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Integration-Valuation Nexus in Invasive Species Policy

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This paper reviews recent work examining two topics of economic research vital for invasive species policy—integration and valuation. Integration requires bioeconomic models that blend invasive biology with economic circumstances and the feedback loops between the two systems. Valuation requires nonmarket valuation associated with human and environmental damages posed by invasive species. We argue for a second-level of integration in invasive species economics—valuation based on integration models. Policy prescriptions based on integration models need valuation work; valuation surveys need integration models—the two are complements. Valuation could be enhanced with integration in mind; integration could be made better with valuation in mind. An example from blending the two research areas is presented and its merits demonstrated.

Key Words: invasive species, integrated economic-ecological modeling, nonmarket valuation

Invasive species questions have long challenged policymakers in many countries and organizations around the globe. In the United States, for instance, the number of non-native species is estimated to be as high as 50,000 (Pimentel et al. 2000). Of these, some 5,000 have become established, and of those about 500 have become invasive (U.S. Congress, Office of Technology Assessment 1993). An official *invader* is defined as a species that both is non-native and triggers costs to human or ecosystem health that outweigh any attendant benefits. Invasive species contribute to biodiversity loss and can cause large ecosystem changes (Mack et al. 2000, Mooney and Hobbs 2000).

The 2001 National Invasive Species Management Plan highlighted the urgent need for more rigorous and comprehensive frameworks for invasive species policy so that prevention and control strategies can be targeted appropriately (National Invasive Species Council 2001, Committee on Environment and Natural Resources 1999, Federal Interagency Committee for the Management of Noxious and Exotic Weeds 1998). Yet invasive species policy continues to be developed using inadequate quantitative scientific guidance, including a dearth of economic reasoning (see, e.g., Barbier 2001). Economic reasoning matters for invasive species policy because people are one of the main vectors of their spread around the globe. Historically, however, invasive species policy has been dominated by the biological sciences and related disciplines. This trend is reversing, however, as over the last decade economists have become more involved in understanding how basic economic principles such as relative prices, incentives, trade, and regulation work with biological circumstances to define the risks of damages caused by invasive species (see Perrings, Williamson, and Dalmazzone 2000). By including economics into policy debates, decision makers come closer to meeting President Clinton's Executive Order 13112, pledging decisions that work "to minimize the economic, ecological,

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and human health impacts that invasive species cause."

This paper reviews our recent work examining two economic research topics vital for better invasive species policy—integration and valuation. By *integration* we mean bioeconomic models that blend invasive biology with economic circumstances and the feedback loops between the two systems so that we can better assess the risks posed by invaders. By *valuation* we mean the nonmarket valuation associated with reduced human and environmental damages posed by invasive species so that we can better understand the net benefits of prevention, control, and eradication efforts. We also argue for a second level of integration in invasive species economics—valuation based on integration models. The idea we stress is that integration models need valuation work and valuation surveys need integration models—they are complements. Granted, comparative advantages exist in allowing *modelers* to model and *valuationists* to value; we still contend that working together can serve to improve both areas of research. Valuation could be enhanced with integration in mind; integration could be made better with valuation in mind. We focus first on integration and then valuation, and then provide an example of the blending of the two research areas. Finally, we offer some brief remarks on two directions of research in invasive species economics needing more attention—trade/growth and incentive design.

Integration

The key policy question is to determine the risks posed by invasive species. Models defining these risks, however, have for the most part been dominated by the biological sciences. At first glance this makes sense because invasive species are a biological question. Traditionally, economists have stepped in at the end to evaluate the monetary damages associated with different alerted states of nature. This is the classic "damage function" (DF) approach used to examine invasive species (see Freeman 1993, Perrings, Williamson, and Dalmazzone 2000). The DF approach assumes that the economic system and the ecosystem affect each other in a one-sided way. A change in the economic system is viewed as changing only the pressure on the ecosystem (e.g., Vitoussek et al. 1997), *or* a change in the ecosystem is viewed as changing only the economic system (e.g., Daily 1997). The DF approach therefore does not address the idea of coevolution—the two-way interactions between human and natural systems (see Crocker and Tschirhart 1992).

