Economic Incentives for Controlling Trade-Related Biological Invasions in the Great Lakes

Richard D. Horan and Frank Lupi

Ballast water from commercial ships engaged in international trade has been implicated as the primary invasion pathway in over 60 percent of new introductions of invasive alien species (IAS) in the Great Lakes since 1960. Recent policies have recognized that IAS are a form of biological pollution and have become focused on preventing new introductions. Given that emissions-based incentives are infeasible for the case of biological emissions, we investigate the cost-effectiveness of various performance proxy-based and technology-based economic incentives to reduce the threat of new invasions of Ponto-Caspian species in the Great Lakes.

Key Words: aquatic nuisance species, ballast water, uncertainty, risk management, performance-based incentives, environmental subsidies

The economic and environmental impacts of invasive alien species (IAS)—species that establish and spread in ecosystems to which they are not native—can be significant (Perrings, Williamson, and Dalmazzone 2000). Invasive alien species are argued to be the second-most important cause of biodiversity loss worldwide (Holmes 1998, U.S. Environmental Protection Agency 2001) by, for example, out-competing or preying upon native species. In addition, IAS can cause or spread diseases to cultivated plants, livestock, and human populations, and they often encroach on, damage, or degrade assets (e.g., power plants, boats, piers, and reservoirs). In the Great Lakes, at least 145 IAS have been introduced since the 1830s. Many early invasions such as sea lamprey and alewife were associated with the opening of shipping canals that, although they facilitated trade, removed natural barriers. About one-third of the documented invasives in the Great Lakes have been introduced during the past thirty years, in part as a result of increased trade-related shipping following the opening of the St. Lawrence Seaway (Michigan Department of Environmental Quality [MDEQ] 1996, Great Lakes Commission 2000). Although only about 10 percent of introduced species are suspected of having caused any damage (Mills et al. 1993), the impacts that have occurred are extensive (U.S. Environmental Protection Agency 2001, MDEQ 1996, Coscarelli and Bankard 1999, Reeves 1999). The zebra mussel alone is predicted to cost society $5 billion over the next decade (MDEQ 1996).

Until recently, most IAS management efforts focused on post-invasion control or eradication (Lupi, Hoehn, and Christie 2003). But there is now an increasing emphasis on prevention (National Research Council Committee on Ships' Ballast Operations [NRC] 1996). This shift in focus has possibly occurred because most new IAS introductions are now recognized as a form of "biological pollution," with the risk of new invasions being an endogenous function of hu-

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This paper was presented at the International Trade and the Environment Workshop sponsored by the Northeastern Agricultural and Resource Economics Association and the U.S. Environmental Protection Agency, National Center for Environmental Economics, in Halifax, Nova Scotia, on June 23, 2004. The views expressed in this paper are those of the authors', who are solely responsible for any errors, and do not necessarily represent those of the U.S. Environmental Protection Agency.

The research reported here was supported by the Michigan Agricultural Experiment Station and the Michigan Sea Grant College Program, Project No. R/EP9 under Grant No. NA76RG0133 from the Office of Sea Grant, National Oceanic and Atmospheric Administration (NOAA), U.S. Department of Commerce, and funds from the state of Michigan. The U.S. Government is authorized to produce and redistribute reprints for government purposes notwithstanding any copyright notation appearing herein.

man activities such as trade and travel. For example, commercial shipping in the Great Lakes has been implicated in over 60 percent of new introductions since 1960 (Mills et al. 1993), with the primary pathway being ballast water.[1] Ballast water is often carried in the hulls of ships to maintain stability and hull integrity. Ballast water levels are altered in ports to adjust for changes in cargo, or in transit to improve stability or to change hull depth. During ballast water exchange, species may be inadvertently transferred into or out of a ship. To understand the risk in the Great Lakes, consider that each year, approximately 200–300 ocean-going vessels enter the Great Lakes, and these vessels account for 400–600 round trips in and out of the region. Over 70 percent of these vessels are engaged in the "triangle trade" route, which moves grain, coal, and ore from the Great Lakes to the Mediterranean, and then on to Northern Europe (Reeves 1999). Major overseas markets are Western Europe, the Baltics, the Mediterranean, and the Middle East. This "triangle trade" route involving the Ponto-Caspian region has supplied approximately 70 percent of Great Lakes invaders between 1985 and 2000 (Reid and Orlova 2002). Thus, in the Great Lakes there is increased emphasis on the prevention of trade-related biological invasions associated with ballast water.

