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Bioeconomics of invasive species: using real options theory to integrate ecology, economics, and risk management

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Abstract Policy makers face two countervailing incentives in invasive species management—the Pull-incentive to move quickly and the Push-incentive to wait-and-see before making irreversible investments. Real options theory is used to help understand this fundamental trade-off both in design and application. In designing policies, real options theory shows how the management of invasive species should account for the intertwined concepts of ecological risk/ecological irreversibility and economic risk/economic irreversibility. In applying policies, real options theory shows for species spreading slowly with little uncertainty, the push-incentive dominates, advocating a wait-and-see approach. In contrast, for fastspreading species, their diffusion is too fast and too unpredictable to do anything other than act immediately – the pull-incentive dominates. In addition, results indicate both the source and the magnitude of uncertainty matter, but the nature of the impact depends on the irreversibility of the policy decision highlighting the key value of flexibility in policy design and application.

Keywords Decision-making · Policy options · Pull-incentive · Push-incentive · Uncertainty

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Introduction

Today scientists and policymakers recognize that invasive species pose a risk to biodiversity around the globe (e.g., Didham et al. 2005; Davis et al. 2011; Simberloff et al. 2013). Invasive species are expanding in scope worldwide, encroaching into and causing unwelcomed changes to agriculture, forestry, fisheries, and ecosystem services (see e.g., Lodge 2001; Mack et al. 2000; Archer and Shogren 1996; Feder and Regev 1975, Lichtenberg and Zilberman 1986, Knowler and Barbier 2000; Holmes et al. 2009; Rothlisberger et al. 2010). Economists define efforts to reduce these invasive species risks as weak-link public goods for two reasons—(i) they are public goods because invasive species ignore political boundaries causing private control efforts to convey broader social benefits of control, and (ii) these are weak-link risks because the least effective protector's efforts determines the overall level of risk to all (see Perrings et al. 2002). This characterization of invasive species risks implies that policies must be publicly-funded, implemented over the long-term, and applied at broad scales.

Policy makers react to the risks posed by invasive species by investing in mitigation or adaptation or both (Shogren, 2000). Defining cost-effective risk reduction strategies, however, remains a challenge. Cost-effective risk reduction strategies require collective investments given that the weakest link in the protective chain determines the risk to all. Risks are defined by both ecological risk (the possibility and severity of an adverse ecological event) and economic risks (the possibility and severity of economic losses) which emerge over the long run. The stream of damages occurs over long time horizons, which limits how well one can predict the benefit of mitigation (avoided future damage). In this dynamic world, risk reduction strategies become risky investments.

Herein we discuss how to use bioeconomics to help define a coherent framework to evaluate cost-effective invasive species risk reduction policies. We view public policies for invasive species control as risky long-term lotteries, in which the mix of ecological processes, economic systems, and management objectives work together to define the level of damages and associated probabilities. In contrast to many private control efforts, we treat public policy as a classic case of investment under uncertainty since control strategy effectiveness is uncertain and typically implies a long-term commitment due to either sunk costs or political commitments (sunk political capital). We frame these risky investments using real options theory—a popular decision tool from financial management and natural resource economics (see Dixit and Pindyck 1994). We use real options theory to explore the push-pull tensions facing policymakers who must consider the optimal stringency in response to invasive species and the optimal timing of the deployment of these policies. This tension asks policymakers to balance a push-incentive to hurry up to prevent an irreversible biodiversity loss against the pull-incentive to wait-and-see until they are convinced no better alternative investments exists (i.e., control other species or control in other areas).

The push-incentive: upside ecological risk

We first focus on ecological risk and how it provides a push-incentive to invest in control sooner rather than later. Ecological risk (the likelihood and severity of an unfavorable ecological event) defines the risk associated with inaction (the likelihood and severity of higher than expected damages) and risk associated with investments in risk reduction strategies (the likelihood and severity of a lower than expected return). Undesirable events that result in ecological damage are treated as a random variable with a known distribution. The uncertainty in ecological damage may arise from both ecological sources of uncertainty (inability to accurately predict the evolution of an invasion) and economic sources of uncertainty (inability to predict consequences of human responses to invasion).

As shown in Fig. 1, outcomes left of the expected damage represent good news for society (ecological damage is lower than expected) and outcomes to the right represent bad news. If ecological damage turns out to be larger than expected, a policymaker who did nothing ex ante to reduce the risk will regret this irreversible damage. Risks posed by inaction (the likelihood and severity of larger than expected irreversible damage) argue for a policy to be implemented immediately (Chichilnisky and Heal 1993; Fisher and Hanemann 1993).

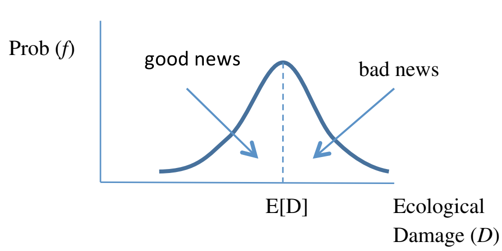


Fig. 1 Distribution function that describes ecological risk

The pull-incentive: downside ecological risk

Consider now the countervailing source that arises when making irreversible investments in invasive species control. Most policymakers value flexibility—they do not want to commit "too early" when making an irreversible investment. They have a pull-incentive to wait-and-see. If realized ecological damages are smaller than expected, the policymaker regrets moving too quickly since lower-than-expected ecological damage constitutes bad news for the return on risk reduction investments (Pindyck 2007). He or she wants to learn more before committing scarce investment resources that cannot be retrieved. The bad news for the return on investment means limited public funding could have been reallocated to yield a larger reduction in ecological risk. This environmental analog of Bernanke's (1983) "bad news principle" provides a case against more immediate action. Given risk and irreversible commitment, people who value flexibility will delay investment until "enough" uncertainty about the nature of the damages is revealed over time.