Though the broad outlines of ecological-economic system reciprocities have been acknowledged, the acknowledgements usually lump together key parameters of one or both systems. The lumping hinders predictions and evaluations of which ecosystem services are to be usurped by humans, which waste flows are to be allowed to enter ecosystems, and which ecosystem biota and physical attributes are to be maintained. One can neither discriminate among the key parameters nor identify exactly how they enter the systems. Arguably the contributions of natural environments to human well-being depend as much upon the complex details of ecological and economic states as upon the gross relationships between ecological systems and economic systems. Those analyses capable of admitting both the empirical detail and the jointness of ecological and economic systems are few (see Ayres and Kneese 1969, Crocker and Tschirhart 1992). These general equilibrium treatments work only with steady states. More recently, a few authors have set the structural details of jointness in a dynamic context (e.g., Swallow 1996, Perrings 1998, Sohngen and Mendelsohn 1998, Brown and Layton 2001).

When an ecosystem changes, people change their behavior, which in turn reshapes the ecosystem, and so on. Ecosystem changes alter human productivity in the economic system. People recognize the change in their productivity when using the ecosystem, and they adapt to this change, either by adapting the environment or by adapting to the environment. When people adapt, they alter the pressure they put on the ecosystem, leading to further changes in the ecosystem. The cycle continues. Recent work has addressed whether an explicit accounting of the specifics of these feedback links between the two systems yields different policy-relevant results than does assuming that no joint determination occurs. Consider three illustrations of three modeling approaches that create an explicit analytical framework to integrate and account for feedbacks.

The first illustration is the bioeconomic endogenous risk-stochastic dynamic programming (SDP) model developed in Leung et al. (2002). They constructed a framework that incorporates assessment and management, includes uncertainty distributions, and optimizes prevention and control options. Operationalizing the approach described by Shogren (2000) by using SDP, they were able to optimize strategies by forecasting into the future and accommodating expected changes, societal responses, and their feedback interactions. SDP accommodates environmental changes, management responses to those changes, and in turn the impact of new management strategies on the environment. This enables the authors to identify the combination of prevention and control efforts that maximizes social welfare even given the uncertainty of invasion events. SDP also allows the determination of the global optimum even for computationally difficult problems. SDP has been used in behavioral ecology to identify optimal foraging strategies (Mangel and Clark 1988), and in natural resource management, including biocontrol (Shea and Possingham 2000, McCarthy, Possingham, and Gill 2001). Following a hypothetical example, these authors applied the modeling framework to zebra mussel invasions of un-invaded lakes providing cooling water for power plants. Their results suggest that society should be spending about $240,000 per year to keep zebra mussels from invading each lake with a power plant to prevent fouling of pipes. This is in contrast to the $825,000 that the U.S. Fish and Wildlife Service spent in FY2001 for prevention and control efforts for *all* aquatic nuisance species for *all* lakes.

The second modeling framework is the optimal control approach, as illustrated by, say, Olson and Roy (2002) and Settle, Crocker, and Shogren (2002). They constructed an integrated bioeconomic model to examine how invasive lake trout affect native cutthroat trout in Yellowstone Lake. Their optimal control/STELLA model simulated the integrated ecological-economic model of Yellowstone Lake to address three questions. First, they showed how the integration of the economic and biological systems leads to population results different from those from treating the two systems as separate. Consider now a sketch on how this approach would work for an example of an integrated bioeconomic model for invasive plants on agricultural lands. First, a plant-based ecosystem consists of two species—the crop and invasive species—that interact with each other for survival. The populations of both plants are linked with each other: invasive species crowd crops for survival, and crops try to survive. We represent the interaction between the two species through two state equations for species populations: the population of invasive species and the population of crop at time t. The equation of motion for the invasive species is a function of the population of the species, the crop, farmer harvest of the invasive species, and a manager's expenditures to cull invasive species. Assume that the population growth of the invasive species is an increasing function of its population and a decreasing function of culling. The equation of motion for crop is a function of populations of invaders, crops, and harvest. Let more crops increase survival success, more invasive species cause more crop mortality, and greater harvest of the crop reduce the crop population. Using the population state equations, the steady state is determined where the population has reached its new steady state and both populations are stable. Since reaching this new steady state equilibrium may take several decades or even centuries, the model can be used to account for both the difference between the two equilibria and the behavior in the intermediate time when the system is not in a steady state.