Mandating oceanic ballast water exchange has been the predominant preventive approach to IAS in the Great Lakes, beginning in 1993 with the implementation of the U.S. Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990, and later by the U.S. National Invasive Species Act of 1996 and the Canadian Shipping Act of 1998 (Reeves 1999).[2] However, the success of oceanic exchange programs is imperfect because new introductions have occurred since 1993 and because there are known limitations to the practice of ballast water exchange.[3]

The limitations of current regulatory approaches are now generally recognized, as is the need for new policy options that promote both safety and cost-effectiveness (NRC 1996, Rigby and Taylor 2001). Economists traditionally prescribe emissions-based incentives to encourage reductions in emissions, or emissions-based regulatory standards to mandate the reductions, at least when dealing with conventional pollutants. But such emissions-based approaches are not applicable in the IAS case. Two features of vessels' biological emissions complicate matters (Horan et al. 2002). First, not every vessel will actually emit a species, yet ex ante each vessel is a potential emitter, and so society is expected to benefit from all vessels undertaking biosecurity actions to reduce the probability of an invasion. Second, biological emissions are highly stochastic and essentially unobservable given current monitoring technologies—much like nonpoint source pollution (Shortle and Dunn 1986). Consequently, there is no obvious method for directly observing or otherwise indirectly measuring whether a vessel caused an introduction.

Because IAS emissions cannot be measured cost-effectively, policies must be based on some alternative compliance measure. Options include

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[1] Solid ballast and hull-fouling were once important causes of introductions. But solid ballast is now seldom used, and steel hulls combined with anti-fouling techniques have greatly diminished introductions due to hull-fouling.

[2] All vessels entering the Great Lakes with ballast on board (BOB) are required to exchange ballast at sea beyond the Exclusive Economic Zone (EEZ) in a depth of at least 2,000 m, so that ballast salinity levels are raised to 30 parts per trillion (ppt) (ocean salinity levels range between 34 to 36 ppt). This ballast must then be retained for the duration of the voyage into the Great Lakes (NRC 1996). Ballast retention is the primary prevention measure, while oceanic exchange is secondary. For instance, if some ballast exchange were to take place in the Great Lakes—for example, in order to pass through a lock or for safety reasons—it is thought that organisms that might survive in the fresh or brackish waters of the Great Lakes could not survive in the high saline levels that would result from the oceanic exchange, and vice versa (Rigby and Taylor 2001). The primary purpose of increasing salinity levels in the tanks is not necessarily to kill freshwater organisms in the tanks, although this is a secondary effect. Rather, the intent is a 100 percent exchange of water and organisms, as it is believed that oceanic organisms that could survive in the Great Lakes would have already migrated there long ago. Hence, oceanic ballast exchange represents an exchange of organisms across two distinct ecological zones by which reciprocal introductions do not occur (Reeves 1999). In 2001, the State of Michigan enacted Public Act 114, which requires reporting of ballast management and ties eligibility for state grants, awards, and loans to satisfactory ballast management.

[3] First, a vessel does not have to conduct an oceanic exchange if it is deemed to be unsafe. Hull stress increases and stability decreases during an oceanic exchange (Reeves 1999), and it is not uncommon for captains to opt out of an exchange for safety reasons. Second, ballast exchange typically does not result in a 100 percent replacement of all ballast water and sludge (Rigby and Taylor 2001, Reeves 1999). Many organisms are left in the tanks, and the high salinity levels do not kill them all (Rigby and Taylor 2001, Reeves 1999). A third limitation is that the regulations do not apply to vessels entering the Great Lakes with no ballast on board (NOBOB)—vessels that typically carry tons of unpumpable sludge at the bottom of their hulls. This sludge may be home to many foreign organisms that can be introduced when the ship initially takes on ballast at its first stop in the Great Lakes and/or when it exchanges ballast at subsequent ports. Farley (1996) estimates that ships entering the Great Lakes with a NOBOB status accounted for 84 percent of the discharged ballast containing foreign water in 1995.

specific biosecurity choices made by a vessel, or a performance proxy—estimates of emissions, where the estimates are derived from a model that relates vessel characteristics and observable biosecurity investments to emissions estimates. Performance proxies are used in other contexts, such as in existing point-nonpoint water quality trading programs, where reductions in nonpoint loads are estimated based on a farmer's management practices (Horan 2001). In the present context an emissions estimate is more accurately described as the probability of a species introduction by the vessel. California has implemented a permit program that is essentially based on such a performance proxy. Vessels are issued permits based on ballast exchange, but they are free to adopt alternative technologies that achieve similar outcomes in terms of risk reduction (Karaminas et al. 2000).

