The Economics of Terrestrial Invasive Species: A Review of the Literature

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This paper reviews the literature on the economics of invasive species management as it applies to invasive species in general and terrestrial invasive species in particular. The paper summarizes a number of recent studies that assign values to the economic impact of terrestrial invasive species on a national scale. This is followed by a review of the economic literature on control and prevention of a biological invasion and the literature on international trade and trade policy with invasive species. The paper then reviews selected studies on terrestrial invasive plants, animals, and microbes, respectively.

Key Words: terrestrial invasive species, prevention, control, international trade, bioeconomic modeling

Throughout history the spread of plants, animals, and other organisms has been governed by natural ecological processes and has accompanied trade in goods and services and the movement of humans. As a consequence, species are continually introduced to areas outside their native geographic location and some of these species establish themselves as harmful invaders. Invasive species are one of the leading causes of global ecological change. Of 256 vertebrate extinctions with an identifiable cause, 109 are known to be due to biological invaders, while 70 are known to be caused by human exploitation (Cox 1993). In the United States, it is estimated that 40 percent of the threatened or endangered species are at risk due to pressures from invasive species (Nature Conservancy 1996, Wilcove et al. 1998). Invasive species also impose significant economic losses to consumer and producer welfare.

The problems associated with invasive species are not new and have long been recognized. U.S. invasive species policy dates to the Lacey Act of 1900. In recent years, however, increased globalization has led scientists and policymakers to focus more attention on the potential costs associated with invasive species introductions. Of the nearly 30 federal U.S. acts pertaining to invasive species, approximately half have been adopted since 1990 (National Agricultural Library 2006).[1]

As recognition of invasive species problems has grown, so has the economics literature. The purpose of this paper is to review the methodological literature on the economics of invasive species management as it applies to invasive species in general and terrestrial invasive species in particular. The paper is organized as follows. It begins with a summary of a number of recent studies that assign values to the economic impact of terrestrial invasive species on a national scale. This is followed by a review of the economics literature on control and prevention of a biological invasion. The section after that surveys the literature on international trade and trade policy with invasive species. The paper then reviews selected studies on terrestrial invasive plants, animals, and microbes, respectively. The final section contains brief, concluding remarks.

There are important aspects of the literature that are not reviewed here. The focus of this survey is only on terrestrial invasive species. Aquatic invasive species are examined in the companion

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[1] The fact that policy measures have increased contemporaneously with trade volumes has helped mitigate potential introductions. A 1993 report by the Office of Technology Assessment found no clear evidence that the rate of observed invasive species imports increased over the previous 50 years (Office of Technology Assessment 1993).

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paper by Lovell, Stone, and Fernandez (2006). In addition, this paper does not attempt an exhaustive survey of the numerous case studies of individual species in specific locations that assign values for control costs or damages or both. Many of these are referenced in the recent studies that value the economic impact of invasive species on a regional or national scale. These studies combine estimates of invasive species impacts from a large number of different sources whose reliability varies tremendously. It is beyond the scope of this paper to evaluate the assumptions that underlie each disaggregate estimate. Instead, it is left to the interested reader to consult the sources cited in the next section.[2]

A number of issues are closely related to the economics of invasive species. Among them are the economics of intraseasonal pest management (Carlson and Wetzstein 1993), the economics of resistance (Laxminarayan 2002), and the economics of infectious disease control in humans (Philipson 2000). While the literature on these topics is relevant for the management of invasive species, it is not the focus of this survey.

Invasive species are biological resources. As such, the economic modeling of invasive species problems has its roots in the literature on the bioeconomics of renewable resources (e.g., Clark 1990). At the same time, there exist important differences in the characteristics of the two problems and the policy questions of concern. While renewable resources are typically viewed as valuable, invasive species are pests that cause damage and are sometimes referred to as biological pollution. For example, the value of annual crop losses to weeds in the United States has been estimated at $20 billion in 1991 dollars (USDA 2000), with roughly 50–75 percent of the costs attributed to nonindigenous weed species (Office of Technology Assessment 1993). In economic models of renewable resources, a primary concern is to characterize the harvest policy that maximizes economic welfare over time, including consumptive benefits from harvest, the cost of harvest, and non-consumptive benefits associated with the resource stock. For invasive species, the primary concern of management is to reduce damages through prevention or control, or both, in order to minimize the discounted sum of damages and prevention and control costs over time. A significant amount of resources is spent on these activities. In 2000 and 2001, global pesticide expenditures were over $30 billion (Keily, Donaldson, and Grube 2004). This includes expenditures on both indigenous and non-indigenous pests. Prevention activities account for approximately half of U.S. federal expenditures for invasive species (National Invasive Species Council 2001).

Both control and prevention can involve a variety of inputs or policy instruments. Harvesting, chemical or biological controls, and mechanical or manual removal may be used to reduce the size of an invasion. In many cases, cooperation between public agencies and private stakeholders is important if effective control is to be achieved. Monitoring can improve control efforts by allowing rapid, targeted responses to pest outbreaks. Preventive mechanisms include trade bans, inspection and quarantine, treatment, and export pre-clearance programs. Each of these inputs and policy instruments has its own characteristics. The diverse set of ecological, economic, and policy issues that distinguish invasive species problems has led to many interesting research ideas.