Second, one can now link the plant state equations to the two economic agents—the representative farmer and the manager. Harvest functions are defined to determine the removal rate of the invasive species given their populations and the time spent removing them by the average farmer. The harvest function for invasive species is a function of the population of invaders and the time spent killing the invasive species. The harvest function for the crop is a function of the population of the crop and the time spent harvesting the crop. The second link between the ecosystem and the economic system is through the manager. A manager directly interacts with the ecosystem by monitoring and controlling invasive species to reduce pressure on the crop. The question is, how much pressure should the manager put on invasive species to help the crop species? Suppose that the manager has a fixed budget to allocate across his or her various ac-

tivities that is determined by outside funding. Assume that the manager expends his fixed budget on one of two activities—culling invasive species or other activities in R&D to improve agricultural productivity.

Third, assume that a representative farmer splits his limited time between two activities: the farmer gains utility from the net profits from harvesting the crop given expenditures on harvesting and killing invasive species. These conditions define the optimal allocation of time between harvesting the crop and killing the invasive species, as determined by the population of each species in the ecosystem and the state of the R&D. Finally, suppose that a manager considers how much effort to devote to culling invasive species versus investing in R&D for improved productivity. Given the average farmer's best-response functions on how he will respond to the current conditions of the field, the manager acts as a social planner and maximizes the discounted stream of intergenerational utility of future farmers. Given this framework, one can explore the importance of including or excluding feedback loops between the systems. We will also explore alternative control policy strategies that relate to the invasive species under consideration.

The third modeling approach is the general equilibrium ecosystem modeling (GEEM) approach developed by Finnoff and Tschirhart (2003, 2005a) and applied to the problem of invasive species in Finnoff and Tschirhart (2005b) and Finnoff, Strong, and Tschirhart (2005). The appeal of a general equilibrium (GE) approach for analyzing the ecosystem consequences of invasion is similar to the appeal of GE in economics. GE methods allow for the representation of the feedback effects between all markets and agents in an economy. Because feedbacks abound in both economies and ecosystems, there are many problems that are intrinsically general equilibrium in nature.

One can exploit the similarities between economies and ecosystems to construct linked general equilibrium models of both systems. This GEEM approach avoids the tradition in economics of modeling ecosystems as a mere technical constraint on economic activity and explicitly represents the ecological system, the economic system, and their linkages. Finnoff and Tschirhart (2005a) link a dynamic economic computable general equilibrium (CGE) model with a dynamic general equilibrium ecosystem model (GEEM). CGE/GEEM is applied to the Alaskan economy to value the welfare consequences of endangered Steller sea lion recovery measures via alternative pollock quotas (a primary prey species). The models are linked through two ecosystem services, fishing and recreation, to an eight-species marine ecosystem. The bioeconomic model introduced admits a second ecosystem service, and more importantly it accounts for how the two services are impacted by interactions within the eight-species ecosystem. Steller sea lion recovery measures via alternative pollock quotas are shown to result in regional welfare gains through altered levels of all ecosystem populations, economic factor reallocation, changes in all regional prices, incomes, demands, outputs, imports, exports, and differential rates of factor accumulation.

Of the eight species modeled, four are used directly in the economy either as consumption goods (fish) or non-consumption goods (marine mammals). While non-use values associated with the ecosystem (e.g., existence values) are not considered, all species have value for the economy because the other four species are used indirectly as support for ecosystem services. A portion of the regional welfare gains from reduced pollock quotas follow from the regional economy's relying less on resource extraction ecosystem services and more on non-extraction ecosystem services.

With respect to invasive species, Finnoff and Tschirhart (2005b) extend the GEEM approach to describe a model of plant resource competition and how energy capture and allocation efficiency can determine the outcome of this competition. The method allows the identification of species that are likely to be successful invaders based on their individual physiological parameters. In many instances the parameters can be found in the existing botany literature or determined through experimentation. After identifying successful invaders, the most effective means of preventing their spread or controlling them once they are established can be related to how the means impact the physiological parameters.

