CHAPTER 19

ECONOMIC ASPECTS OF INVASIVE FOREST PEST MANAGEMENT

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1. INTRODUCTION

The past decade has evidenced growing concern with the causes and consequences of biological invasions, many of which are economic in nature (Perrings et al. 2002). The risk of a new pest introduction is positively correlated with world trade flows (Costello and McAusland 2003, Margolis et al. 2005) and new invasions threaten the productivity and biological diversity of native ecosystems (Mack et al. 2000). A recent study reports that roughly 50,000 exotic species are established in the United States and annual domestic costs and annual losses from invasive species (forest and non-forest) may exceed $120 billion (Pimentel et al. 2005). The passage of Executive Order 13112 (Clinton 1999), which enhances federal coordination and response to invasive species, and the creation of the National Invasive Species Council (NISC 2001, NISC 2005), are evidence of the federal government's substantial concerns with these emerging threats to terrestrial and aquatic ecosystems.

Forests provide suitable habitat for an assortment of invading organisms (Liebhold et al. 1995) and invasive species have been ranked as one of the four critical threats to our Nation's forest ecosystems by the Chief of the U.S. Forest Service (USDA Forest Service 2004). Although most people might argue that it is laudable to counter threats to the structure and functioning of forest ecosystems, relatively few exotic organisms become a major pest (Williamson 1996). It is the main thesis of this chapter that decisions regarding budget allocations and the targeting of forest protection efforts would benefit from a clear understanding of the costs and benefits of invasive forest pest management. Interventions designed to mitigate damages from exotic forest pests are costly—the Forest Service spent $95.1 million dollars for the management of invasive forest pests in fiscal year 2005 (USDA Forest Service 2005, p. 14-55). However, very little is known about the magnitude of economic damages caused by exotic forest pests, or the efficacy of the money spent on pest control. This lack of knowledge impedes economic analyses of pest management programs and it

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remains unclear whether current expenditures on invasive forest pests are too little, too large, or about right.

The goal of this chapter is to provide an overview of some salient economic aspects of invasive forest pest management. Our synopsis begins with a broad economic characterization of the invasive species management problem. Following this, we provide a brief review of management strategies that have been applied to combat select invasive forest species in the United States. We then turn to a case study of a recent threat to forests in the eastern United States, the hemlock woolly adelgid, and emphasize the importance of public awareness and private action to strengthen links in forest health protection.

2. ECONOMIC ANALYSIS OF INVASIVE SPECIES

To begin, a generalized biological invasion can be described by a sequence of ecological states (fig. 19.1). Economic analysis is relevant to the design of invasive species management programs because each ecological state is associated with one or more management actions and a vector of economic costs and losses. An economic approach to invasive species management seeks to minimize the sum of management costs plus losses to trade, domestic market production, and non-market economic values.

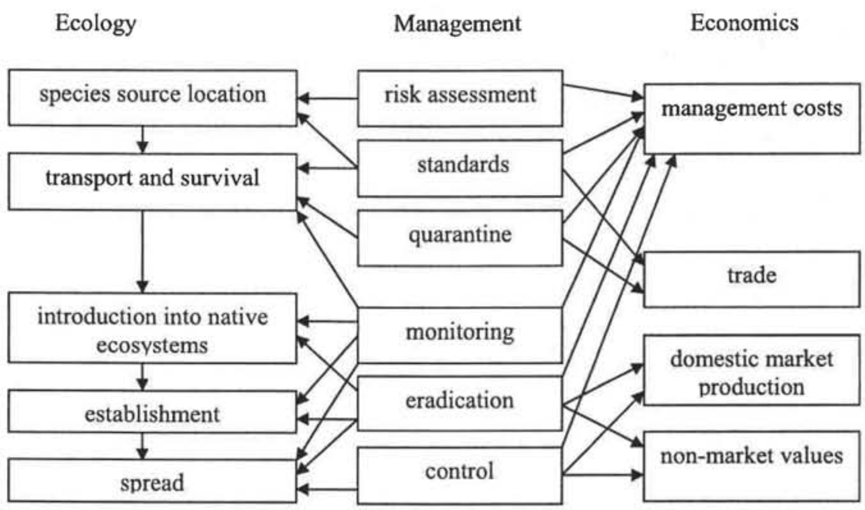

Figure 19.1. A general, conceptual model of the relationships between the ecological stages of a biological invasion, management responses, and economic impacts. Social welfare is optimized by minimizing expected management costs plus the expected losses to trade, domestic market production, and non-market values.

In this section, we describe two prominent issues in the design of optimal economic programs for invasive species protection. First, forest protection programs provide services that are public goods, and private landowners cannot be expected to provide the socially efficient level of forest protection. Thus, government has an essential role in the provision of forest health. Second, the time lag between the investment of capital and labor targeted at forest protection and the observation of a change in physical or economic damages to forest ecosystems introduces substantial uncertainty into both public and private decision-making. Although economic frameworks have been developed to improve decision-making under conditions of uncertainty, major challenges remain in the implementation of optimal economic programs. In such an environment, Bayesian methods provide a promising approach to adaptive management given uncertainty regarding models, parameters, and data (see section 2.2.3).

2.1 Weakest-Link Public Goods

Any evaluation of the optimal level of investment in forest health protection, either from the perspective of private or public forest owners, needs to recognize that forest health protection is a public good. The benefits of a quarantine, for example, are non-excludable (if quarantine benefits are made available to one person, they are available to every member of the community), and non-rival (the benefits received by any individual do not decrease the level of benefits available to others). As is well known, the standard theory of public goods argues the private provision of public goods is sub-optimal if self-interested individuals equate the marginal cost of their investment in public good provision with their marginal private benefits, but do not account for the benefits received by other members of society (Samuelson 1955).[1] Because the level of private provision is socially sub-optimal, governments have a key role to play in providing the socially optimal level of forest health.