To formalize this concept of a pull-incentive, consider an investment (I) that lowers the expected value of ecological damage from E[D] to E[D′] into perpetuity. The expected return from this investment is the difference in expected damage, $E[R] = E[D] - E[D']$, and the expected net present value of benefits from investing, $E[NB]$, is the discounted flow of these expected returns over time. Discounting future reductions in ecological damage accounts for the time value of public funds devoted to reducing ecological risk – public funds are typically limited and when they are devoted to reducing a specific ecological risk they are unavailable to be invested in reducing other ecological risks. According to benefit-cost analysis, the investment should be undertaken if $E[NB]$ is greater than or equal to I or when the expected *net* present value of the investment is non-negative. But benefit-cost analysis is based only on the expected return (first moment of the ecological risk distribution) and does not consider the likelihood that investment cost may exceed expected net benefits, $E[NB] < I$. To account for this, a decision framework must account for the likelihood and severity of a return that is lower than expected. We coin this interval of the distribution, to the left of the expected value, as investment risk (Fig. 2). Outcomes to the left of E[D] where damages are less than expected, are good news for society but *bad news* for the return on investment if an investment had been made. The point is that if damages turn out to be in this interval there was less of a need to deploy the investment to lower expected damages. In contrast, outcomes to the right of E[D] where damages are higher than expected, are bad news for

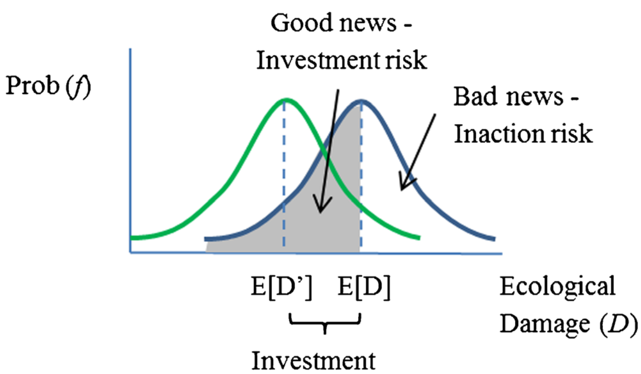


Fig. 2 Relationship between ecological risk and investment risk for a risk reduction investment

society but provide a higher than expected return if an investment has been made (good news for return on investment). Events viewed as good news for society make it more likely that $E[NB] < I$.

Hurry up and wait: the push and pull incentives

This relationship between ecological and investment risk leads to two implications. First, high levels of ecological risk require investments that are prone to more risk. When faced with a more risky investment, such that the tails of the distributions in Fig. 2 are further from the mean, decision makers can be expected to decrease the amount of risk reduction investment, or at least postpone the investment (look to other less risky investments). This response to risk is intuitive in many settings (e.g., new car purchases) but is counter-intuitive for risk reduction investments since many of the most contentious questions concerning investment in risk reduction strategies are characterized by substantial variance in the distribution of payoffs. This counter-intuitive result highlights how making a long-term risk reduction investment changes the decision maker's incentives – good news for the environment (low ecological damage) becomes bad news for a risk reduction investment. This investment incentive runs counter to the precautionary principle.

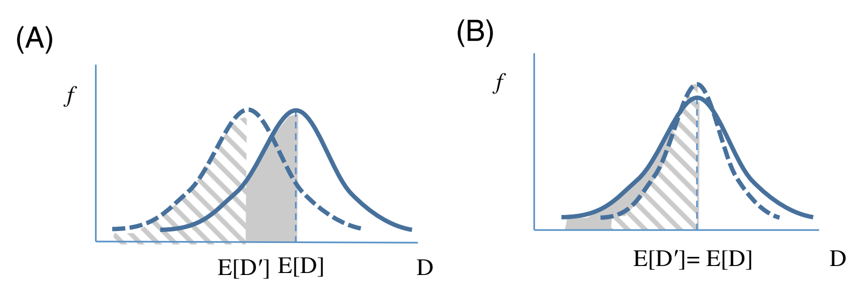
Second, risk reduction investments may make future investments more risky. Consider a risk reduction investment that reduces the expected damage (Fig. 3a). The shaded area represents the investment risk associated with the initial risk reduction investment and the hatched area represents investment risk for the subsequent investment. While lowering expected damages reduces the likelihood of bad news from society's perspective, it does not necessarily reduce the likelihood of bad news from the policy maker's perspective. Lowering expected damages will decrease the probability associated with small losses (nearly expected damage) but will increase the probability of large losses (low damage). Likewise, a risk reduction investment that reduces the variance of the distribution (Fig. 3b) will decrease the probability associated with large losses (low damage) and will increase the probability of small losses (nearly expected damage). The net effect of shifting the mass of the distribution will determine whether the future investments are more or less risky.

Ecological and investment risk in invasive species control

The tension between ecological and investment risk has implications for the magnitude and timing of invasive species risk reduction investments. The mechanism driving spread is species-specific, requiring specific management strategies that may slow, stop, or reverse population growth and spread. For example, quarantines have been employed to slow the spread of emerald ash borer through firewood and nurseries (Poland and McCullough 2006). Inspection and quarantines of used tires have limited the spread of the Asian tiger mosquito (Moore and Mitchell 1997). Eradication strategies are attempted, with varying degrees of success. For example, while eradication of the gypsy moth failed, suppressing outlying populations along the population front slowed the overall spread (Sharov and Liebhold 1998).

Quarantine and eradication policies are examples of costly investments made to reduce risk, in the face of uncertainty. Ecological risk or natural system variability flows from biogeophysical factors and biological processes of the species (growth, mortality, movement), as well as from the perturbations caused by human interventions that randomly alter the dynamics of invasive species. This variability in spread dynamics translates into uncertain costs and benefits of risk

Fig. 3 Effect of risk reduction investments on investment risk of subsequent investments. **a** Decrease in the expected damages. **b** Decrease in the uncertainty in future expected damages



reduction investments. Compounding the uncertainty is that environmental damage may be partially or totally irreversible. In turn, risk reduction investments require or commit policy makers to expenditures of resources (sunk costs) that depend on when they are deployed and with what stringency.