Finnoff, Strong, and Tschirhart (2005) consider multiple simultaneous invasions of rangeland by annual grasses (cheatgrass) and forbs (leafy

spurge). Focusing on rangeland stocking decisions reveals the critical roles played by producer foresight and lease duration on the speed of leafy spurge invasion. By incorporating the resource competition process of multiple plant species, the results indicate that although leafy spurge may not exist in a non-foraging equilibrium, once grazing animals are introduced at profitable levels, a niche may be created within the ecosystem, allowing spurge the potential to thrive. This niche creation would not have been observed in a standard single-species bioeconomic model. The use of GEEM provides insights on how within-period competition may drive the population dynamics across time. Even small perturbations across longer time horizons can have drastically different effects on the ecosystem, and in turn on the economic health.

A key to integration work is to decide whether including all the details and all the feedback loops between the ecological and economic system is worth the trouble. More detail is better only if one can make better predictions about both systems. Settle, Crocker, and Shogren (2002) find that integration matters a great deal for predictions about the physical environment. Using the population of cutthroat trout as a yardstick, they found that ignoring feedbacks biases risk estimates by overestimating cutthroat populations in the worst case and by underestimating them in the best-case scenario. The difference arises from fishermen's behavior. Without feedback, fishermen continue to fish as before, putting constant pressure on the cutthroat. With feedback, fishermen exploiting declining cutthroat populations adapt by fishing less and visiting other attractions more. They also found a troubling result from a species protection perspective—based on visitor preferences, a small difference between the present value of net benefits between the best- and worst-case scenarios existed, which suggested that the gill-netting policy for lake trout was inefficient.

Other cases can differ. Finnoff, Strong, and Tschirhart (2005) consider two feedback loops for zebra mussels in a representative lake in the Midwest—the link between the biological system and firms, and the link between the manager and the firm. For both loops, the beliefs of the decision maker in question regarding invasions are central. In the absence of the link between the biological system and firms, the firm behaves *as if* there is no change in the biological system. The firm either uses too few or too many inputs relative to the optimal baseline. In turn, output correspondingly either under- or over-shoots its targeted level; either way, this results in opportunity cost losses from production shortages or surplus, determined ex-post. For the case of the benevolent manager and firm, removing feedback causes the manager to act *as if* the firm does not respond to changes in state. When excluding feedbacks, the model necessarily determines the consequences of the invasion and behavior of firms, even though the firm or social planner does not take them into account. The results suggest that including feedbacks matters, but not in every dimension. Both biological and economic consequences of not addressing feedbacks are sensitive to the initial conditions of the environment, behavioral perceptions about the state of the environment, and the completeness of the manager's beliefs.

Valuation

All integration models make some presumption about the relative costs and benefits of invasive species, usually based on the best market data available. And in general, the most visible costs and benefits of invasive species neglect or control have primarily been the numbers generated by Pimentel et al. (2000), who are biologists. Most economists are skeptical about these numbers because they are derived from a summation of replacement costs. Seen in engineering economics, these numbers are constructed by simply multiplying price times quantity rather than by using the standard surplus welfare measures developed over two centuries of research.

Consider now an economic framework to establish values for reducing some of the risks posed by invasive species. Following Rosen (1988), consider a valuation model constructed under the assumption that invasion time (or time when damages occur) is a known parameter. This reflects a common belief (by biologists and economists) that these invasions are inevitable,

and that it is just a matter of time until they occur. In this setting the objective is to determine a representative individual's willingness to pay (WTP) to delay the inevitable invasion. This presumes an invasion *will* occur at some point in time, our interest thus being in the value of delaying this certainty.

Another characteristic of this setting is that with a certainty of invasion, control is the only relevant investment. To determine the WTP to marginally delay invasion damages—the WTP to delay the inevitable, i.e., temporary prevention—it is necessary to construct a model of intertemporal behavior with periodic utility defined over consumption and environmental quality. A representative individual is framed as facing an intertemporal budget constraint and making choices to maximize his or her utility over non-invaded and invaded states. In the non-invaded state, the individual receives constant utility from the corresponding environmental quality. When the invasion occurs, utility depends on consumption minus some market damage and the lower level of environmental quality. As expenditures influence only market damages, the setting is purely self-insurance, and the individual's utility function is given by