The Economic Impacts of Terrestrial Invasive Species

A number of recent studies attempt to value the economic impact of invasive species on a national scale. Table 1 summarizes the findings of these studies for terrestrial invasive species. Not all invasive species are included, nor are all invasive species impacts. Individual estimates are calculated in a variety of different ways, including extrapolation from small to large scales, and the estimates include measures of both damages and control costs. It is not uncommon for aggregate values to be obtained by multiplying a constant marginal damage per pest by an estimate of the total pest population (or pest units if population is not the measure). While this provides useful information about the potential magnitude of damages, it is not a reliable statistical estimate. If the pest population is very large, then a relatively small change in the damage assigned to one individual can lead to differences in aggregate values

[2] Born, Rauschmayer, and Bräuer (2005) provide a non-exhaustive review of 23 papers and classify them as decision aids or impact assessments. They examine 10 papers in detail that focus primarily on agricultural pests in only a few countries, and on an *ex post* evaluation of control that typically does not account for uncertainty.

Table 1. Annual Economic Impact of Terrestrial Invasive Species on a National Scale

	Type of Invasive		
Country	Plant	Animal	Microbial
Australia (in $AU)	4 billion[a]	491.5 million[b] (9 vertebrates) 703.9 million[c] (10 vertebrates, includes environmental costs)	
Canada (in $CAN)	38.21 million[d] (leafy spurge and knapweed)	101.3 million[d] (3 invertebrates) 14–16 million[e] (emerald ash borer)	1.5 million[d] (Dutch elm disease) 73.34 million[e] (potato wart fungus) 1,000,000[e] (BSE)
Germany (in €)	103 million[f] (8 species)	60.2 million[f] (6 species)	5 million[f] (Dutch elm disease)
New Zealand (in $NZ)	100 million[g]	270 million[h] (vertebrates) 2 billion[i] (invertebrates)	
United States (in $US)	34.5 billion[j]	59.4 billion[j]	39.7 billion[j]

[a] Sinden et al. (2004).
[b] Bomford and Hart (2002).
[c] McLeod (2004).
[d] Colautti et al. (2006).
[e] One-time event, Colautti et al. (2006).
[f] Reinhardt et al. (2003).
[g] Williams and Timmins (2002)
[h] Clout (2002).
[i] Barlow and Goldson (2002).
[j] Pimentel, Zuniga, and Morrison (2005).

of large magnitude. Further, when pest damages affect the market for a good, as with many agricultural products, the average price of the product lost to the pest is less than the observed market price. If market price of the product is used as a proxy for the value of each unit lost to pest damage, then an upward bias in total damages is introduced, although conservative assumptions about damages per pest may offset this to some extent. More generally, if damages are nonlinear, then policy evaluation can be improved by a more accurate assessment of the damage function. In spite of these difficulties, the national values do provide some information about the economic impacts of invasive species. The large magnitudes indicate that terrestrial invasive species impose significant social costs. The remainder of the paper reviews how the economics literature models invasive species problems and the insights and policy conclusions that are obtained.

The Economics of Invasive Species Control

In the simplest intertemporal models of invasive species control, the state of the invasion, or its capital stock, is defined by its size. This may be the population or biomass of the invasive species, or it may be the area contained within the frontal boundary of the invasion, depending on the context of the problem. The growth and spread of the invasive species is governed by a biologically determined transition equation. Control involves reducing the size of the invasion by chemical, biological, mechanical, manual, or other means. It is useful to think about the economics of control in two stages. First, for each possible reduction in the size of the invasion, a vector of inputs is selected to achieve the desired level of control in the least-cost manner, given input prices.[3] This

[3] Lichtenberg and Zilberman (1986) provide a useful discussion of econometric considerations in the estimation of production-based models of pest control.

may or may not involve integrated management. The resulting control cost function depends on both the amount controlled (the reduction in the size of the invasion) and on the size of the invasion being controlled.[4] Given the control cost function, the manager chooses control at each point in time to minimize expected discounted control costs and invasion damages over time, subject to the biological transition function for the invasion. The resulting dynamic optimization problem can be analyzed using dynamic programming or optimal control.[5] A key point that emerges from such models is that the value of an additional unit of control is not simply the additional damages avoided today, but the discounted future sum of damages avoided, compounded by the growth in the invasion that would result from the unit of invasive species being controlled.

A precursor to the recent literature on the economics of invasive species is Jaquette's (1972) analysis of the existence of an optimal policy and the monotonicity of the optimal state transition in a finite-horizon, discrete-time biological population control model. Regev, Gutierrez, and Feder (1976) point out that a number of factors lead individual pest control decisions to diverge from the social optimum. These include interseasonal dynamics, biological relationships with other pests and predators, pest resistance, environmental and health effects of pesticides, and neighborhood externalities. They derive the first-order necessary conditions for an interior optimal solution and compare the resulting steady states to the private (myopic) optima. They apply their model to alfalfa weevil control and calculate a steady state shadow cost of 2.3 cents per 1,000 emergent adults per acre. Shoemaker (1981) reviews early applications of dynamic programming to the problem of pest management, but these tend to focus on issues such as pesticide resistance and intraseasonal management.