Government-sponsored provision of forest health protection proceeds using an assortment of strategies (section 3), some of which necessitate the involvement of private landowners and households. In the standard public goods model, the socially available amount of a public good such as forest health protection (H) is the sum of the amounts (h_i) produced by community members (i): $H = \Sigma h_i$. As highlighted by Hirshleifer (1983), other social composition functions are possible. Of particular importance to the design of invasive species management programs is the concept of a weakest-link public good (Shogren 2000, Perrings et al. 2002). The weakest-link aspect of biological invasions arises from the condition that each member of a social group (say, forest landowners) has a "kind of veto power

[1] A game theoretic formulation of the provision of public goods by self-interested individuals is a Prisoner's Dilemma where the dominant strategy, not providing the public good, is sub-optimal to both players providing the public good (Harrison and Hirshleifer 1989).

over the extent of collective achievement" (Hirshleifer 1983, p. 373). Just as the strength of a chain depends upon its weakest link, or the protection provided by a system of levees depends upon its lowest height, the aggregate provision of forest health protection is compromised by forest landowners within a community who fail to take actions to protect their land from invasive forest pests, thereby increasing the risk for other forest landowners in the community. Hirshleifer (1983) argues that the weakest-link social composition function is given by the level of protection provided by the weakest member. Thus, in the case of forest health protection, the weakest-link social composition function is $H = \min(h_i)$.

When the weakest-link character of the social provision function is understood by each member of the community, the value of $\min(h_i)$ can be raised, perhaps dramatically. Anecdotal evidence supports the proposition that improvements to the weakest-link occur when a social threat is perceived to be overwhelming, as is sometimes evidenced in the aftermath of a natural disaster (Hirshleifer 1983). Further, experimental evidence has demonstrated that, when people understand that the social provision of public goods is of the weakest-link variety, the propensity of individuals to free-ride is greatly reduced (Harrison and Hirshleifer 1989). This result can be understood by examining table 19.1, which shows the payoffs to self-interested individuals from either protecting or not protecting their forest land. Letting b represent the forest protection benefit received by each player if both players protect their forest land, and letting c represent the cost of forest protection, the weakest-link model applies if $b > c$. It can be seen that if either player does not protect their forest, then no benefits are forthcoming to either player and the net economic payoff is either $-c$ or zero. However, if either player invests c in forest protection, then the best strategy for the other player is to likewise invest c, and the net economic payoff to each player is $b-c$. This formulation of the forest protection problem provides a rationale for the development of public programs that raise the awareness among stakeholders that the benefits of forest health protection critically depend upon the contributions made by each member of the community.[2]

Table 19.1. Two-player, private forest landowner payoff table illustrating a weakest-link social composition function.

	Protect forest 2	Not protect forest 2
Protect forest 1	$b-c_1$, $b-c_2$	$-c_1$, 0
Not protect forest 1	0, $-c_2$	0, 0

[2] A weakest-link interpretation might be applied to the best known slogan in forest protection: "Remember—only you can prevent forest fires!" Assuming that the avoidance of careless behavior entails a cost, Smokey Bear can be thought of as reminding the public that the benefits of forest protection are conditional upon the (costly) actions taken by each member of the community.

The weakest-link concept can be applied to specific stages of invasive species management such as early detection and citizen response. A good example is provided by the hemlock woolly adelgid (*Adelges tsugae*, or HWA), an exotic insect that is responsible for widespread mortality of hemlocks throughout the eastern United States from Georgia to Maine. To contain the spread of HWA in Maine, the state has mounted a public awareness campaign (see section 4). Early detection and removal of HWA infected trees can reduce the risk for other landowners, and this strategy has been pursued by informing landowners of the visible symptoms of HWA infestation and what to do in case a suspected infestation is identified, and by providing maps of areas of known infections. A second example is provided by the emerald ash borer (*Agrilus planipennis*), an exotic insect that is responsible for the death of millions of ash trees, primarily in Michigan. A major emphasis of the current control program is to contain this pest in the Lower Peninsula of Michigan and eradicate it from Ohio and Indiana. A major focus of this program is to change the behavior of the weakest-link—residents who move firewood from infested areas to summer homes or campsites in uninfested parts of the states. Although the cost to individuals or households of changing their behavior may appear to be relatively small, the forest protection benefits will accrue only if everyone subscribes to this program.

2.2 Decision-Making Under Risk and Uncertainty

One of the most challenging obstacles to the development of forest health protection programs, both within public agencies and with broad-based private landowner participation, is the prevalence of risk and uncertainty. Although the risk (which we define as a probability, π) associated with each stage of a biological invasion is rather low[3], the uncertainty (θ) associated with each risk estimate may be quite large.[4] In this section, we provide a broad overview of the ways in which economists have modeled risk and uncertainty, and illustrate these concepts in the context of forest invasive species. We argue that, because the risk and consequences of a biological invasion can be influenced by management actions, and because the characteristics of an invasion might be of a kind not seen before, novel management approaches may be required.

The most general economic approach to decision-making under risk and uncertainty is the state-preference approach (Deaton and Muellbauer 1980,

[3] Williamson (1996) notes that roughly 10% of exotic species arriving in a non-native habitat become introduced into the wild, 10% of introduced species become self-sustaining, and that 10% self-sustaining species become pests. The so-called *10-10-10 rule* indicates the inherent difficulty of predicting which exotic forest organisms will ultimately become invasive forest pests.

[4] We use the term uncertainty to refer to limited knowledge of fixed quantities such as model parameters. As sample sizes increase, for example, uncertainty will decline. This allows learning as data accumulate.

p. 383-386). Three concepts are essential to this framework—states of the world, acts, and consequences. Acts (such as invasive species management programs) must be taken before the state of the world (such as the true ability of a new organism to invade an ecosystem) is known.[5] Consequences result after actions are taken and the true state of the world is revealed. From the perspective of invasive species management programs, consequences represent the sum of program costs and economic losses as well as other ecological or social impacts which cannot be monetized. At the social level, the major categories of economic damages are the loss of trade benefits, losses to agricultural, forest, and range productivity and losses to non-market economic values.