In this paper we examine the relative efficiency of various economic incentives for reducing the risk of IAS invasions in the Great Lakes, where the type of incentive differs according to the compliance measure used. Specifically, we consider subsidies to reduce the risk of an invasion and subsidies to implement certain biosecurity measures.[4] The particular forms of these subsidies are not first-best in the sense of minimizing the expected cost of reducing invasion risks. Such subsidies, as we describe below, would be excessively complex to administer because they would have to be tailored to individual vessels in accordance with each vessel's marginal environmental impacts. Rather, we consider second-best subsidies that are not tailored to individual vessels and, in the case of biosecurity subsidies, do not target each possible biosecurity choice that a vessel could make. A lack of differential targeting reduces efficiency but may reduce administrative or other transaction costs associated with implementation, and it may circumvent the political or legal consequences of treating vessels differentially, as uniform subsidy rates are typically viewed as fairer. These same issues arise in other pollution contexts, most notably the control of nonpoint source pollution (Shortle and Abler 1997, Helfand and House 1995).

[4] Subsidies (as opposed to taxes) are considered because shipping in the Great Lakes is a highly subsidized industry, and there is fear that costly regulatory measures could significantly reduce shipping volume in the Great Lakes and thereby hurt local industries that are dependent upon this sector (Reeves 1999).

A Model of IAS Invasions

The theoretical model we present is based on the probabilistic model of Horan et al. (2002). All vessels entering the Great Lakes are considered to be potential carriers, or vectors, of biological pollution. Each vessel makes various biosecurity choices that affect the likelihood of species introductions. For commercial shipping, biosecurity choices might include the effort devoted to ballast water exchange, the number and location of stops, the time spent at sea, and the use of treatments such as biocides, filtering, and heat. Denote the ith vessel's choices by the $(1 \times m)$ input vector $\mathbf{x}_i$ (with jth element x_{ij}). Although in our empirical model many inputs will be "lumpy" investments, we treat them as continuous for now. Vessel i's private biological pollution control costs (not including economic damages to other parties) are a function of the vessel's biosecurity choices, $c_i(\mathbf{x}_i)$.

Denote e_{is} as the biomass of species s (s = 1,...,S) that is introduced in the given habitat by vessel i. Vessels cannot control e_{is} with certainty. Introductions are random due to the influence of stochastic variables not directly under the vessel's control (e.g., environmental drivers), although the probability of a particular level of biomass emissions is conditional on the vessel's biosecurity choices. The probability that e_{is} is introduced, conditional on input choices and vessel characteristics ($\mathbf{b}_i$), is $p_{is}(e_{is}|\ \mathbf{x}_i, \mathbf{b}_i)$.

An introduced species may or may not establish (invade). Conditions, such as the *in situ* control regime (taken as given here), must be right for a successful invasion. Successful invasions occur with some probability, conditional on the scale of the introduction and also location and habitat characteristics (e.g., predators and food sources), denoted by $\mathbf{h}$. Denote the probability that introduction e_{is} leads to a successful invasion by $\Pr_s(survival\,|\,e_{is}, \mathbf{x}_i, \mathbf{h})$, which is increasing in e_{is}. The biosecurity choices a vessel makes influences this probability to the extent that these choices influence the quality of an introduction. For instance, the state of health of an introduced species may be directly influenced by a vessel's biosecurity choices. Thus, for discrete levels of e_{is}, the probability that introductions of species s by vessel i lead to an invasion is

$$q_{is}(\mathbf{x}_i,\mathbf{b}_i,\mathbf{h}) = \sum_{e_{is}} \Pr_s(survival \mid e_{is},\mathbf{x}_i,\mathbf{h})\, p_{is}(e_{is} \mid \mathbf{x}_i,\mathbf{b}_i) \,.$$

This specification assumes that invasions arise from particular vessels and that the probability of an invasion from one vessel is independent of any introductions by other vessels. Although this may be a simplification for some cases in which the alien population depends on a large number of introductions to become established in the new habitat, it is a realistic assumption for species that are fairly well-suited to the new ecosystem and that can establish with only small numbers.[5]

Because a species is able to proliferate *in situ* once it has invaded and because new invasions tend to start out small, any further invasions of the same species are likely to have small, if not negligible, impacts on damages. Hence, we focus our concern on the initial invasion, which is also consistent with the policy focus on prevention.[6] This property, that subsequent invasions have negligible marginal damages, distinguishes the biological pollution problem from many classic pollution problems in which the current level of emissions matters. To reflect this, a species invasion is modeled as a Bernoulli event: an invasion either occurs or it does not occur. The probability of an invasion of species s from any one of n vessels is

$$(1)\quad P_s(\mathbf{x}_1,\ldots,\mathbf{x}_n) = P_s(Z_s \geq 1) = 1 - P(Z_s = 0) = 1 - \prod_{i=1}^{n}(1 - q_{is}(\mathbf{x}_i,\mathbf{b}_i,\mathbf{h})),$$

where Z_s represents the number of times that species s invades a given ecosystem. The invasion probability P_s is decreasing in biosecurity measures that make introductions less likely, and increasing in biosecurity measures that make introductions more likely. The probability P_s is also increasing in the number of vessels. As $n \to \infty$, an invasion becomes a virtual certainty (i.e., $P_s \to 1$).