Olson and Roy (2003) use a dynamic programming approach to characterize the optimal control of a biological invasion when both the biological growth function and the control cost function are allowed to exhibit non-convexities. They show that if the marginal costs of control are more sensitive to changes in the invasion size than to changes in control, the optimal policy may involve periodic control. In addition, they characterize conditions under which eradication, maintenance control, and no control are economically efficient.

Wilman (1996) and Knowler and Barbier (2000) examine models with an invasive predator whose prey is harvested for its economic value. Eiswerth and Johnson (2002) develop an optimal control model of invasive species management where growth in the invasive species follows a logistic growth function. They derive the first-order necessary conditions and study comparative statics of the resulting steady state with respect to parameters of the model. They provide a numerical illustration of their results based on invasive weeds on rangeland in the western United States. Barbier and Shogren (2004) examine an endogenous growth model in which the stock of invasive species is a function of the aggregate stock of capital in the economy. Invasive species are analogous to a pollution externality induced by the capital stock. They analyze the effect of this "biological pollution" on the balanced growth path when invasives affect only production and when invasives affect both production and welfare. A key assumption is that the stock of invasives is completely determined by the aggregate capital stock. It is difficult to see how this relationship might exist even as an approximation to any practical situation.

Environmental disturbances such as weather events can either accelerate or slow the spread of invasive species. A 1938 hurricane blew the gypsy moth across a barrier zone that had been established along the Hudson River to slow its spread (Animal and Plant Health Inspection Service 1985). Hurricanes in 2004 and 2005 contributed

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[4] For example, historical attempts to eradicate invasive species indicate that it may cost as much to remove the last 1 to 10 percent of an invasion as it does to control the initial 90 to 99 percent (Myers, Savoie, and van Randen 1998).

[5] In these models one complication arises from the fact that standard sufficiency conditions are typically not satisfied. To see why, consider the simplest possible case where the control cost and damage functions are both convex. Then the shadow price on the invasive species stock is negative, reflecting the fact that it imposes a cost. For this case, the standard Mangasarian (1966) sufficiency condition for optimal control requires the growth function to be convex. However, since all biological invasions are bounded at some point, their growth function must be concave over some interval. In fact, following standard convention from the literature on renewable resources, it is common to assume that the growth function is globally concave. The literature typically derives first-order necessary conditions for an interior solution and analyzes the economic implications under the explicit or implicit assumption that the solution is optimal [e.g., Regev, Gutierrez, and Feder (1976), Zivin, Hueth, and Zilberman (2000), and Eiswerth and Johnson (2002)].

to the spread of citrus canker in Florida and forced the USDA to abandon an eradication program begun in 1996 (Florida Department of Agriculture and Consumer Services 2006). There is debate about the extent to which climatic conditions assisted the eradication of screwworm in the United States (Readshaw 1986, Krafsur et al. 1986). Variations in climate also have an important influence over the growth of invasive species populations. The successful eradication of nutria from the United Kingdom was aided by an above average number of harsh winters that slowed reproduction and increased juvenile mortality (Gosling and Baker 1989). Olson and Roy (2002) examine the economics of controlling a biological invasion whose natural growth and spread is nonconvex and subject to environmental disturbances. They characterize conditions under which it is optimal to eradicate the invasive species and conditions under which eradication is not optimal. The disturbance that produces the slowest expansion in the invasion plays a critical role. Eiswerth and van Kooten (2002) examine a model of an invasive species infestation that has four possible states: minimal, moderate, high, and very high. They use an expert judgment questionnaire to develop fuzzy membership functions for each of the states and to construct a state transition probability matrix. Using stochastic dynamic programming, they analyze the control of yellow starthistle in California and compare the efficiency of five different management regimes. The policy choice varies depending on productivity and discount parameters, with expected net returns ranging from $292 to $2,411 per acre.

Invasive species problems are often characterized by important spatial considerations. Brown, Lynch, and Zilberman (2002) examine a static model of the spatial control of an invasive species emanating from a source. Control involves variable crop inputs, a barrier zone to reduce transmission, and source control. The model is applied to analyze the control of Pierce's disease and its transmission by sharpshooter leafhoppers in California wine grapes. They find the optimal barrier width and grower profit to be sensitive to barrier effectiveness with profit per acre ranging from $3,054 to $5,201 as barrier effectiveness increases from 0 to 1.

Another important spatial issue is the possibility that reduced competition will encourage inward migration of the invasive species from areas bordering the control zone. Huffaker, Bhat, and Lenhart (1992) develop a continuous time dynamic model of a nuisance species that occupies two adjacent parcels of land. Control on one parcel decreases population pressure relative to the environmental carrying capacity. This can increase dispersal from the adjacent parcel. The authors derive the first-order necessary conditions and characterize the singular solution under specific functional forms. Sensitivity analysis with respect to parameters is done using numerical simulation. For the values examined, increases in the dispersal rate result in less control and a greater population in both areas.