2.2.1 Expected utility

The major state-preference paradigm developed since World War II is expected utility theory (Shoemaker 1982). The conceptual framework can be visualized as a two-dimensional matrix where rows represent management actions, columns represent states of nature, and matrix cells represent economic consequences (table 19.2). The implementation of this framework necessitates two strong assumptions: (1) only consequences matter to the decision-maker—states of nature do not, and (2) the decision-maker cannot influence the probability of various states of nature—they are exogenous to human control (Deaton and Muellbauer 1980, p. 389). Given this framework, a rational decision-maker should choose the action (α) that maximizes expected utility:

$$\underset{\alpha}{Max}\, U = f\left[\pi^1 \upsilon(\gamma_1(\alpha)) + \pi^2 \upsilon(\gamma_2(\alpha)) + \ldots + \pi^n \upsilon(\gamma_n(\alpha))\right] \qquad (19.1)$$

where π^i is the probability of occurrence of state i, υ is a sub-utility function, γ_i is the consequence associated with state i, f is a function that aggregates sub-utilities into total utility, and $\Sigma_i \pi_i = 1$, that is, all states of nature are assigned a probability. The utility function shown in equation (19.1) not only expresses the decision-makers' preferences over various possible outcomes but also includes their assessment of the relative likelihood of the various states of the world that might occur. The decision-maker would choose the action (or, more generally, management program) that maximizes their expected utility.

As suggested by the example illustrated in table 19.2, application of expected utility (EU) theory to the optimal economic design of invasive species management programs is complex and requires a wealth of detailed information including estimates of the probability that each state of nature will occur, a list of feasible

[5] Decisions taken to prevent the occurrence of unwanted states of the world, such as terrorist acts or ill health, are more akin to solving a mystery than a puzzle (Treverton 2007). Preventive actions must be taken before states of the world are revealed and it is often not clear whether non-events are due to prevention efficacy, luck, or some other cause.

Table 19.2. Social payoff table showing hypothetical economic costs and losses of public management programs and ecological states associated with a biological invasion. Management programs are undertaken before ecological states are revealed. Other combinations of ecological states and management actions are possible.

	Ecological state				
Management acts	Transport unviable	Transport viable; establish unviable	Transport viable; establish viable; slow spread; low virulence	Transport viable; establish viable; slow spread; high virulence	Transport viable; establish viable; rapid spread; high virulence
Assess risk + quarantine	medium cost + low loss (t)	medium cost + low loss (t)	medium cost + medium loss (t,m,nm)	medium cost + high loss (t,m,nm)	medium cost + very high loss (t,m,nm)
Monitor ports + ecosystems	medium cost	medium cost	medium cost + low loss (m,nm)	medium cost + medium loss (m,nm)	medium cost + high loss (m,nm)
Monitor ports + ecosystems + aggressive eradication	medium cost (no eradication)	medium cost (no eradication)	very high cost + low loss (m,nm)	very high cost + medium loss (m,nm)	very high cost + high loss (m,nm)
Monitor ports + delayed control	low cost (no control)	low cost (no control)	low cost + low loss (m,nm)	medium cost + high loss (m,nm)	high cost + very high loss (m,nm)
Beliefs as to states	Π^1	Π^2	Π^3	Π^4	Π^5

Note: t refers to trade losses, m refers to market losses, and nm refers to non-market losses.

management actions, and estimates of the costs and losses associated with each combination of ecological state and management act. Table 19.2 also illustrates various economic trade-offs that must be considered when designing a biological invasion protection program. Understanding the tradeoffs between program costs and economic losses is a major challenge in the design and development of programs to counter the threat of biological invasions.

2.2.2 Endogenous risk

The standard EU model cannot address a class of economic phenomena known as moral hazards, which are acts people undertake that alter the risks they face. Insurance companies, for example, are concerned with moral hazard because people who buy insurance might then engage in extra risky behavior. Ehrlich and Becker (1972) used state-preference theory to evaluate insurance purchase decisions recognizing that risk can be influenced by decision-makers, and that alternatives to market insurance exist that can reduce the consequences of undesirable states of nature. They defined self-protection as actions designed to decrease risk, and self-insurance as actions designed to reduce consequences. This so-called endogenous-risk approach has been applied to analyses of optimal programs for invasive species management by integrating economic and ecological information (Shogren 2000, Leung et al. 2005).

Analytical endogenous risk models typically simplify states of the world to be dichotomous—either a state of the world occurs or it doesn't. For example, Leung et al. (2005) present an invasive species model where nature is either invaded or uninvaded. As with the EU model described in equation (19.1), social utility in the endogenous risk model is the welfare associated with a state multiplied by the probability of being in that state. The public decision-makers' problem is to invest in mitigation (M, or self-protection) and adaptation (A, or self-insurance) programs so that social welfare is maximized:

$$\begin{aligned} \underset{M,A}{Max}\, U = {} & \pi^u(M)[\upsilon^u(w - L^t(M) - C(M,A)] + \\ & (1-\pi^u(M))[\upsilon^i(w - L^p(A) - C(M,A))] \end{aligned} \qquad (19.2)$$

where u is the uninvaded state, i is the invaded state, v^u (v^i) is the utility associated with being in the uninvaded (invaded) state, w is endowed forest wealth (market and non-market values), L^t is the loss to trade from quarantines and standards, L^p is the loss to the production of market and non-market values in the invaded state, and C is a cost function. Equation (19.2) illustrates that some costs (such as risk assessments and port inspections) and losses (such as trade losses due to quarantines) will be incurred even if an invasion does not occur. Likewise, investments in adaptation programs are needed prior to the state of nature being revealed so that resources are in place in the event of an invasion.