Given these characteristics and the tradeoff between ecological and investment risk for invasive species risk reduction policies, we employed real options modeling to consider the optimal timing and stringency of policies directed at several high profile invasive species (Sims and Finnoff 2013). Real options analysis applies tools from finance that specify a stochastic process for the asset of interest, and determines action thresholds describing circumstances under which investment in the policies is optimal (and at what stringency) and thresholds when it is optimal to abandon the investment (see Dixit and Pindyck 1994).

Real options theory: combining push and pull incentives

Implementing policies to reduce the risks of invasive species requires a balancing of the economic and ecological damage with the cost-effectiveness of the policies, in the face of ecological and economic uncertainty, coupled with the irreversibility of invasive species damages and the sunk costs. The ability to time the deployment of policies and the stringency (intensity) has been shown to be of significant value, and can be incorporated through the method of real options (Dixit and Pindyck 1994). In the context developed in Sims and Finnoff (2013), the real options method derives points in the invasion process within which a policy maker should make an irreversible investment in the risk reduction policy. Technically, the model is one of regime switching under uncertainty (details are provided in the Appendix and see Brekke and Øksendal 1994; Miranda and Fackler 2002), in which the timing of a policy is found by optimally switching between a regime with and without the risk reduction policy. The method allows the relative influence on decision making of ecological and economic risk and factors such as spatial scale (whether physical or perceived) to be quantified.

As a motivating example to illustrate the usefulness of real options theory for invasive species policy, we apply the Sims and Finnoff (2013) wait-and-see framework[1] of an advancing population front to five well-known invasive species: bighead carp, silver carp, cereal leaf beetle, muskrat, and Japanese beetle. Based on annual spread data for the species (see Fig. 4), models of spread by reaction diffusion (continuous spread along a population front) were found to be best fit. Range (x, kilometers from the introduction point the edge of the currently invaded area) expands linearly over time ($dx/dt=r_0$) for invasions characterized by reaction diffusion (Liebhold and Tobin 2008).[2] Because the course of some invasions is more predictable than others, range expands by an arithmetic Brownian motion process, $dx=r_0dt+sdz_x$, in which range expansion is driven by the rate of invasive species spread (r_0) and volatility (s) of the process, where dz_x is the increment of a standard Weiner process. Estimates of r_0 and s for each species were found from linear regressions, shown in Fig. 4.

When initially discovered, each species has spread x_0 and caused economic damages D_0. As the invasion spreads, economic damages (D) are assumed to grow in an increasing fashion dependent on the range of the invader, at rate γ, so $D = D_0e^{\gamma(x-x_0)}$. The implication of this specification is that as the invader's range expands, damages increase by more than the increase in range. The assumption allows us to differentiate the damages of localized infestations with the damages of broad scale regional infestations. In general, the intrinsic rate of damage increase is difficult to predict due to the difficulty of quantifying economic damages (especially at the time scale of invasion) and predicting human responses to invasion. Given this we allow the intrinsic rate of damage accumulation to vary stochastically over time, $d\gamma=\delta\gamma dz_\gamma$ with volatility (δ) and increment of a standard Weiner process (dz_γ) that may be correlated with invasive species spread: $E[dz_\gamma dz_x]=\sigma dt$. If $\sigma>0$, a positive shock to invaded range is likely to be accompanied by a positive shock to the intrinsic rate of damage accumulation and $\sigma=0$ reflects completely independent sources of uncertainty. The inclusion of an upper bound on range, $\bar{x}$, implies that damages evolve stochastically around a trend until the invasion runs its course. At this point damages remain at $\overline{D} = D_0e^{\gamma(\bar{x}-x_0)}$. Similar to assuming a mean-reverting process for x, this cuts off the upside potential for damage making the damage process lognormally distributed only over the range $[0, \bar{x}\,]$.

Future damage is uncertain due to its dependence on the random variables x and γ. In the face of unknown future damages (ecological risk), a risk neutral policy maker can implement a costly risk reduction policy, at some optimal point in time. The policy reduces the expected rate of invasive species spread into uninvaded area. However, the efficacy of the policy is uncertain since the post-policy spread process remains stochastic. As the impact of the invasion is uncertain, the policy can be canceled if the invader does not cause as much damage as expected and reinstated if the invader reemerges as a problem in the future. The difference in spread, pre and post policy, is termed the "stringency" of the policy, and can be such that the invader is slowed, stopped, or reversed. Costs depend on the stringency of the policy and the size of the

[1] See appendix for details.

[2] In contrast, stratified dispersal is characterized by continuous spread and discontinuous, long-range dispersal that causes range to increase exponentially over time. See Sims and Finnoff (2013) for an analysis focused on stratified dispersal.

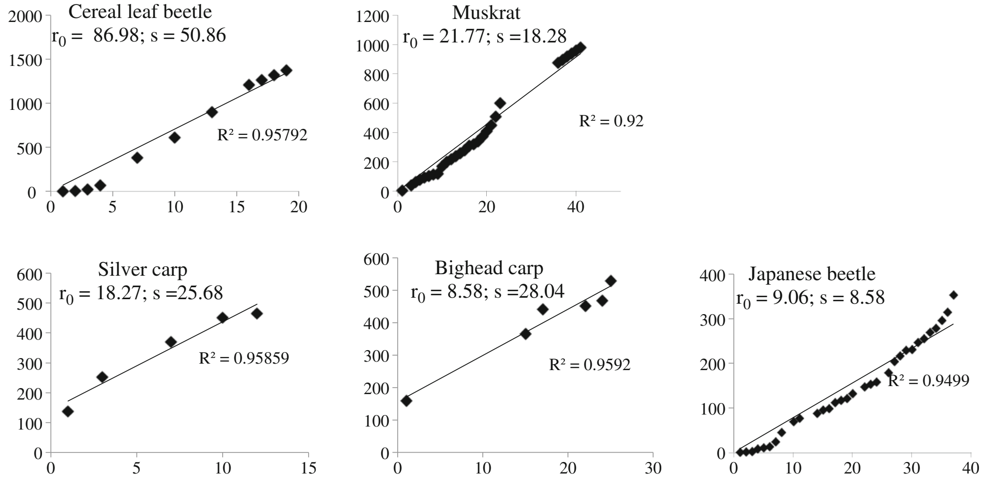


Fig. 4 Invasive species range (in km along the y-axis) over time (in years along the x-axis) with linear trend and ordinary least squares estimates of drift and volatility parameters. Data sources: Cereal leaf beetle (Andow et al. 1993), Muskrat (Andow et al. 1990), Silver and big head carp (Jerde et al. 2014), Japanese beetle (Allsopp 1996)