Barrier zones have been used to slow or prevent the spatial spread of several insect species including the gypsy moth, screwworm, boll weevil, and Africanized honey bee. Sharov (2004) develops economic models of a barrier zone designed to slow the spatial spread of an invasive species. Sharov first considers uniform spread on an infinite habitat strip, where damages are proportional to the area invaded, or $D \times v \times t$, where D is marginal damage, v is the rate of spread, and t is time. A barrier that reduces the rate of spread by an amount Δv lowers damages by $\delta D \Delta v$, where δ is the discount rate. Hence, the optimal policy equates the marginal cost of slowing the spread to its marginal value, δD. Sharov also discusses how barrier zones can be modeled when spread occurs in a limited area. It is possible for multiple local optima to exist, and numerical methods may be required to determine the global optimum. The management of gypsy moth spread in North America is used to illustrate how the methods can be applied in practical situations [see also Sharov and Leibhold (1998) and Sharov, Leibhold, and Roberts (1998)].

For some species, management depends on life history traits of the species. Buhle, Margolis, and Ruesink (2005) analyze cost-effective control in two- or three-stage matrix population models. Using population elasticity analysis, they consider the combination of life-stage interventions that minimize the total cost of halting the population growth of an established invasive species.

Adaptation by species can be viewed as resulting from optimal responses to environmental conditions over time. Guitterez and Regev (2005) and Finnoff and Tschirhart (2005) consider the analogy between economic and ecological optimization and its implications for invasive species. In Guitterez and Regev (2005), species choose biological consumption to maximize adaptability of individuals over an infinite time horizon while still preserving the resource base. They discuss the implications of their model for the cowpea and cabbage aphids and the cotton boll weevil. Finnoff and Tschirhart (2005) focus on plant species and assume that each plant maximizes net energy intake per unit of time. The ecosystem reaches a steady state when the available space is filled, each plant maximizes its net energy, and net energies are zero. Simulations illustrate how the model can be used to make simple predictions about species composition and vulnerability to invasion under different management regimes.

In recent years, biological control has received increased attention as a policy instrument for invasive species management. McConnachie et al. (2003, Table 1) review 10 benefit-cost studies of successful biological control programs, including four insect pests, four terrestrial weeds, and two aquatic weeds. For terrestrials, the benefit-cost ratios range from 1.9:1 to 24:1. Van Wilgen et al. (2004) estimate the costs and benefits of biocontrol of six invasive weed species in South Africa, where biocontrol has been practiced since 1910. They estimate benefit-cost ratios ranging from 8:1 for red sesbania to 709:1 for jointed cactus. The estimates are sensitive to assumptions about the rate of spread with a 3 percent decrease in benefits for each one percent decrease in the rate of spread. Biological control programs are not without risks, however. Control species may become invasive themselves and adversely impact non-target species. For example, feral cats introduced on many islands to control rats have proved so damaging to island ecology that they are now subject to eradication programs (Nogales et al. 2004). The cane toad (*Bufo marinus*) is another example of biological control gone awry. Introduced in Australia in 1935 as a biological control for scarab beetles, pests of sugar cane, they failed as a control and have subsequently become a significant ecological pest (McLeod 2004).

The Economics of Invasive Species Prevention

Prevention is the second primary policy instrument that can be used to mitigate the damages caused by invasive species. While the goal of control is to reduce or eliminate the damages caused by invasive species, the goal of prevention is to avoid damages and/or control costs. The two policies are necessarily interdependent. The optimal strategy for prevention necessarily depends on the social costs of an invasion, should it occur. Likewise, the optimal strategy for control must account for the possibility that an invasion may recur. Olson and Roy (2005) examine a static model of the trade-off between prevention and control under uncertainty. An established invasion is managed through control. Invasive species introductions are a random variable, but can be reduced through prevention. The objective is to minimize the expected costs of prevention, control, and damages. All costs and damages are convex. The optimal control is increasing in the invasion size. If marginal control costs are more sensitive to changes in control than to changes in the invasion size, then both optimal prevention and the optimal post-control invasion size increase with the initial invasion size. If marginal control costs are more sensitive to changes in the invasion size, then both optimal prevention and the optimal post-control invasion size decrease with the initial invasion size. Both optimal control and optimal prevention are increasing in the invasion growth rate. The results also show how prevention and control vary with a shift in the distribution of invasive species introductions that satisfies monotone likelihood ratio (MLR) dominance. When absolute aversion to risk decreases in the introduction size, prevention increases as the distribution shifts upward, while control increases if the elasticity of marginal damage is decreasing in the introduction size. In somewhat related work, Leung et al. (2005) use a stylized model with specific functional forms to analyze the effect of parameter changes on prevention and control.

Sumner (2003) and Sumner, Bervijillo, and Jarvis (2005) point out that invasive species policies such as border control and eradication programs have attributes of public goods for affected consumers and producers, in that they are often non-rival and non-excludable. Sumner (2003) suggests that funding invasive species programs through commodity levies has an advantage over the use of general tax revenue in that levies transfer much of the cost of invasive species policy to the beneficiaries.

Horan et al. (2002) examine a model of invasive species prevention. In their model there are n independent potential invasion pathways. Invasion is treated as a Bernoulli event: an invasion either occurs or it does not. The optimal policy for each pathway balances the marginal cost of prevention against the expected marginal damages prevented, where damages are prevented only if an invasion does not occur through another pathway. They examine different decision models, including expected utility and a model of decision making under ignorance.