In this simple model, the first-order condition for the optimal level of investment in mitigation programs yields the following expression:

$$\pi^u\left[-\left(\frac{\partial\upsilon^u}{\partial L^t}\frac{\partial L^t}{\partial M}\right)-\left(\frac{\partial\upsilon^u}{\partial C}\frac{\partial C}{\partial M}\right)-\left(\frac{\partial\upsilon^i}{\partial C}\frac{\partial C}{\partial M}\right)\right]=\frac{\partial\pi^u}{\partial M}\left[\upsilon^u-\upsilon^i\right]. \qquad (19.3)$$

The left-hand side of the expression is the expected marginal welfare loss from trade reduction plus mitigation expenditures. This is equated with the change in welfare induced by the marginal effectiveness of mitigation efforts in altering the risk of an invasion. As welfare losses from trade reductions and mitigation expenditures constitute social costs, and the welfare changes induced by mitigation effectiveness are benefits, Equation (19.3) simply states that mitigation should be undertaken up to the point where expected marginal costs equal expected marginal benefits. Similarly, the first-order condition for the optimal level of adaptation programs is:

$$\frac{\partial \upsilon^i}{\partial L}\frac{\partial L}{\partial A}=\frac{\partial \upsilon^i}{\partial C}\frac{\partial C}{\partial A}+\frac{\partial \upsilon^u}{\partial C}\frac{\partial C}{\partial A} \tag{19.4}$$

providing the result that adaptation investments should be made up to the point where the marginal benefits of adaptation (the reduction in losses) equal their marginal social costs.

2.2.3 Uncertainty and subjective probability

One of the critical variables highlighted in equations (19.1) through (19.4) is π, the probability that a well-defined state of nature occurs. Assigning an accurate value to π is difficult because biological invasions are novel events. Although estimates of the average risk that an introduced species will become a pest can be computed using lists of introduced species for which their success or failure is known (Reichard and Hamilton 1997), it is not known how well past invasions can realistically predict the risk of future invasions.[6]

Invasion probability might be considered as a degree of belief, which is applicable to both unique and repetitive events (Pratt et al. 1964). Treating probability as a degree of belief is the Bayesian approach to decision making and allows the analyst to incorporate both prior information and uncertainty in a model of subjective probability. Given limited information on ecological states for biological invaders, the decision-maker chooses a prior distribution to represent their degree of belief regarding the stages of a biological invasion (Clark 2005, Wikle 2003). As new information is acquired, the prior probability distribution can be updated using Bayes' rule:

$$\pi(\theta \mid y, x) \propto \pi(y \mid \theta, x) \bullet \pi(\theta) \tag{19.5}$$

where $\pi(\theta)$ is the prior probability, $\pi(\theta|y,x)$ is the posterior probability, y is the dependent variable of interest (such as the extent of a biological invasion), x is an explanatory variable (such as the level of adaptation effort), and $\pi(\cdot)$ now represents a distribution rather than a scalar value. The posterior distribution describes the subjective uncertainty about the probability which is proportional to the likelihood of observing y (given x and parameter θ) times the prior probability. As data accumulate, the posterior probability becomes the prior probability and learning occurs.

Uncertainty can be incorporated in the endogenous risk model of optimal invasive species management. For example, Shogren (2000) presents an economic model where uncertainty is an integral part of the decision-making framework:

[6] However, Reichard and Hamilton (1997) found that the single best predictor for invasive plants is whether a species was known to invade elsewhere in the world.

$$\underset{M,A}{Max}\,U = \left(\int_a^b \left\{ \begin{array}{l} \pi^u(M;\theta)\upsilon^u[w - L^t(M;\theta) - c(M,A)] \\ + (1-\pi^u)(M;\theta)\upsilon^t[w - L^p(A;\theta) - c(M,A)] \end{array} \right\} dF(\theta;\beta)\right) \qquad (19.6)$$

where most variables are as defined in equation (19.2), θ is a random variable reflecting uncertainty about parameter values, and F is the cumulative distribution bounded over the support (a,b) of the random variable θ. Note this model introduces uncertainty not only in the probability of observing states of nature but also in the level of realized damages. Although stringent data requirements would render this model difficult to operationalize, Shogren (2000) demonstrates a manager will maximize expected welfare by selecting levels of M and A that equate the marginal cost of influencing the severity and probability of an invasion with the marginal wealth acquired (or damages avoided). Perhaps of greater interest are the implications that a lower value of π^u will always increase investment in adaptation activities (A) and may decrease or increase investment in mitigation activities (M).

Although equations (19.2) through (19.6) were presented to represent the social welfare maximizing problem of a public decision-maker, the expected utility model is quite general and can be applied to the decisions facing private forest landowners. Linking an expected utility decision-making model for private forest landowners with the weakest-link social composition function introduces a new source of uncertainty—will all landowners in a community make investments in forest protection? A general expression for the private forest landowners' decision can be written:

$$\underset{M,A}{Max}\,U_i = \int_a^b \left\{ \begin{array}{l} \pi^u(M_i \mid \sum_{j=1}^{n-1} M_j(\theta))\upsilon^u[w_i - c_i(M_i,A_i)] + \\ (1-\pi^u)(M_i \mid \sum_{j=1}^{n-1} M_j(\theta))\upsilon^t[w_i - L_i^p(A_i \mid \sum_{j=1}^{n-1} M_j(\theta)) - c_i(M_i,A_i)] \end{array} \right\} dF(\theta;\beta)) \qquad (19.7)$$

where the summation over $M_j(\theta)$ expresses the idea that the mitigation expenditures made by each member of the community influences both the probability of a biological invasion and the losses if an invasion occurs. Subjective uncertainty about the forest protection behavior of one's neighbors is a key element in decision-making by each individual (i) in the community of n landowners (equation 19.7). In the case study presented in section 4, we will note how the weakest-link character of private forest health protection decisions are influenced by recognition of the positive externalities created by individual investments in invasive species control. We also note the cost of control c_i (M_i, A_i) might include an argument for externalities (E_i), such as the unanticipated effects of mitigation and adaptation on non-target species.