(1)
$$\underset{c,x}{\bar{U}} = \int_0^{\tau} \bar{U}^0 e^{-\rho t} dt + \int_{\tau}^{T} \bar{U}^1(c(t) - \alpha D(x(t) + \tilde{x}(t)), Q^1) e^{-\rho t} dt,$$

where $\bar{U}^0$ is the constant utility in the non-invaded state, ρ is the rate of time preference, $\bar{U}^1$ is the utility in the invaded state, c is the consumption in period t, α is the proportion of damages faced by the individual, D is the damage function where $D_x \leq 0$, x is the monetary contribution to invasion prevention/control, $\tilde{x}$ is all contributions by other parties, Q^1 is the environmental quality in the invaded state, τ is invasion time, and T is time of death. It is possible to simplify expression (1) by assuming that either ρ or T is sufficiently large such that $e^{-\rho T} \to 0$ such that

$$\underset{c,x}{\bar{U}} = \int_{\tau}^{T} \left(\bar{U}^1(c(t) - \alpha D(x(t) + \tilde{x}(t)), Q^1) - \bar{U}^0 \right) e^{-\rho t} dt + \frac{\bar{U}^0}{\rho}.$$

The constant $\bar{U}^0 / \rho$ may be dropped since any monotonic transformation of $\bar{U}$ preserves orderings:

(2) $$\to \underset{c,x}{U} = \int_{\tau}^{T} \left(\bar{U}^1(c(t) - \alpha D(x(t) + \tilde{x}(t)), Q^1) - \bar{U}^0 \right) e^{-\rho t} dt.$$

The individual's intertemporal expenditures are constrained by his endowed wealth, W, he is assumed to confront a pure-consumption-loans market at interest rate r, and he cannot die in debt. Thus he exhausts his wealth over the course of his lifetime, so the choice of consumption path $c(t)$ is constrained by

(3) $$W = \int_0^{T} \left(c(t) + x(t) \right) e^{-rt} dt.$$

The associated Lagrangian can then be formed:

(4) $$\underset{c,x}{L} = \int_{\tau}^{T} \left(\bar{U}^1(c(t) - \alpha D(x(t) + \tilde{x}(t)), Q^1) - \bar{U}^0 \right) e^{-\rho t} dt + \lambda \left(W - \left(\int_0^{T} \left(c(t) + x(t) \right) e^{-rt} dt \right) \right),$$

where λ is the shadow value of an increment in wealth. Differentiating (4) with respect to consumption and control expenditures and assuming an interior solution leads to the following first-order conditions (supplemented with a binding budget constraint):

(5) $$\bar{U}^{1\prime}\left(c(t) - \alpha D\left(x(t) + \tilde{x}(t) \right), Q^1 \right) e^{-\rho t} - \lambda e^{-rt} = 0,$$

(6) $$-\alpha \bar{U}^{1\prime}\left(c(t) - \alpha D\left(x(t) + \tilde{x}(t) \right), Q^1 \right) D_x e^{-\rho t} - \lambda e^{-rt} = 0.$$

Subscripts and primes indicate derivatives. Equation (5) requires that the individual consume until the discounted marginal utility of consumption equates the discounted value of an increment in wealth. Similarly, (6) requires the individual to invest in control of the invader until the discounted marginal utility of control equates the discounted value of an increment in wealth. Together, these conditions require the individual to divide his expenditures between consumption and control (post invasion) so that the periodic mar-

ginal rate of substitution between the two goods is unity (ratio of prices, both one).

The value of delaying a harmful invasion by a marginal amount can be interpreted as simply what an individual would be willing to pay to stay in the non-invaded state an increment of time longer. This can be found by applying the envelope theorem to (4):

$$\frac{\partial L}{\partial W} = \lambda \,, \tag{7}$$

$$\frac{\partial L}{\partial \tau} = -\left(\bar{U}^1\left(c(\tau) - \alpha D\left(x(\tau) + \tilde{x}(\tau)\right), Q^1\right) - \bar{U}^0\right)e^{-\rho\tau} \,, \tag{8}$$

$$V = -\frac{dW}{d\tau} = \frac{\partial L / \partial \tau}{\partial L / \partial W} = \frac{-\left(U^1\left(c(\tau) - \alpha D\left(x(\tau) + \tilde{x}(\tau)\right), Q^1\right) - \bar{U}^0\right)e^{-\rho t}}{\lambda}. \tag{9}$$