In reality, private agents act to reduce private damages caused by invasive species, and if government agencies fail to recognize this there may be a misallocation of resources. In a set of papers with a common theme, Finnoff and Shogren (2004) and Finnoff et al. (2005a, 2005b) examine how interactions between public managers, myopic private agents, risk aversion, and the perceived state of an invasion affect the allocation of resources for prevention and control and find that the results are sensitive to initial conditions.

Heikkila and Peltola (2004) examine the cost of maintaining the Finnish protected zone for the Colorado potato beetle using measures to prevent an invasion and control to eradicate invasions that occur. They estimate deterministic prevention costs of €350,000 and eradication costs of €946,931.

Intentional introductions of non-native species raise a distinct set of economic and policy issues. Such introductions will occur only if some agent expects a benefit from the species. The potential for limited liability exists if the releasing agent is not fully responsible for negative consequences that may arise if the species turns out to be invasive. This creates incentive problems for the design of effective policy. Thomas and Randall (2000) examine this problem in a principal-agent setting. In their model, the release of a non-indigenous species generates a private benefit X for the agent and possibly a large (yet reversible) social loss S where $S > X$ and the probability of S, $P(S) = \theta$. Success in revoking the negative consequences of a release is random with probability r. Maintaining the option to revoke S and the act of revoking S involve expenditures of $c(r, \theta)$. When the incentives of society and individuals align the solution is a value of r, or equivalently, $c(r, \theta)$, that balances the marginal cost of revoking S against the expected social loss of the introduction. In reality, releasing agents do not bear all of the social costs of invasive species, even when releases are intentional. Thomas and Randall (2000) suppose that the agent selects the level of revocability, but faces limited *ex post* liability in the event a large social loss occurs. They then consider the decision problem of the agent if the principal requires an assurance bond to cover potential losses, should they occur. In this case the agent chooses r to equate the marginal cost of revocability to the minimum of the expected social loss and the private loss if the bond is forfeited. Shogren, Herriges, and Govindasamy (1993) discuss the advantages and disadvantages of assurance bonds as a means of reducing environmental externalities. They draw on previous work on labor economics that examines the limits of bonds as a mechanism to prevent worker shirking and they identify three difficulties associated with environmental bonds. These are (i) regulator moral hazard where the principal may impose liability without cause, (ii) liquidity constraints that prevent the agent from posting the required bond, and (iii) legal restrictions on contracts that provide avenues for an agent to challenge the loss of a bond. All three of these are likely to be issues for using bonds as a mechanism to reduce invasive species introductions.

The Economics of International Trade and Invasive Species

Many invasive species introductions occur as a result of trade. Not surprisingly, public policy aimed at reducing the potential risk and scale of biological invasion has targeted regulation of international trade as one of the primary means of preventing domestic control costs as well as the ecological and economic damages that arise when alien species establish and expand over time. International trade agreements recognize that it is important for individual countries to "have the right to take sanitary and phytosanitary measures necessary for the protection of human, animal or plant life or health" [Article 2, World Trade Organization (WTO) Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement)]. The main objective of non-tariff or technical barriers to trade is to correct externalities or market inefficiencies caused by invasive species in the production, distribution, and consumption of goods.

Roberts, Josling, and Orden (1999) and Roberts (1999) propose that technical trade barriers be classified by policy instrument, scope, and regulatory goal. They use the proposed classification scheme to analyze the results of a 1996 USDA survey of over 300 foreign technical barriers to U.S. agricultural exports. Roberts, Josling, and Orden (1999) also examine partial equilibrium models that can be used to study the effects of technical trade barriers. Beghin and Bureau (2001) survey methods to quantify the impact of SPS and other non-tariff trade barriers on market equilibrium, trade flows, economic efficiency, and welfare. They review the price wedge method, inventory, survey, and gravity based approaches, risk assessment based cost-benefit analysis, microeconomic based approaches, and multi-market models. Studies that utilize the methods are surveyed and the practical validity of each method is discussed. Smith (2003) provides a useful summary of the SPS compliance requirements of the WTO Agreement and examines their role in a number of prominent disputes, including the Australian ban on Canadian salmon, the Japanese ban on U.S. apples, foot-and-mouth disease, and exotic Newcastle disease. Mumford (2002) provides a general overview of the economic issues related to quarantine policy and trade. He discusses the relation between increased trade and quarantine threats, the impacts of quarantine on trade, international agreements related to quarantine, and criteria to evaluate quarantine policy, including the appropriate level of protection, effectiveness (in a technical sense), economic efficiency, distributional concerns, and cost recovery. Lynch (2002) develops a model of import bans and subsidies for control in the exporting country and applies it to the Mexican fruit fly problem and trade between the United States, Mexico, and Central America. She points out that, by reducing demand and hence prices, SPS regulations can lead to less pest control by foreign producers. This can increase the risk of pest infestation through other pathways not affected by SPS regulations. Costello and McAusland (2003) examine the link between trade, protectionism, and invasive species damage. They examine circumstances under which more protection reduces invasive species damage. They also make the important point that changes in trade restrictions alter the mix of domestic outputs. If freer trade leads to a shift away from outputs susceptible to pest damage, and if this shift is large enough to offset the increase in invasions that accompany greater trade volumes, then freer trade can lower invasive species damages, contrary to intuition.