potential range of invasion. The policy maker makes a control investment now in exchange for an uncertain reduction in the impacts from invasion.

The policy implementation decision follows Dixit and Pindyck (1994) by making a comparison of the optimal value functions (expected discounted payoffs of taking optimal actions) that arise from doing nothing and incurring the damages at high spread rates, with the value function from implementing the policy, incurring the costs, changing the spread rate and lowering expected present value impacts from invasion. In the comparison of value functions, we can determine the value of delaying long-term commitments associated with policy implementation to gain more information about the impact of the invasion, which is called the *option value*. The decision rule is to implement the policy if the value function that arises from implementation exceeds the value function of continuing to do nothing (i.e., preserving the control option).

The relative values of implementing or not depend on the current state of the world, described by stochastically evolving spread (x) and damage accumulation rate (γ). The optimal timing of the policy in-turn depends on both the stringency of the policy, and the degree of irreversibility. Policies that can be implemented only once and last indefinitely (completely irreversible policies such as the release of biological control agents) also influence decision making in a different way from policies that can be canceled and additional new policies adopted in the future (partially irreversible policies such as state-level quarantines). The option value induces a cautionary element to decision making (i.e., pushing for a delay), whereas the rapidly advancing spread of the invader calls for more immediate action.

We now compare generic invasions from each species that differ by spread rates and volatility. For the starting conditions, we assume invasions have initially spread 0.5 km of a potential range of 1784 km. Initial damages are $D_0 = \$5000$ and the expected intrinsic rate of damage accumulation is assumed to be $E[\gamma] = 0.002$. With no economic uncertainty ($\delta = 0$), these assumptions suggest the expected percent change in damage $\left(\frac{E[dD/dt]}{D}\right)$ varies from 1.8 % (Japanese beetle) to 17.9 % (cereal leaf beetle). The cost of stopping the invasions range from \$736,000 to \$75 million.

We group the results into two types of invasive species: Those that spread slowly with little uncertainty (Japanese beetle, bighead carp, silver carp, muskrat), and those that spread fast with large amounts of uncertainty (cereal leaf beetle). Estimated spread rates, r_0, and levels of uncertainty (standard deviation) s, for the species are shown in Fig. 5 in relation to a unitary signal-to-noise ratio (r_0/s).

The slow-spreading group is characterized by more certain future damages, whereas the fastspreading group are characterized by larger expected damages (see Fig. 6).

Optimal thresholds (distance, in km) and policy stringencies (in % reduction of spread rates) for both completely and partially irreversible scenarios are shown in Table 1, across a range of levels of economic uncertainty (δ) with no correlation between spread and damage accumulation ($\sigma = 0$). If policies are completely irreversible and the invasion is relatively slow and predictable, there is an incentive to delay the

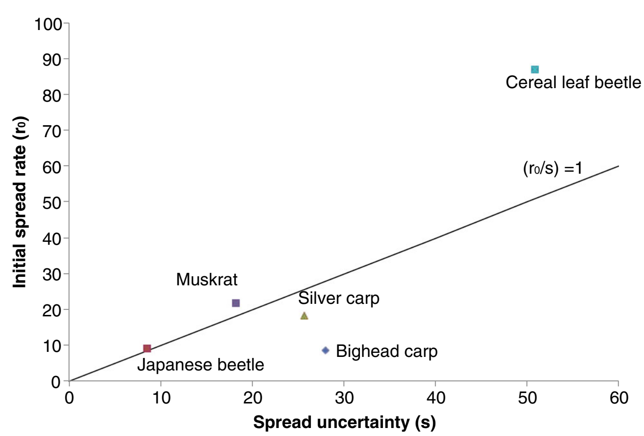


Fig. 5 Spread rates and standard deviation

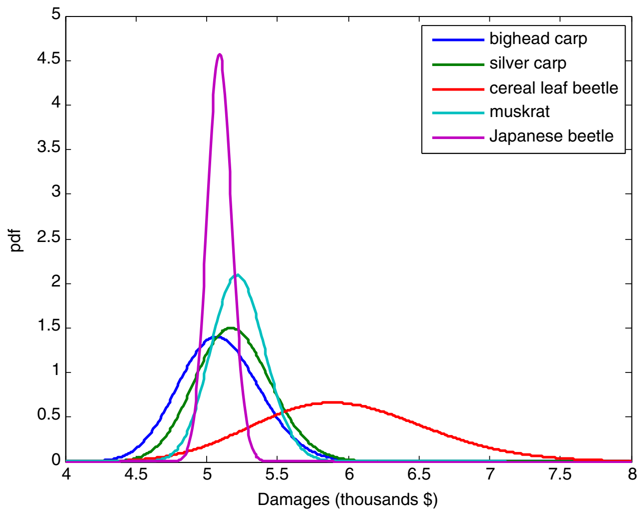


Fig. 6 Likelihood of future damage at $t = 1$ associated with select invasive species that cause \$5000 in damage at $t = 0$

implementation of the risk reduction policy, and take a wait-and-see approach to policy implementation. Surprisingly, taking a wait-and-see approach is an optimal strategy for relatively predictable invasions because the cost of waiting to gain information is minimal given their slow rate of spread. For instance, cereal leaf beetle is spreading so fast and with so much uncertainty that immediate policy implementation is called for (similar to that seen with other invaders in Sims and Finnoff 2013).

