts on the suite of non-timber ecosystem services are not adequately addressed.

Our review of existing microeconomic studies suggests that total nonmarket economic impacts from invasives might exceed the value of timber losses. Nonindigenous forest species

are typically introduced via ports-of-entry into areas of high population density, and highly populated areas are often located in or near forested landscapes that are used for residential and recreational uses. Thus, we hypothesize that large nonmarket economic impacts can occur in forested landscapes that are modified for human uses or are located close to population centers. Nonmarket economic impacts on the aesthetic value of trees on residential properties, and on the value of protecting forest ecosystems on public land need to be included in damage assessments, to the extent possible.

Aggregate estimates of economic impacts from nonindigenous forest pests should use existing microeconomic studies as a foundation. The aggregation problem can be greatly simplified by focusing on fast-time-scale variables and treating slow-time-scale variables as constant. Independent economic sectors should be identified, and the transitory economic dynamics within those sectors, including substitution patterns and other interactions, should be studied. This can be accomplished by constructing models for each of the sectors identified. Forest-sector market models can be used to estimate changes in economic welfare for timber producers and wood-products consumers. Benefit-transfer models can be used to aggregate nonmarket values across geographic areas and the population of people who are impacted by changes in ecosystem dynamics.

Aggregation of microeconomic studies to the landscape scale can be further simplified by focusing attention on the most damaging nonindigenous forest pests. Microeconomic studies are typically conducted where impacts are large enough to be detected by economic methods. If the damages from nonindigenous forest pests are concentrated in the sense that most damages occur in a few instances, then the extrapolation from case studies to the most damaging replicates on the landscape may constitute a reasonable lower bound to estimates of aggregate damage from specific pests.

This idea may be equally applicable to the estimation of countrywide economic damages

from forest-invasive species. If most invasive species are innocuous, and a few species are highly consequential, then the magnitude of aggregate economic damages from all nonindigenous forest pests is concentrated and might reasonably be estimated by the magnitude of damages associated with the worst-case examples of individual nonindigenous forest pests. Linking theoretically sound market and nonmarket economic damage estimates with conservative procedures for aggregation across scales would provide a sound foundation for the development of economic policies and management strategies.

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Conflicts of Interest

The authors declare no conflicts of interest.

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