McAusland and Costello (2004) consider a static model of the use of tariffs and inspections to reduce trade-induced invasive species damages. When inspections are costly, the optimal policy has the following characteristics: the optimal inspection intensity never detects all incoming pests (it is assumed that the marginal productivity of inspections declines to zero as the detection rate approaches one), and the optimal tariff recovers the cost of inspection plus the damage expected from invasive species imported on goods that are not rejected by the imperfect inspections. Because the inspection costs are recovered by the tariff, inspections are optimally chosen to minimize the cost of consumer goods. Inspection intensity increases with the damage parameter, and increases and then decreases with the invasion rate; for high invasion rates it is not worth spending money to confirm that goods are contaminated. The optimal tariff increases with the damage parameter and the invasion rate unless better inspections lead to a decline in expected damage that more than offsets the increased inspection cost. McAusland and Costello (2004) also examine two variations of the model. The first allows exporting firms to treat exports to reduce the invasion rate, while the second considers a two-period model. For the latter, it turns out that the optimal dynamic inspection level is at least as high as the static level, while the optimal dynamic tariff can be more or less than the optimal static tariff.

Political economy considerations and interest group lobbying have played an important role in trade policy toward goods impacted by invasive species. Romano and Orden (1997) discuss the political economy of U.S. import restrictions on nursery stock and ornamental plants. Roberts and Krissoff (2004) examine the use of SPS trade barriers in international horticultural markets, the extent to which countries have harmonized their standards under the WTO agreement, and the status of 33 complaints related to SPS restrictions on horticultural products filed during 1995–2002.

Margolis, Shogren, and Fischer (2005) incorporate an invasive species externality into the Grossman and Helpman (1994) political economy model of trade. They show that the equilibrium tariff with an externality is simply the optimal tariff (marginal damages) plus the equilibrium tariff found by Grossman and Helpman for the case where there is no externality. The result simply restates the Grossman-Helpman result in a way that accounts for marginal damages; however, it does point out the potential difficulty in distinguishing disguised interest group protectionism from legitimate SPS trade measures.

Balancing protection from invasive species against the costs of regulation to consumers and producers poses significant challenges to regulators, particularly because decisions need to be made in an environment where there is uncertainty about the consequences of different policy alternatives. A sound empirical approach is needed. Sumner and Lee (1997) illustrate how the effects of SPS rules on export supply and import demand functions might be incorporated into empirical trade models. Glauber and Narrod (2001, 2003) provide a critical examination of the quarantine program designed to prevent the spread of karnal bunt. In the first paper they argue that a failure to adequately integrate risk assessment with economic analysis led to suboptimal regulations that cost producers, consumers, and taxpayers more than $350 million per year, when losses due to restrictions on seed development are included. James and Anderson (1998) conduct a partial equilibrium analysis of Australian quarantine policy to protect domestic banana production from pests and diseases. They consider both a fixed and ad valorem marketing margin, and for the latter they consider three different supply elasticities. In all four scenarios the gains in consumer surplus outweigh the costs to producers if the import ban is lifted, even though in two of the scenarios domestic production is not competitive under free trade and would be eliminated. Likewise, Orden, Narrod and Glauber (2000) report results by Orden and Romano (1996) that suggest that the U.S. trade ban on Haas avocados from Mexico imposed a net welfare loss on the United States, even in the worst-case scenario where lifting the trade ban would be certain to result in a pest infestation. In contrast, Hoddle, Jetter, and Morse (2003) estimate a welfare loss of $4.6–7.6 million if an avocado pest establishes in California and raises industry costs by 3.6 percent. The regulatory process surrounding the Haas avocado case is described in detail in Roberts and Orden (1997). (Recently, U.S. trade policy toward Haas avocados has loosened, although some restrictions remain.) Calvin and Krissoff (1998) investigate the effects of the Japanese tariff and phytosanitary standards for Fuji apples on imports from the United States. Using a tariff-rate equivalent of the import ban, they estimate that an average yield loss of 30 percent or more would be required to eliminate the welfare gains from free trade. In 2002 the United States requested that the WTO review Japan's phytosanitary protocol, and in 2005 Japan issued a new protocol that eliminated standards designed to prevent fire blight, but maintained standards to prevent codling moth. Using the same methodology as their earlier paper, Calvin and Krissoff (2005) measure the cost of the fire blight protocol and the expected volume and value of imports that would have occurred under the new protocol during the period 1998–2004. They estimate a change in the value of Fuji apple imports ranging from $8.9 million to $228 million, depending on the year. This points out that there may be substantial variation in market conditions over time and that a single year may not provide an accurate representation of the true economic impact of changes in phytosanitary standards.

Acquaye et al. (2005) focus on the consequences for producers and consumers of policies to reduce invasive species damages in the face of preexisting agricultural polices for commodities. They point out that the effects of invasive species policy on supply are similar to the effects of an improvement in production technology. A case study of the economic consequences of citrus canker illustrates that it is important to recognize existing agricultural policies when calculating the costs of invasive species. In their example, the costs of citrus canker are overestimated by $10 million annually, or by 7 percent, if the effects of existing tariffs and subsidies for frozen concentrated orange juice are ignored.