The results reveal three clear patterns. First, if policies are partially irreversible and can be adjusted, risk reduction policies should be implemented immediately in the invasion for all species (i.e., with thresholds of 0.56 km = 1 km^2) but with typically less stringency than in the completely irreversible case. All else equal, a wait-and-see strategy may be justified if a very irreversible policy is being applied to a species with a low signal-to-noise ratio (SNR): r_0/s.

Second, risk reduction policies are more stringent (magnitude of the spread reduction) for invaders with a large SNR in either irreversibility case. Invaders with a high SNR present a high reward, low risk investment, which encourages a larger investment in controlling these species. Figure 7 demonstrates this, comparing cereal leaf beetle with bighead carp. The investment and reduction in expected damage is significantly larger for cereal leaf beetle (high SNR), in comparison to bighead carp (low SNR). The exception is Japanese beetle with a moderate SNR and the least stringent policy, due to the lowest spread standard deviation. Japanese beetle represents the most predictable invasion and the least risky investment in control. This low risk encourages a more expedient response when spread reaches 350 km. This more expedient response discourages a more stringent policy since policy costs will be less heavily discounted. This highlights the tradeoffs between policy timing and stringency.

Third, we see that the source of the uncertainty matters. Economic uncertainty only influences the completely irreversible policies. This result arises because economic uncertainty tends to be correlated with more rapid spread, ecological uncertainty, influences both partially and completely irreversible policies. For a completely irreversible policy, increases in the ecological signal-to-noise ratio lead to a lower threshold of the risk reduction policy at a higher level of stringency (see Fig. 8). The Japanese beetle invasions are an exception—it is the most predictable invasion in the group, which results in low thresholds and low stringency (Table 1) causing the waves in contours in

Table 1 Optimal thresholds and stringency

Species	r_0	s	Signal to noise (r_0/s)	δ	Completely irreversible		Partially irreversible	
					Stringency (%)	Threshold (km)	Stringency (%)	Threshold (km)
Bighead carp	8.581	28.043	0.306	0	0.217	405	0.113	0.56
				0.05	0.262	531		
				0.1	0.317	657		
Silver carp	18.270	25.681	0.711	0	0.332	424	0.224	0.56
				0.05	0.355	513		
				0.1	0.394	661		
Japanese beetle	9.057	8.577	1.056	0	0.178	350	0.103	0.56
				0.05	0.217	460		
				0.1	0.242	526		
Muskrat	21.768	18.278	1.191	0	0.383	442	0.305	0.56
				0.05	0.400	522		
				0.1	0.430	665		
Cereal leaf beetle	86.976	50.865	1.710	0	0.802	0.56	0.802	0.56
				0.05				
				0.1				

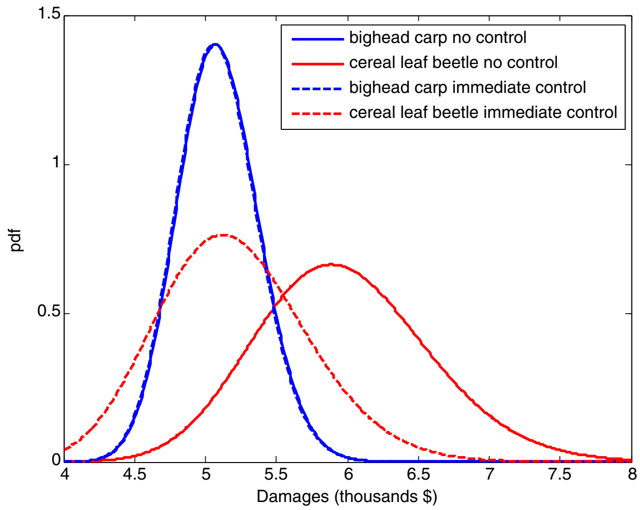


Fig. 7 Ecological and investment risk at $t=1$ for select species and partially irreversible policies

Fig. 8. The implication is that increases in ecological uncertainty make the returns available elsewhere in the economy superior to the returns from immediate investment in the completely irreversible policies. This comparison to other competing investments in control provides an incentive to delay any implementation of risk reduction policies especially when the SNR is high. This delay incentivizes implementation of a policy with greater stringency which again highlights the tradeoffs between policy timing and stringency. In our analysis, however, increases in economic uncertainty are not correlated with more rapid spread, leading in all cases to a greater delay in implementation of the completely irreversible policy, again at a higher stringency. A useful avenue for future work is to examine if economic and ecological sources of uncertainty in bioinvasions have similar relationships to the rate of invasion.

Real options theory: policy implications for design and application

We have shown how a policy maker faces two countervailing incentives in invasive species management—the Pull-incentive to move quickly and the Push-incentive to wait-and-see before making irreversible investments. We have shown how real options theory can be used to help understand this fundamental trade-off both in design and application. In design questions, the management of invasive species should account for the intertwined concepts of ecological risk/ ecological irreversibility and economic risk /economic irreversibility. Real option theory allows one to account for theses interactions, the feedbacks between systems, and the joint determination of outcomes.

In application, real options theory shows how invasive species can be clustered into two general types: species that spread slowly with little uncertainty; and those that spread fast with large amounts of uncertainty. We illustrate how the invasion signal-to-noise ratio can be used to distinguish between these two groups. For species spreading slowly with little uncertainty, the push-incentive dominates—a wait-and-see approach at (eventually) increased levels of investment may be an optimal strategy, depending on the irreversibility of the policy decision. In contrast, for fast-spreading species, their diffusion is too fast and too unpredictable to do anything other than act immediately - the pull-incentive dominates. The large potential returns from controlling these species immediately incentivize larger investments even though the volatility of spread makes these investments more risky. Other than the release of biological control agents or permanent separation of watersheds to combat aquatic invaders, most control action will only be partially irreversible making a wait-and-see approach hard to justify.