2.2.4 Predictability and the base rate effect

Over time, as data accumulate, the subjective posterior probabilities converge towards some objectively correct probabilities. Even armed with correct probabilities, however, decision-makers face the problem that invasion probabilities are quite low. For example, Williamson (1996) has argued that roughly 0.1 percent of introduced exotic species eventually become pests. The rarity of an event greatly complicates predictability, even if predictions are accurate (Smith et al. 1999).[7] This is known as the base rate effect, and can be illustrated as follows. Suppose that an invasive species screening system is 90 percent accurate in identifying true invasive and true noninvasive organisms, and that the risk of an introduced species becoming a pest is 0.1 percent. If 10,000 organisms are screened, then roughly 1,000 true noninvasive species will be incorrectly identified as invasive. This is roughly two orders of magnitude greater that the number of true invasive species that are correctly identified. Although the screening system is quite accurate, the predictions of which organisms are truly invasive are quite poor (Smith et al. 1999, Keller et al. 2007). The base rate effect may therefore induce risk reduction policies that are overly restrictive. On the other hand, the potential for catastrophic forest damage from a novel invader suggests that application of the precautionary principle may be warranted.

A final issue that needs to be raised is the fact that rational decision-making under conditions of risk and uncertainty requires effort. When decision-makers are confronted with events that have a very low probability of occurrence, they often rely on *ad hoc* decision rules, or heuristics, rather than fully rational responses (Camerer and Kunreuther 1989). In particular, the *threshold effect* posits that, if probabilities fall below some threshold, they are treated as though they are zero (Slovic et al. 1977). Consequently, some individuals might entirely discount the risk of a biological invasion, thinking that "it can't happen to me". We expect that this behavior is especially likely when private forest owners are confronted with forest protection decisions, and may be a contributing factor to weakest link behavior in communities.

3. MANAGEMENT OF INVASIVE FOREST PESTS

As previously mentioned, each stage of a biological invasion can be linked with a strategy for mitigation or adaptation (fig. 19.1). In this section, we provide an overview of some major forest pest management programs in the United States that were undertaken to combat invasive species. Unfortunately, very few economic analyses have been conducted to assess the relative success or failure of these programs.

[7] Accuracy here refers to a situation where true invaders and true non-invaders are correctly identified.

3.1 Risk Assessments, Standards, and Quarantines

The first line of defense against a biological invasion is to prohibit potential invaders from crossing national borders. This strategy is implemented by conducting risk assessments of potential invaders that may hitch-hike in products of international trade (USDA Forest Service 1991) and by establishing mitigation standards that would assure the destruction of unwanted organisms either at the port of origin or the port of entry. In some cases, quarantines may be warranted.

Prior to the late 19th century, the idea of protecting agricultural and forest systems from biological invasions was not seriously considered (Popham and Hall 1958). The first legislation used to protect plant resources in the United States from potential biological invasions was the Quarantine Act of 1912. This Act was passed largely in response to the devastating effects resulting from two forest diseases—the chestnut blight and white pine blister rust (Anderson 2003), and Quarantine Number 1 prohibited the importation of 5-needle pines (Maloy 1997). Further protection to agricultural and forest producers was provided by the 1957 Plant Protection Act which allowed the USDA to make predeparture inspections of plant material at sites such as Hawaii and Puerto Rico, and impose quarantines without a public hearing and without notice (Bryson and Mannix 2000).

By definition, quarantines limit trading opportunities between countries and they have long been accused of functioning as tariffs to protect favored industries (Campbell 1929). The Uruguay round of talks on the General Agreement on Tariffs and Trade include Sanitary and Phytosanitary Standards (SPS) which are designed to limit the risk posed by trade in agricultural and nursery products. Although trade liberalization has generally reduced tariffs on agricultural and nursery products, it is widely acknowledged that SPS can restrict trade, especially for developing countries that cannot afford the means to attain imposed standards (Henson and Loader 2001). The benefits of quarantines to the country that impose them directly depend on their effectiveness in preventing new invasions.[8] However, quarantine effectiveness is difficult to evaluate due to the scarcity of comparative data that would permit scientific analysis, and it has been noted that many damaging pests have been introduced into the United States since the Quarantine Act of 1912 (Mathys and Baker 1980).

[8] The U.S.D.A. agency charged with responsibility for implementing plant inspections and quarantines is the Animal and Plant Health Inspection Service (APHIS). This agency was created in 1970 by removing the regulatory functions from the research oriented Agricultural Research Service and creating an independent agency. The Plant Protection and Quarantine Program was formed that year and placed under the new agency. Also during that year, the United States became a signatory to the 1952 International Plant Protection Convention.

3.2 Eradication

If an exotic organism slips through a quarantine, plant inspection or treatment, the second line of defense is to initiate an eradication program with the intent of forcing the extinction of a newly introduced organism before it becomes established in native ecosystems. The processes by which exotic organisms become established are highly stochastic (Liebhold et al. 1995), and strongly influenced by propagule pressure (Von Holle and Simberloff 2005). Forced extinctions are more likely to result from early and aggressive suppression efforts while population numbers are limited and Allee effects may be used to advantage.

As noted by Maloy (1997), "it is human nature to do something in a crisis even if it is a long shot" (p. 105). The largest invasive forest pathogen eradication program undertaken in the United States, in terms of time, money and labor, was in response to white pine blister rust, which was first discovered in 1906 on pine seedlings imported from Europe (Maloy 1997). White pine blister rust requires an alternate host to complete its life cycle—cultivated and wild currants and gooseberries (*Ribes* spp.)—and control was focused on destroying these extensively distributed hosts. Destruction of wild *Ribes* was labor intensive, especially in the remote and rugged terrain of the western U.S. A federal government eradication program wasn't initiated until 1933, some 27 years after the disease was first discovered. Federal involvement in white pine blister rust eradication may have been as much of a political decision as a forest management decision, as the initiation of the program coincided with the Great Depression. During the years 1933-40 the program rapidly expanded due to low-cost labor provided by the CCC. Although *Ribes* eradication efforts were ultimately applied to more than 20 million acres nationwide, the success of the eradication program was often called into question. An economic analysis of the program in the Lake States was very critical (King et al. 1960) and the program was discontinued in 1966. Over the roughly 34 years of the program, it is estimated that $150 million (in nominal dollars) was spent on control (Maloy 1997).