Economic Models of Invasive Plants

Wu (2001) develops a dynamic model of weed control with the weed seed bank as the state variable. The manager's problem is to choose herbicide applications in each period to maximize the discounted sum of net benefits over time. Wu analyzes the first-order necessary conditions and imposes functional forms that allow a closed-form solution for optimal weed and seed densities and herbicide applications. A numerical example based on weed control in Iowa corn production is provided. Odom et al. (2003) develop a discrete-choice, dynamic programming model of Scotch broom control in Australia. Economic benefits depend on weed density. In their model, both the weed seed bank and weed density are included as state variables. Five control instruments are available, with feasible values of 0 or 1 (each control instrument is either used or not). They are as follows: exclude tourists, remove weeds manually, apply herbicides, control wild pigs, and biological control. The dynamic programming problem is solved numerically to obtain the optimal mix of control policies as a function of weed and seed density. At very low weed and seed densities, the optimal policy is to do nothing, but over a broad range of higher densities the optimal policy involves a mix of manual removal, herbicides, and biological control. Jetter et al. (2003) analyze the expected benefits and costs of a statewide biological control program to manage the rangeland weed, yellow starthistle, in California. They calculate a break-even probability of success that determines when the expected benefits of a control program exceed the costs. The probability depends on assumptions about the benefits from rangeland restoration. As benefits increase from \$1 per acre for both infested and susceptible lands to \$50 per acre for susceptible land, the break-even probability declines from 21 percent to 0.6 percent. Rangeland restoration costs under a successful biological control program are estimated to be 25 percent less than under a chemical control program.

Eiswerth et al. (2005) use input-output analysis to estimate the effect of nonindigenous invasive weeds on the economy of Nevada. The rate at which invasive weeds reduce wildlife recreation expenditures is taken from Leitch, Leistritz, and Bangsund (1996). For a range of parameter values they estimate a total impact on Nevada of –\$5.9 million to –\$22.3 million per year. Higgins et al. (1997) conduct a dynamic simulation of an ecological-economic model of alien plant control in a mountain fynbos ecosystem in South Africa. Under the simulation the cost of a proactive clearing policy ranges from 0.6 to 4.76 percent of the economic value of ecosystem services, but increases the value of these services between 138 and 149 percent, depending on the assumptions of the model.

DiTomaso (2000) identifies a number of noxious weeds that have significant impacts on rangelands and reviews some of the literature on the economic and ecological costs of these weeds. He then discusses options for noxious weed management including prevention, eradication, and control. Options for control include mechanical, cultural, biological, chemical, and integrated management.

Knowler and Barbier (2005) develop a model of a commercial nursery industry that imports ornamental plants that pose a risk of becoming invasive. In the private market equilibrium, nurseries do not internalize the social costs of potential invasions, and the industry expands until the last nursery to enter the industry earns zero profit. Potential invasions are modeled using a hazard function that defines the probability that an invasion occurs at *t*, given that it has not occurred prior to *t*. The social optimum balances industry profit against the expected social cost of managing a potential invasion. Consequently, the industry size is smaller in the social optimum than the private market equilibrium. In principle, a Pigouvian tax can be used to implement the social optimum. An illustration of the model based on saltcedar is developed. The numerical results vary significantly depending on assumptions about the dependence of the hazard function on the number of firms in the industry.

Economic Models of Invasive Animals

In the United States invasive wildlife species cause damage in every state and all U.S. territories. Bergman, Chandler, and Locklear (2002) summarize, by species, the geographic locations that requested assistance from USDA's Animal and Plant Health Inspection Service Wildlife Services Program in order to alleviate damage caused by vertebrate wildlife species from 1990 to 1997. More than 45 vertebrate invasive species were verified as a cause of economic damage.

Many wildlife populations cause damage to agricultural or environmental systems but are also valued for commercial or recreational reasons or for their contribution to biological diversity. Examples include fur-bearing animals, deer, and feral pigs. Whether a species is a pest or a resource may depend on social, economic, regulatory, and environmental circumstances. Zivin, Hueth, and Zilberman (2000) examine a bioeconomic model of a species that causes pest damage but has value to hunters. The landowner can control the species through trapping or the sale of hunting rights. They characterize first-order necessary conditions that maximize intertemporal welfare for three possible steady state outcomes: trapping and hunting, trapping only, and hunting only. In the first two cases the shadow value of the species is negative, while in the third case the shadow value of the resource may be positive to reflect the fact that the marginal value to hunters may exceed marginal pest damages. The model is illustrated with a case study of feral pigs in California rangeland. Skonhoft and Olaussen (2005) consider a spatial version of this problem, where there is migration between two locations that are managed separately using only harvest from hunting. The objective is to maximize static, steady state welfare, rather than the true dynamic optimum. The static, steady state welfare maximum is achieved by equating the value of marginal growth in each location to marginal pest damage.