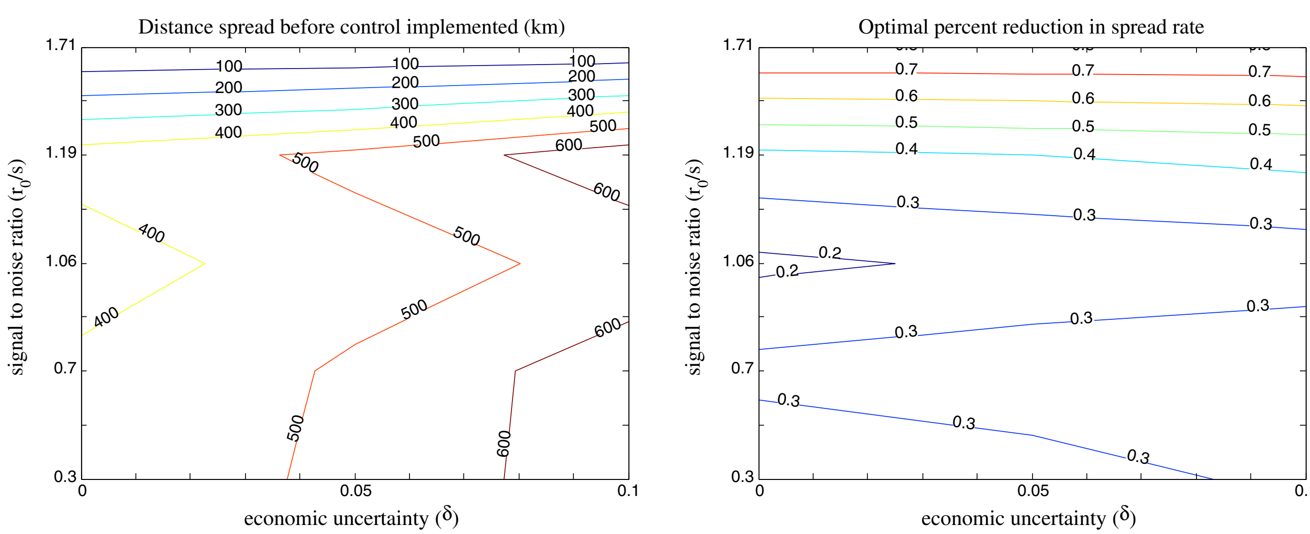


Fig. 8 Threshold and stringency by source of uncertainty, complete irreversible

Finally, we show how and why both the source and the magnitude of uncertainty matter. For species in which a wait-and-see approach is desirable, economic uncertainty has a minimal effect on policy stringency and timing. Spread uncertainty has a larger impact but the nature of the impact depends on the irreversibility of the policy decision - flexibility matters.

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Appendix

The policy maker must evaluate, at each instant in time, whether or not the policy should be implemented (t_1) and if so at what stringency (r_1) given all future policy cancellation $\{T_1, T_2, \ldots\}$ and implementation $\{t_2, \ldots; r_2, \ldots\}$ decisions are made optimally. A post implementation spread rate of $0 < r_1 < r_0$ will slow spread while those with $r_1 < 0$ reverse spread of the species. Razor's edge policies $r_1 < 0$ halt spread at the current extent of the invasion. Given the risk adjusted discount rate ρ, the optimal policy implementation decision (t_1, r_1) satisfies

$$W(\gamma,\ x;\ r_0) = \min_{t_1} \int_0^{t_1} D(\gamma, x) e^{-\rho t} dt + E_0 \left\{ \left[W(\gamma,\ x;\ r_1) + C\left(\bar{x},\ r_1\right) \right] e^{-\rho t_1} \right\} \tag{A.1}$$

subject to $\frac{d\gamma}{dt}, \frac{dx}{dt}, \gamma(0) = \gamma_0,\ x(0) = x_0, \lim_{t \to \infty} x(t) = \bar{x}$, and the first-order condition for r_1. In short, the evaluation at each instant in time minimizes expected damages and costs from that point forward by making a simple choice to continue to wait or to take action and lower the spread rate to some optimal level r_1 at cost $C(\bar{x}, r_1)$.

Following Dixit and Pindyck (1994),the policy implementation decision will be made based on a comparison of the optimal value function that arises when continuing with the status quo, denoted W^C, and the present value of damages that arise when the control policy is implemented, denoted W^I, plus the cost of policy implementation $C(\bar{x}, r_1)$. W^C is the expected net present value of damage with spread rate r_0 plus the value of taking a "wait and see" approach, the option value. This option value represents the value of delaying policy implementation to preserve the ability to respond to new information about the spread of the species and the economic consequences of that spread. The option value is terminated when the policy is implemented making it an additional opportunity cost of the risk eduction policy. The decision rule is simply to implement the policy if $W^I + C(\bar{x},\ \ r_i) \leq W^C$ where $C = \upsilon \bar{x} (r_0 - r_1)^2$ with $\upsilon > 0$.

The relative values of W^I and W^C depend on the current state of the world described by x and γ. This implies there is an endogenous threshold level $x^* = x(t_1, \gamma) > 0$. Implementing the risk reduction policy at this point in the invasion will minimize the expected discounted damages net of policy costs. However, as the magnitude of r_1 is a choice affecting the value of W^I, this threshold is influenced by the stringency of the policy - timing and stringency of the policy decision are thus linked.

Complete irreversibility implies a strict commitment. It is characteristic of the construction of physical barriers or the release of predatory species that may be impossible to remove from the environment. This complete irreversibility makes implementing the risk reduction policy akin to an optimal stopping problem. The policy maker faces an obligation to the flow of pre-policy damages. The obligation is treated as an asset whose value W^C must be optimally managed (i.e., minimized).