From (6) we have

$$V = \frac{-\left(\bar{U}^1\left(c(\tau) - \alpha D\left(x(\tau) + \tilde{x}(\tau)\right), Q^1\right) - \bar{U}^0\right)e^{-\rho\tau}}{\left(\dfrac{-\alpha \bar{U}^{1\prime}\left(c(t) - \alpha D\left(x(t) + \tilde{x}(t)\right), Q^1\right) D_x e^{-\rho t}}{e^{-rt}}\right)} = \frac{\left(\bar{U}^1\left(c(\tau) - \alpha D\left(x(\tau) + \tilde{x}(\tau)\right), Q^1\right) - \bar{U}^o\right)e^{-rt}}{\alpha \bar{U}^{1\prime}\left(c(t) - \alpha D\left(x(t) + \tilde{x}(t)\right), Q^1\right) D_x} \,. \tag{10}$$

At $t = \tau$ we have

$$V = \left(\frac{\bar{U}^1\left(c(\tau) - \alpha D\left(x(\tau) + \tilde{x}(\tau)\right), Q^1\right) - \bar{U}^0}{\alpha \bar{U}^{1\prime}\left(c(\tau) - \alpha D\left(x(\tau) + \tilde{x}(\tau)\right), Q^1\right) D_x}\right)e^{-r\tau} > 0 \,. \tag{11}$$

Assuming that $\bar{U}^1 < \bar{U}^0$ and $D_x < 0$, the value term is always positive and equal to the discounted difference in utility from the invaded state to the non-invaded state over the marginal benefit of invasion control expenditures.

The key to such valuation exercises is that they are based on plausible scenarios that account for both economic circumstances and biological parameters, and the feedbacks between the systems. Developing scenarios for people to value without accounting for the ecological economic interdependencies can lead to biased valuation estimates, at best, or off-point numbers at worst—ones that do not capture or relate to the underlying physical and social problems associated with the invasive species. Consider now one example in which we used both an integrated model and a valuation survey to estimate the value of protecting cutthroat trout in Yellowstone.

An Application of a Joint Integration-Valuation Approach

We combined integrated models and valuation to measure preferences for reduced risks to the native species cutthroat trout in Yellowstone National Park [for complete details see Settle and Shogren (forthcoming)]. The survey was designed to explore the degree to which people are concerned about protecting a native species like cutthroat trout within the Park against threats from invasive species like lake trout. The goal was to apply into the field the insight of laboratory valuation work from the past decade, and to use our empirical results to parameterize the composite visitor's demand within an integrated bioeconomic model. Using a seven-step experimental strategy, we created a *Yellowstone Interactive Survey*.

The first step was to develop *wildlife lotteries* to which people can assign an economic value. These wildlife lotteries represent the probability that a visitor will *experience* a species and the core attractions. Due to the large number of possible permutations for the set of wildlife lotteries, we limited our probability distributions of species to those most important in order to parameterize the integrated bioeconomic model. The 90 lottery pairs capture the reality of most environmental policy by defining many of our wildlife lotteries as low-probability/high-outcome lotteries.

The second step was to use *market-like arbitrage* as a disciplining device in valuation. Asking people to value low-probability/high-outcome lotteries introduces the possibility that people might not act as rational as expected utility theory presumes. The fear was that people could have inconsistent preferences over the wildlife lotteries, and therefore state inconsistent values. *Rational valuation* is defined by consistency between valuation and preference. One solution to

this is to introduce arbitrage into the experimental design. Following Cherry, Crocker, and Shogren (2003), over the course of 10 rounds, each participant was presented with two lottery markets side-by-side. One market was a real market: people played money lotteries for actual cash. For these real lotteries, we also used a low probability of a high payout and a high probability of a low payout. The second market was hypothetical wildlife lotteries: people played these lotteries but did not get paid in cash.

The third step was to construct the actual *interactive valuation survey* questions such that they matched up with the integrated model. The instructions explained each stage in the survey: each person was presented with two separate situations, each with two options (option A and B). Each person said which option he or she preferred in each situation (A or B), and then stated a dollar value for the two options in both situations. The people were aware that a *computer market* could buy, sell, or trade with them in situation 1 according to indicated preferences and values. At the end of a round, the next round appeared with a new initial money balance in situation 1 and new options for both situations. Once they completed the last round, we determined their earnings for the survey.