Rats are commonly believed to be the world's most widespread invasive mammal, with the greatest economic impact. They have had enormous ecological impact on islands and are responsible for more island extinctions of birds, snakes, and lizards than any other predators (Matthews 2004). They cause enormous damage to agriculture. Singleton (2003) estimates that rodents in Asia cause losses to rice production of 5–10 percent per annum, enough food to feed 180 million people annually, while Pimentel et al. (2000) value the economic damage caused by rats in the United States at $19 billion per year. Stenseth et al. (2003) discuss how bioeconomic modeling might be used to improve rodent management, and they provide useful background information for five different pest rodents. Skonhoft et al. (forthcoming) investigate the economics of control of the multimammate rat, an African pest rodent that causes significant damage to maize production. The ecological model is a density-dependent matrix population model that incorporates the effect of stochastic rainfall on rat population growth. The economic model is based on village-level data from Tanzania. Rats are controlled by poison, and social costs are measured by the discounted sum of control and damage costs. The model is too complex for analytical solutions so policy options are evaluated numerically using the median social cost. The results suggest that the most cost-effective policy is to control 3–4 months every year, particularly at the end of the dry season/beginning of the rainy season. The current practice of applying poison when heavy rodent damage is observed appears to be associated with a substantial reduction in the welfare of maize-producing farmers.

Economic Models of Plant and Animal Disease

Animal and plant diseases represent microbial forms of invasive species. Horan and Wolf (2005) develop a two-state, linear control model of management of an infectious wildlife disease. The two state variables are the wildlife biomass and the fraction of the population infected. The two controls are harvesting and feeding. Harvests cannot differentiate between infected and healthy animals and affect only the dynamics of wildlife biomass, while feeding influences both wildlife growth, through nutrition, and disease transmission, by reducing disease-related mortality and because wildlife congregate around feeding stations. Harvested wildlife has economic value, while infected animals impose damages through their potential to transmit the disease to livestock or other commercially valuable species. Horan and Wolf (2005) characterize the double singular solution and solve a numerical example to illustrate the control of bovine tuberculosis in Michigan deer populations. They find that because harvested deer are valuable, eradication of the disease is not likely to be economically efficient unless the state of Michigan faces fixed costs associated with livestock testing or trade restrictions. Horan et al. (2005) discuss how the analysis can be extended to capture spatial interactions between wildlife populations and disease transmission.

One animal disease that has received significant attention is foot-and-mouth disease (FMD). Paarlberg and Lee (1998) develop a framework to determine the optimal tariff on imports that carry a risk of disease, such as livestock imports and FMD. From the perspective of the importing country, the optimal tariff should account for the risk of importing FMD and the resulting loss in domestic output. A numerical, partial equilibrium model of U.S. beef imports is used to compare tariffs with and without FMD risk. The findings are sensitive to assumptions about risk and output loss. Thompson et al. (2002) estimate the economic impact of the 2001 FMD outbreak in the United Kingdom in which 4 million animals were slaughtered under disease control measures. They calculate a total cost of £3.1 billion. Because farmers were compensated for slaughtered animals, these costs were largely borne by the public, and the cost to agricultural producers was estimated to be £355 million. Ekboir (1999) and Ekboir, Jarvis, and Bervejillo (2003) simulate the effects of a potential FMD outbreak in California under different scenarios about disease transmission and policy response. They estimate total costs ranging from $6.8 to $13.5 billion, including direct costs of sacrificing animals, cleaning, and disinfecting, plus indirect costs of disruptions to trade. This estimate does not consider potential effects on tourism and wildlife, meat processors, and distributors, or environmental consequences from disposing of diseased animals. Paarlberg, Lee, and Seitzinger (2002) estimate that an FMD outbreak in the United States would result in a reduction in farm revenue of $14 billion. Paarlberg, Lee, and Seitzinger (2003) suggest that the welfare effects for producers and consumers should be decomposed. Slaughter and quarantine measures will be imposed only on some producers, and the remaining producers will respond to changes in market prices. In addition, some consumers will reduce or stop consuming beef while others will not, with correspondingly different welfare effects. Paarlberg, Lee and Seitzinger (2005) survey these articles and others that examine the economic impacts of livestock disease. Another recent review of this literature is provided by Pritchett, Thilmany, and Johnson (2005). They classify models based on the level of the analysis: consumer, producer, meat processors, input suppliers, supporting activities, and marketing channels, and studies of regional, national, and international scope.

Plant diseases such as soybean rust, citrus canker, and karnal bunt pose potentially serious threats to U.S. agricultural production. Citrus canker has been detected in Florida on three occasions. Twice it has been declared eradicated—in 1933 following detection in 1910—and in 1994 following detection in 1986. It was detected again near Miami in 1995, and a third eradication campaign was begun (Florida Department of Agriculture and Consumer Services 2006). Zansler, Spreen, and Muraro (2005) estimate the discounted sum of net benefits of Florida's citrus canker eradication program from 1996 onward at $2.26 billion, under the assumption that no reinfestation occurs. Unfortunately, the USDA now believes that eradication is no longer feasible due to potential spread of the disease by hurricanes in 2004 and 2005, and a new citrus canker management plan is being formulated (Conner 2006). Jetter, Civerolo, and Sumner (2003) analyze the economic effects of a potential citrus canker invasion in California. They consider welfare effects under different policy scenarios involving urban and/or commercial eradication programs with or without compensation, as well as potential trade embargo considerations.

Conclusion

The literature on the economics of invasive species management has developed rapidly in recent years, but there is much room for further work. Uncertainty, spatial modeling, prevention, trade, and conflict between private and public incentives are all areas where there is a need for more sophisticated analyses. There is also a significant need for the development of better data and techniques to support more accurate empirical assessments of invasive species damages and control costs. Work in these areas should help improve invasive species policy and achieve a more effective use of resources.

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