The unknown continuation value function can be found explicitly by employing dynamic programming with Bellman equation

$$\rho W^C = D_0 e^{\gamma(x - x_0)} + \frac{E_t\left(dW^C\right)}{dt}$$
$$= D_0 e^{\gamma(x - x_0)} + r_0 \frac{\partial W^C}{\partial x} + \frac{1}{2} s^2 \frac{\partial^2 W^C}{\partial x^2} + \frac{1}{2} \delta^2 \gamma^2 \frac{\partial^2 W^C}{\partial \gamma^2} + \sigma s \delta \gamma \frac{\partial^2 W^C}{\partial \gamma \partial x} \tag{A.2}$$

The left-hand side is the return a decision maker would require to delay policy implementation over the time interval *dt*. The right-hand side is the expected return from delaying policy implementation over the interval *dt*. The Bellman equation acts as an equilibrium condition ensuring a willingness to delay prior to policy implementation.

Three optimality conditions are used to solve for the optimal values of x^* and r_1 as well as the unknown option value. The first condition is the Bellman Eq. (A.2). The second condition is the well-known value matching condition (Dixit and Pindyck 1994). Value matching ensures the continuation value equals the termination value at x^*: $W^C[x^*] = W^I[x^*, r_1] + C$. The third condition required for a solution is the optimality condition for the level of r_1 which minimizes the implementation value function $W^I[x^*, r_1]$.

Many risk reduction policies allow a portion of the policy cost to be recouped by canceling the policy at some future time. For example, quarantines, trade restrictions, and control "programs" that fund continuous control actions require fixed costs but may be terminated or adjusted in the future in response to new information. This moves the problem from one of optimal stopping to optimal switching. According to Brekke and Øksendal (1994), the optimal switching problem can be rewritten as a set of variational inequalities. Prior to policy implementation (regime $R=1$), the continuation value function and the policy implementation curve $x^*(t_1, \gamma)$ satisfy the following Bellman equation

$$\rho W^C \le D_0 e^{\gamma(x-x_0)} + r_0 \frac{\partial W^C}{\partial x} + \frac{1}{2} s^2 \frac{\partial^2 W^C}{\partial x^2} + \frac{1}{2}\delta^2\gamma^2 \frac{\partial^2 W^C}{\partial \gamma^2} + \sigma s \delta \gamma \frac{\partial^2 W^C}{\partial \gamma \partial x} \tag{A.3}$$

and value matching condition

$$W^C[\gamma(t), x(t)] \le W^I[\gamma(t), x(t), r_1] + C \tag{A.4}$$

with one of the conditions satisfied at each point in the state space of $xx\ tt$ and $\gamma\gamma\ (tt\ )$. If (A.3) holds as an equality, it is optimal to delay policy implementation (remain in regime $R=1$). If (A.4) holds as an equality, it is optimal to implement the policy immediately (switch to regime $R=2$). The policy implementation curve is the set of points where both conditions are met.

With a policy currently enacted ($R=2$), the implementation value function and policy cancelation curve $x^*(T_1, \gamma)$ satisfy

$$\rho W^C \le D_0 e^{\gamma(x-x_0)} + r_1 \frac{\partial W^I}{\partial x} + \frac{1}{2} s^2 \frac{\partial^2 W^I}{\partial x^2} + \frac{1}{2}\delta^2\gamma^2 \frac{\partial^2 W^I}{\partial \gamma^2} + \sigma s \delta \gamma \frac{\partial^2 W^I}{\partial \gamma \partial x} \tag{A.5}$$

and

$$W^I[\gamma(t), x(t), r_1] \le W^C[\gamma(t), x(t)] - kC \tag{A.6}$$

where k is the proportion of the policy cost recouped if the policy is cancelled. If (A.5) holds as an equality, it is optimal to continue with the policy (remain in regime 2). If (A.6) holds as an equality, it is optimal to cancel the policy (switch to regime 1). With complete irreversibility, W^C includes an option value that delays policy implementation. With partial irreversibility, W^I includes an additional option value associated with canceling the policy.

The multi-dimensional nature of the state space and the dual policy regimes require numerical methods to approximate the unknown value functions (Judd, 1998; Miranda and Fackler 2002). We approximate $W^C[\gamma(t), x(t)]$ and $W^I[\gamma(t), x(t), r_1]$ over a subset of the state space using piecewise linear basis functions (Balikcioglu et al., 2011; Marten and Moore, 2011). The approximation procedure solves for the $2 \times n^2$ basis function coefficients which satisfy (A.3)–(A.6) and relevant boundary conditions at a set of $n=300$ nodal points spread evenly over the two-dimensional state space extending from 0 to 0.004 in the γ dimension and from 0 to 2000 in the x dimension. Specifically, the unknown value function is approximated with a linear spline constructed using upwind finite difference approximations. For more information see Miranda and Fackler (p. 129, 2002). The boundary conditions are $W^C[0,\ x(t)]=0$, $W^C[\gamma(t),\ 0]=0$, and $W^C[\gamma(t),\ \overline{x}] = \frac{D_0 e^{\gamma(\overline{x}-x_0)}}{\rho}$. The first two ensure that the value of controlling an invasive species that has not invaded or has caused no damage is 0. The second boundary condition arises as the option value goes to 0 as invaded range approaches its upper bound. The final condition required for a solution is first-order condition for r_1. The resulting complementarity problem is solved in Matlab using the smoothing-Newton root finding method (Qi and Liao, 1999). The implementation curve x is a set of $n=300$ points where these conditions are met. Increasing the number of nodal points beyond 300 or extending the state space in either the $x(t)$ or $\gamma(t)$ directions does not alter our general results.

References

Allsopp, P. G. (1996). Japanese Beetle, Popillia japonica Newman (Coleoptera: Scarabaeidae): rate of movement and potential distribution of an immigrant species. *The Coleopterists Bulletin, 50*, 81–95.

Andow, D. A., Kareiva, P. M., Levin, S. A., & Okubo, A. (1990). Spread of invading organisms. *Landscape Ecology, 4*, 177–188.