The first forest insect eradication program implemented in the United States was the attempt to wipe out the European gypsy moth. Although the pest was accidentally introduced in 1869, the initial governmental appropriation, made by the Massachusetts legislature, did not occur until 1890. Some have argued that the aggressive eradication program over the next 10 years was successful, and that eradication was nearly achieved. However, perhaps due to the apparent success of the program, funding was discontinued and the range of the insect spread rapidly. Subsequent to World War II, DDT was sprayed on outlying infestations which led to the successful eradication of the pest on nearly 4 million acres in Michigan, Pennsylvania and New Jersey, and complete eradication was considered a possibility (Popham and Hall 1958).

3.3 Control

Once an exotic organism becomes established in a native ecosystem, eradication becomes difficult, if not impossible, and control programs can be attempted to limit the growth and spread of the organism. Such adaptation programs buy time for both public and private forest owners to alter their management strategies (Waring and O'Hara 2005) and allow scientists the opportunity to discover new methods for eradication (Hain 2006). Control programs attempt to alter the spatial and/or temporal population growth dynamics of an invasive species while recognizing that complete eradication is unlikely.

With the elimination of DDT as an eradication tool in the United States, the gypsy moth has steadily continued to expand its range. Recent efforts have shifted from a strategy of eradication to control, by "slowing the spread" (STS) of the organism.[9] The contemporary STS gypsy moth program focuses control efforts on creating a barrier zone along the leading edge of the population front by targeting isolated insect communities. Sharov and Liebhold (1998) conclude that the STS program has recently slowed population spread in the Appalachian region of the United States by 59 percent.

Another important strategy for controlling invasive forest insects is the use of biological controls. Classical biological control is the control of exotic pests by means of importing their natural enemies from their country of origin.[10] The identification of potential biological control organisms is a complicated and lengthy process (Pschorn-Walcher 1977) and concerns have been raised about risks to native ecosystems (Simberloff and Stiling 1996).

3.4 Cost-benefit Analysis

Our review of the literature reveals that economic analyses of forest invasive species programs are rarely conducted. Consequently, the efficiency of investments in these programs cannot be evaluated. Commonly used measures of economic damages to forests from invasive species are solely focused on lost timber values, and are computed by multiplying the price of timber by an estimate of the annual quantity of timber destroyed (Pimental et al. 2000). This approach does not include the broader impacts of exotic forest pests on non-market economic values and is therefore biased downwards, perhaps severely. For example, *P. ramorum* infections in California and Oregon are causing enormous mortality to oaks and other tree species on public and private landscapes and yet none of the impacted species have important uses as timber (Rizzo et al. 2005). We anticipate that understanding the non-market economic impacts of *P.*

[9] This strategy was initially attempted in 1923 by creating a barrier zone from Long Island to Canada along the Hudson River.

[10] The National Biological Control Institute was established in 1990 to provide leadership for biological control and functions under the auspices of APHIS.

ramorum and other exotic forest pests on forest ecosystems will make a major contribution to cost-benefit analyses of invasive forest pest programs (Holmes and Kramer 1996, Rosenberger and Smith 1997, Kramer et al. 2003, Holmes et al. 2006).

4. PUBLIC AWARENESS AND THE HEMLOCK WOOLLY ADELGID

The risk and uncertainty associated with most biological invasions, combined with the weakest-link public good characteristics of forest health protection programs, may help to explain why mitigation and adaptation strategies have lagged far behind the initial arrival and establishment of exotic species. One of the key factors in developing a rapid response to invasive forest species is the participation of the public (U.S. Government Accountability Office 2006). This is especially true in the eastern United States where private forest land dominates the forest landscape.

Ongoing research funded by the USDA Forest Service to better understand the economic impacts of the hemlock woolly adelgid (HWA), an exotic forest insect inadvertently introduced from Japan, demonstrates how economic analysis can be used to support management responses to invasive forest pests. In this section, we bring attention to the results of a pilot project completed as part of this ongoing research project, and focus attention on the role of public awareness in private forest protection actions.

4.1 The HWA Problem

The HWA is currently spreading across the eastern United States and threatens the widespread decline of eastern hemlock forests (Orwig and Foster 1998). The spread of HWA is facilitated by wind as well as the movements of birds, mammals, humans and the leading edge of an infestation travels at an approximate rate of 30 kilometers per year (McClure 1990). Roughly twenty-five percent of the 1.3 million hectares of eastern hemlocks in the United States are currently infested with HWA and experts predict that the remaining 75 percent may become infested within 20 to 30 years (Rhea 2004). There are no known effective native predators of this insect and eastern hemlock has shown no resistance to HWA, nor has it shown any recovery following heavy, chronic infestation (Orwig and Foster 2000). Eastern hemlock forests provide a suite of public and private goods that have economic value, including wildlife habitat (Benzinger 1994, Evans et al. 1996), aesthetic quality in residential areas (Holmes et al. 2006, chapter 11 of this book), sales of nursery stock (Rhea 2004), and commercial timber (Howard et al.1999). As the impacts of this invasion accrue, forest managers' demand for information increases.

4.2 HWA Management

The management of HWA relies on an integrated system of mitigation and adaptation activities. State-level quarantines have been imposed to regulate the transport and sale of infested ornamental stock and infested hemlock logs (Bofinger 2002, Gibbs 2002). Eradication of HWA requires treatments to individual hemlock trees and is not considered a forest-wide option. Arborists eradicate HWA infestations on individual trees through semi-annual drenching with horticultural oils and insecticidal soaps.[11] Trunk injections of chemical insecticide are also effective over the short-term in eliminating HWA on individual trees (Pais and Polster 2000). At the forest level, biological control is attempted via release of an exotic predatory beetle, and experimental releases of beetles have been authorized by federal and state agencies in limited areas of highly infested forest since 1988 (Pais and Polster 2002).[12] Although these experiments have revealed the potential of biological control, the effectiveness of this approach remains uncertain (Knauer et al. 2002, McClure and Cheah 1999).