The fourth and fifth steps were the actual implementation of the survey *inside the Park* and *over the Internet.* In the Park sessions, we set up tables outside the Visitors Center at the south entrance to Yellowstone and Grand Teton National Parks. Visitors were asked if they would be willing to participate in an experiment taking about 30 minutes on laptop computers set up on the tables. We also ran the survey over the Internet and attracted participants through a paid advertisement run over 5 weeks on the New York Times web page (www.nytimes.com). Focusing on Park visitors and New York Times web readers further suggests that our valuation estimate reflects an upper bound on the composite person's value for cutthroat.

The sixth step was to *evaluate* the valuation statements. For tractability, assume that the average respondent's willingness to pay function is separable in each argument—e.g., the value of catching a cutthroat trout does not depend on whether you have seen the core attractions of the park. We used this assumption to back out the value of each probability from a large number and different types of lottery pairs. Using this method, we estimated values for each of the species in the park.

The final step was to add the estimated values as parameters in the integrated model. We next determined the value for each of the probabilities of seeing/catching each species and used these estimates to parameterize the value to see/catch each species in our composite visitor's welfare function. If a visitor valued a cutthroat trout at $4 and had a 25 percent chance of catching one, the visitor valued that 25 percent chance in the lottery at $1. The values for each species were included in our welfare function used in the simulations to measure the welfare to visitors of visiting Yellowstone National Park.

The results from the combination of integration and valuation were interesting. While integrating economics and biology was worth the effort for predicting physical changes, we found it did not matter for welfare estimates. We estimate a trivial difference between the present value of net benefits between the best- and worst-case scenarios. The average person surveyed cared more about improving road quality than protecting cutthroat trout. His bumper sticker would read "fix the roads, forget the fish." This suggests that if Park managers want to justify cutthroat trout protection based on visitor preferences, they would benefit from better educational tools to close the gap between today's average visitor and the far-sighted eco-tourist. Regardless, the point is that the valuation exercise was guided by the demands of an integrated model and that the structure of the integrated model was affected by what needed to be valued. Future work exploring this second level of integration seems most worthwhile.

Concluding Remarks

Economics serves a role in invasive species management through integration of economic and physical circumstances, and a better idea on how people value prevention, control, and eradication. Herein we focused on integration and valuation given our work over the last decade, stressing the usefulness of more research that blends the two areas. We close now by briefly identifying two additional critical invasive species topics needing

more economic attention—trade/growth and incentive design. First, we know that the international movement of people, goods, and raw materials has amplified the rate at which species move beyond their natural environs. While most invasive species do not survive in their new habitats, those that do live generate substantial financial damages at the local and national level. Future research should possibly continue to consider biological invasions as an unintended by-product of capital accumulation and trade policy (e.g., Barbier and Shogren 2004, Costello and McAusland 2003). Production in the economy is the outcome of decentralized decision making by firms such that any increase in an individual firm's capital stock carries the risk of causing a parallel increase in the total stock of biotic invaders. One could examine potential spillover effects from an increase in this stock of invasive species as a direct negative impact on the productivity of all firms, a direct negative or positive impact on the utility of people, and an increase in the risk of future loss of welfare. The open public policy question in trade and growth is what view should dominate—that of the experts or lay public—when people gain welfare from the invader (e.g., muted swans, ornamental flowers).

Finally, studying the design of incentive mechanisms to minimize risks of invasive species on native species should be a priority (e.g., see Horan and Lupi 2005). Invasive species are the second leading risk to native species. Many invasive species flourish along the edges of the habitat reserve. Invasive plants can penetrate the habitat reserve for up to five kilometers, reducing the quality of the habitat along the edge. Also, the habitat edges are susceptible to nest parasitism and increased threat of endangered species to predation by invasive species. To mitigate the influence of invasive species, many scientists have recommended designing habitat reserves that maximize the portion of the conserved acres that contribute to the core habitat area as opposed to the share in which habitat quality is jeopardized due to edge effects. Further, given limited conservation dollars, designing habitat reserves to minimize edge effects, thereby reducing the predation of invasive species, will also create larger core habitat areas for native species, providing them the highest probability of survival. The idea is to study cost-effective incentive mechanisms to agglomerate habitat, private or public or both, in order to minimize edge effects and risks to native species.

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