Andow, D. A., Kareiva, P. M., Levin, S. A., & Okubo, A. (1993). Spread of invading organisms: Patterns of spread. In K. C. Kim & B. A. McPheron (Eds.), *Evolution of insect pests: Patterns of variation*. New York: Wiley.

Archer, D. W., & Shogren, J. F. (1996). Endogenous risk in weed control management. *Agricultural Economics, 14*, 103–122.

Balikcioglu, M., Fackler, P. L., & Pindyck, R. S. (2011). Solving optimal timing problems in environmental economics. *Resource and Energy Economics 33*(3), 761–768.

Bernanke, B. S. (1983). Irreversibility, uncertainty, and cyclical investment. *Quarterly Journal of Economics, 90*, 85–106.

Brekke, K., & Øksendal, B. (1994). Optimal switching in an economic activity under uncertainty. *SIAM Journal on Control and Optimization, 32*, 1021–1036.

Chichilnisky, G., & Heal, G. (1993). Global environmental risks. *The Journal of Economic Perspectives, 7*(4), 65–86.

Davis, M. A., et al. (2011). Don't judge species on their origins. *Nature, 474*, 153–154.

Didham, R. K., Tylianakis, J. M., Hutchison, M. A., Ewers, R. M., & Gemmell, N. J. (2005). Are invasive species the drivers of ecological change? *Trends in Ecology & Evolution, 20*(9), 470–474.

Dixit, A. K., & Pindyck, R. S. (1994). *Investment under uncertainty.* Princeton: Princeton University Press.

Feder, G., & Regev, U. (1975). Biological interactions and environmental effects in the economics of pest control. *Journal of Environmental Economics and Management, 2*, 75–91.

Fisher, A. C., & Hanemann, W. M. (1993). Assessing climate change risks: Valuation of effects. In J. Darmstadter (Ed.), *Assessing climate change risks*. DC: Resources for the Future.

Holmes, T. P., Aukem, J. E., Von Holle, B., Liebhold, A., Sills, E., Holmes, T. P., Aukem, J. E., Von Holle, B., Liebhold, A., & Sills, E. (2009). Economic impacts of invasive species in forests: past, present, and future. *Annals of the New York Academy of Sciences, 1162*, 18–38. The Year in Ecology and Conservation Biology, 2009.

Jerde, C. L., Lodge, D. M., Chadderton, W. L., McNulty, J., Moy, P., Mysorekar, S., M. A. R. (2014). *Guiding early detection of incipient* *invasions of fishes into the Laurentian Great Lakes through the Chicago Sanitary and Ship Canal*. Working Paper. University of Notre Dame.

Judd, K. (1998). Numerical Methods in Economics, MIT Press.

Knowler, D., & Barbier, E. (2000). The economics of an invading species: A theoretical model and case study application. In C. Perrings, M. Williamson, & S. Dalmazzone (Eds.), *The economics of biological invasions*. Cheltenham: Edward Elgar.

Lichtenberg, E., & Zilberman, D. (1986). The econometrics of damage control: why specification matters. *American Journal of Agricultural Economics 68*(2), 261–273.

Liebhold, A. M., & Tobin, P. C. (2008). Population ecology of insect invasions and their management. *Annual Review of Entomology, 53*, 387–408.

Lodge, D. M. (2001). Responses of lake biodiversity to global changes. In F. S. Chapin III, O. E. Sala, & E. Huber-Sannwald (Eds.), *Future scenarios of global biodiversity* (Vol. 8, pp. 277–312). New York: Springer.

Mack, R. N., Simberloff, D., Lonsdale, W. M., Evans, H., Clout, M., & Bazzaz, F. A. (2000). Biotic invasions: causes, epidemiology, global consequences and control. *Ecological Applications, 10*, 689–710.

Marten, A. L., & Moore C. C. (2011). An options based bioeconomic model for biological and chemical control of invasive species. *Ecological Economics 70*(11), 2050–2061.

Miranda, M., & Fackler, P. L. (2002). *Applied computational economics and finance*. The MIT Press.

Moore, C., & Mitchell, C. (1997). Aedesalbopictus in the United States: ten year presence and public health implications. *Emerging Infectious Diseases, 3*, 329–334.

Perrings, C., Williamson, M., Barbier, E. B., Delfino, D., Dalmazzone, S., Shogren, J., Simmons, P., & Watkinson, A. (2002). Biological invasion risks and the public good: an economic perspective. *Conservation Ecology, 6*(1).

Pindyck, R. (2007). Uncertainty in environmental economics. *Review of Environmental Economics and Policy, 1*(1), 45–65.

Poland, T. M., & McCullough, D. G. (2006). Emerald Ash Borer: invasion of the urban forest and the threat to North Americas ash resource. *Journal of Forestry, 104*, 118–124.

Qi, H., & Liao L. (1999). A smoothing newton method for extended vertical linear complementarity problems. *SIAM Journal on Matrix Analysis and Applications 21*(1), 45–66.

Rothlisberger, J. D., Lodge, D. M., Cooke, R. M., & Finnoff, D. (2010). Future of the binational Laurentian Great Lakes fisheries: environmentally and culturally driven declines. *Frontiers in Ecology and the Environment, 8*(5), 239–244.

Sharov, A. A., & Liebhold, A. M. (1998). Bioeconomics of managing the spread of exotic pest species with barrier zones. *Ecological Applications, 8*, 833–845.

Shogren, J. (2000). Risk reduction strategies against the explosive invader. The economics of biological Invasions. In C. Perrings, M. Williamson, & S. Dalmazzone (Eds), Edward Elgar Publishing, Cheltenham, UK, pp. 56–69.

Simberloff, D., et al. (2013). Impacts of biological invasions: what's what and the way forward. *Trends in Ecology & Evolution, 28*(1), 58–66.

Sims, C., & Finnoff, D. (2013). When is a “wait and see” approach to invasive species justified? *Resource and Energy Economics, 35*, 235–255.

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