4.3 HWA in Maine

A pilot study undertaken by Byrne (2004) examined public awareness of HWA and its role in household control decisions for residential landscapes in Maine. Household control of HWA through spraying and tree removal might play an important role in reducing the risk of spread to uninfested areas ($\pi^u(M_i|\Sigma M_j(\theta))$ in equation (19.7) and evidence of whether or not an informed public can effectively aid in the control of HWA is of great value to forest resource managers. The weakest-link characteristic of controlling HWA to protect eastern hemlock raises the question of whether increased awareness can improve the effectiveness of management efforts or policy outcomes. Because public involvement is typically contingent on knowledge or awareness (Janicke 1997), an investigation of the factors that influence levels of awareness is warranted.

HWA was first discovered in Maine in 1999 as an isolated spot infestation resulting from infested nursery stock shipments, which, as of 2000, are quarantined by the State of Maine. HWA was not observed in a natural setting in Maine until 2004. Existing infestations have been controlled and monitored by state forest management agencies since 2000. A 2-year public awareness campaign consisting of newspaper and television announcements has proven to be critical

[11] According to a leading insect control company that specializes in treating hemlock for HWA, trunk and soil injections range from \$35 up to \$75 per tree depending on tree diameter, and foliar spraying costs approximately \$550 per acre depending on the location of the infestation.

[12] Biological control efforts in the Great Smoky Mountains National Park cost approximately \$6,000 per acre, and are being applied in old growth and interior backcountry areas (U.S. National Park Service, *personal communication*).

in the identification of HWA infestations (Ouellette 2002). As part of its management response to HWA infestations on residential properties, the Maine State Forest Service has compensated homeowners for the cost of treatment which, in most cases, involved removing and destroying infested hemlock trees.

4.3.1 Maine pilot study

In 2004, survey responses were collected from a sample of Maine residents using a multi-mode survey method that employed a web-based survey instrument and a mail survey instrument, identical in questions and format. A sample of 415 Maine residents was drawn from a list maintained by the Maine State Forest Service Entomology Lab consisting of residents who had contacted the Maine Forest Service within the previous 3 years and whose interactions with staff had been classified by staff as HWA-related. This sample is expected to have less variation in awareness and control responses than would a random sample of Maine households. However, the lack of information about the presence and location of hemlock trees on residential lands in Maine necessitated the use of this informed sample within the limited time frame of the pilot study.

At the end of the 9-week data collection period, which was supported by the Maine State Forest Service, a total of 81 surveys were completed either online (61) or by mail (20), resulting in a response rate of approximately 25 percent. Of the 81 households responding to the survey, 63 reported having hemlock trees on their property, and of this number 21 households reported that actions had been taken to control or eradicate HWA in their yard. When asked about the extent to which various motives influenced the decision to control or eradicate HWA, the two most influential motives selected were (1) "To maintain the health of hemlocks on my property" (16 households), and (2) "To maintain the health of other hemlocks in my neighborhood" (16 households). These responses indicate that, of the households that have acted to control or eradicate HWA on their property, the majority were motivated to a "very great extent" by their awareness that their actions could affect the health of other trees in their neighborhood.[13] Although the evidence is limited, this response indicates a degree of awareness among landowners regarding the weakest-link nature of household forest protection decisions.

Two empirical models were estimated using survey responses. The first approach employed an ordered logit model of categorical responses ranking self-reported awareness, along a Likert scale, of HWA. This model assumes that individuals are able to make meaningful distinctions between awareness levels in self-reports when asked to what extent they are aware or knowledgeable of HWA. The second empirical analysis employs a binary logit model of household

[13]The degrees of motivational influence included in the survey question were categorized as "very great extent", "to some extent", "a small extent", "not at all", and "don't know".

management decisions featuring individual household awareness as an explanatory variable. The dependent variable in this model is based on responses to the hypothetical question of whether or not a household would control or eradicate HWA if there were an infested tree in their yard.

Variables selected to explain public awareness levels include socio-economic characteristics of individual household members (income, gender, age, education, employment), the types of media they use to learn more about HWA (television, newspaper, radio, internet, and magazine), sources they may have used to gain information about HWA (state government agencies, university extension staff, landscaping firms or nurseries), characteristics expected to affect perceived awareness (membership in an environmental organization, gardening or tree club, prior control/eradication experience with HWA) and household landscape characteristics (acreage, percent tree coverage and the presence of hemlock trees).

As expected, most respondents reported some degree of HWA awareness. Four percent reported being aware to a very great extent, 46 percent reported being aware to some extent, 36 percent reported being aware to a small extent, and 14 percent were not at all aware. Table 19.3 displays results of the ordered logit model, which are largely consistent with research findings in the political science literature examining environmental and public policy knowledge (Steel et al. 1990, Steel 1996).

According to the empirical estimates (table 19.3), socio-economic characteristics play a significant role in HWA awareness. This finding, in combination with spatial proximity of households to hemlock resources, can be used to help target public information campaigns. Reported awareness is positively correlated with income, male gender, age, and membership in an environmental organization. Contrary to expectations, however, the empirical results suggest that respondents with college degrees are less likely to report higher awareness levels.[14] As anticipated, the effect of prior control/eradication experience with HWA is also a significant factor that positively effects reported awareness levels. This factor is important because, as described above, households with prior control/eradication experience were motivated to a "very great extent" by the awareness of their forest protection actions on the health of other hemlocks in their neighborhood.

When asked the question "If you had an infested hemlock tree in your yard, would you control or eradicate hemlock woolly adelgid", eighty-eight percent of the sample responded "yes" while the remaining 12 percent responded "don't know". Presumably, survey participants will respond "yes" if the expected net

[14] A careful examination of the data suggest the possibility that more education might induce greater caution about overstating perceived awareness.

[15] The cost of foliar spraying to control HWA in Maine is roughly $260/tree/year, which can be quite expensive if several hemlocks are located on a landowners' property. For example, more than one-half of the respondents in our sample reported 11 or more hemlocks were located in their yard, indicating that annual treatment per household could cost thousands of dollars.

Table 19.3. Ordered logit regression parameter estimates for variables influencing public awareness of the hemlock woolly adelgid in Maine.

Variable	Coefficients (t-value)	Variable	Coefficients (t-value)
College degree	-1.1773* (-1.82)	Environmental Org	1.2438* (1.83)
Income	0.0207*** (2.58)	Garden/Tree Club	-0.1451 (-0.20)
Male	10.3643*** (2.90)	Prior HWA Experience	1.6793** (2.40)
Age	0.1360** (2.49)	Employed in Forestry	-1.5556 (-1.11)
Age*Male	-0.1791*** (-2.89)	State Government	0.6840 (1.06)
Television	-1.0045* (-1.62)	University Extension	0.8262 (1.29)
Newspaper	0.3824 (0.61)	Landscape/Nursery	-0.9839* (-1.60)
Radio	-0.0048 (-0.01)	Yard Size (acres)	-0.6158*** (-3.04)
Internet	0.5826 (0.91)	Percent Tree Coverage	0.0052 (0.40)
Magazine	0.6113 (0.93)	Have Hemlocks	0.0895 (0.13)

Likelihood Ratio 40.6280
χ^2 Probability 0.0042
Observations 78

Note: * 10% significance, ** 5% significance, *** 1% significance.

benefits from management are positive.[15] The fact that not a single respondent answered 'no' might be interpreted in two ways. First, respondents might truly be uncertain about a variety of factors associated with the scenario including the potential damage that would be incurred by an infested tree and the possibility that an infestation could spread to trees in their yard or neighbors' yards. Second, respondents may be exhibiting compliance bias, a situation where respondents' consciously or unconsciously rationalize that to answer 'no' is "socially irresponsible" (Kemp and Maxwell 1993). Given that sampled households had previous contact with the Maine State Forest Service, it is not surprising they many may have felt it irresponsible to answer 'no'. For purposes of this analysis, 'don't

Table 19.4. Binary logit regression parameter estimates for the household control of hemlock woolly adelgid in Maine.

Variable	Coefficients	t-value
Intercept	-2.1930	-0.93
Awareness	2.2383**	2.23
College Degree	2.8549	1.54
Income	-0.0285*	-1.82
Environmental Org	0.4343	0.26
Garden/Tree Club	-3.1211**	-1.96
Yard Size (acres)	-0.1162	-0.29
Tree Coverage (%)	-0.0125	-0.43
Number of Hemlocks	0.1738	1.49
Driveway/Border	-3.0015*	-1.92
Likelihood Ratio	25.7520	
$\chi 2$ Probability	0.0022	
Observations	63	

Note: * 10% significance, ** 5% significance.

know' responses are interpreted as reflecting uncertainty about intended actions, while 'yes' responses are interpreted as statements that the household will invest in HWA control with certainty.

Only respondents with hemlocks on their property were used in the logit analysis (63 observations). The key finding is the positive, statistically significant effect of awareness on the household control decision (table 19.4). Consistent with results reported by Miller and Lindsay (1993) for a study of gypsy moth control in New Hampshire, this result indicates that individuals who reported higher awareness levels are more likely to invest in the control of invasive species. This result suggests programs designed to increase public awareness about HWA can encourage household control and reduce the risk of spread. We also note the statistically significant, negative signs on "garden/tree club" and "driveway/border" may reflect concern with the effect of chemical treatments on non-target organisms.

Of course, our use of an informed sample does not allow us to generalize these results to the entire Maine population. Nonetheless, this case study identifies characteristics associated with household awareness levels and a stated intention to pursue private adaptation behavior in the context of HWA, and establishes a positive relationship between the two. It also demonstrates how economic theory and methods can be used to support management responses to HWA. Our use of social, economic, and landscape data suggests that, if more extensive data were collected for a larger number of respondents, model results could help target forest protection efforts to areas characterized by low awareness and high uncertainty.

5. CONCLUSIONS

Invasive forest pests have been a persistent problem plaguing forest managers in the United States for more than a century. Despite the fact that hundreds of millions of dollars have been spent on eradication and control of exotic forest pests, comprehensive economic analysis of the costs and benefits of these programs are almost non-existent. The lack of comprehensive economic assessments of the effects of invasive forest species on the production of market goods (such as timber) and non-market economic values (such as aesthetics) has stymied meaningful economic analyses. We see this gap as one of the greatest issues facing the development of more rational and effective forest pest management systems.

This chapter reviewed four key economic concepts that we think are integral to the design of socially optimal programs for combating invasive forest pests. First, forest health protection is a public good. Private provision of forest health protection is expected to be sub-optimal because self-interested individuals do not account for the benefits that flow to other members of society when they make forest protection investments. This context provides the justification for government intervention in forest health protection.

Second, forest health protection is a weakest-link public good. The weakest-link character of forest health protection relegates the level of forest protection attained by a community to the weakest members of the community. Consequently, effective forest health protection programs require that the weakest links be strengthened. We argue that forest health protection programs can be enhanced by targeting information to those most likely to engage in risky behavior. In particular, information describing the weakest-link nature of forest protection should be targeted at private landowners to enhance the likelihood that they will participate in forest protection programs. As indicated in our case study, weakest links can be identified using economic surveys of household behavior.

Third, the design of optimal strategies for managing invasive species is highly complex due to the trade-offs that must be evaluated between the costs of management actions and the economic losses to trade, the production of market goods, and the provision of non-market economic values. Data are costly to obtain and until decision-makers recognize the value of economic information, they are unlikely to invest in its collection.

Finally, each biological invasion represents a novel situation. However, mitigation and adaptation investments must be made prior to the time at which the true state of nature is ultimately revealed. Pervasive uncertainty regarding the parameters of economic and ecologic models argues for the necessity of treating uncertainty in as pragmatic a fashion as possible. Bayesian methods provide a useful approach for incorporating uncertainty about data, parameters, and processes in models of inference and prediction. As new information is realized, and uncertainty is reduced, economic models of optimal decision-making can be updated. We anticipate that Bayesian approaches to learning about the risks and economic consequences of biological invasions will provide a substantial contribution to the adaptive management of invasive forest pests.

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