Cost-Effective Management of Biological Pollution

Ex-ante efficient biosecurity measures minimize the expected social cost of biological pollution and its control.[7] But the reality is that the complete state space and associated probabilities, for both potential invaders and potential damages from known and unknown potential invaders, cannot be identified ex ante. A reasonable alternative to the ex-ante efficient problem is to pursue a cost-effective allocation of biosecurity controls based on a smaller set of probabilistic information that can be developed subjectively. Cost-effectiveness is a standard benchmark for analyzing pollution control policies, and identifying the cost-effective solutions is a necessary input to any balancing of the social costs of biological pollution.

An ex-ante cost-effective allocation of biosecurity measures minimizes the expected cost of biological pollution control,

$$TC = \sum_{i=1}^{n} c_i(\mathbf{x}_i)\,,$$

subject to environmental constraints related to invasions. Useful notions of cost-effectiveness for biological pollution control must consider the unobservable and stochastic nature of species introductions and the stochastic nature of invasions. Hence, we use probabilistic constraints of the form

$$(2)\qquad P_s(\mathbf{x}_1,\ldots,\mathbf{x}_n) \leq \Phi_s\,,\ \forall s \in \hat{S}\,,$$

[5] To the best of our knowledge, there is no statistical evidence to support the notion that an invasion is more likely to occur if there are introductions by multiple vessels. In the case of the Great Lakes, for example, a particular species introduction via ballast water is typically very small, both individually (e.g., measured in microns) and collectively (i.e., their relative density in the water column is minuscule). To say that multiple introductions help an introduced population to establish or invade means that these minuscule populations would have to find each other within this vast ecosystem. It seems more likely that a small, fast-growing species could proliferate with only a single introduction. This is the type of species we consider, and it is the type that Kolar and Lodge (2002) and Ricciardi and Rasmussen (1998) identify as the most likely to invade.

[6] The model is static, and so the issue of eradication and re-invasion does not come up. But it is probably not a serious concern because, to our knowledge, no Great Lakes invader has ever been successfully eradicated. Once an invasion occurs, it is typically assumed that the species is there to stay—even when there are substantial control efforts in place. In the Great Lakes, few invasives have viable control programs. Where successful control programs do exist, for example for sea lamprey (Lupi, Hoehn, and Christie 2003), no means of eradication is known. For settings where eradication and reinvasion are likely, an alternative formulation of invasion risk would be necessary.

[7] Perrings, Williamson, and Dalmazzone (2000) point out that the probabilities of invasion may be quite small and the associated damages quite large, which may give rise to non-convexities. Moreover, managers might not make decisions according to expected utility theory in such instances, instead making decisions based on a reference point. Shackle's (1969) theory of decision making under uncertainty (ignorance) is consistent with a reference point approach, and Horan et al. (2002) illustrate that making decisions in this fashion can be equivalent to using the expected utility model with subjective weights applied to the reference point.

where $0 \leq \Phi_s \leq 1$ and $\hat{S}$ (with $\hat{S} \subseteq S$) represents a set of target species upon which controls are based. The target probability levels (Φ_s) may be chosen based on political and fiscal considerations related to expected industry control costs and to the government budgetary impacts of subsidy outlays. This "safety-first" approach, which has been addressed in research on the control of stochastic pollution (Beavis and Walker 1983, Lichtenberg and Zilberman 1988, Lichtenberg, Zilberman, and Bogen 1989), is consistent with the goals of the International Maritime Organization (IMO). The IMO has accepted that reducing risk, rather than eliminating it, should form the basis for new mandatory ballast water management instruments (Rigby and Taylor 2001).[8] The focus on target species is an approach that has been formally adopted by the Australian Ballast Water Management Council (Rigby and Taylor 2001). The target species may be chosen based on the perceived likelihood of their invasion and of ensuing damages (see Kolar and Lodge [2002] and Ricciardi and Rasmussen [1998] for more on the issue of estimating invasion risks). Moreover, since biosecurity controls are often not specific to any one species, the set of target species may be chosen to represent certain classes of species having similar biological/ecological traits. This practice may help prevent invasions by similar species whose potential to invade is not suspected or known ex ante.

The necessary conditions for an interior, cost-effective solution can be written as

(3)
$$\frac{\partial c_i}{\partial x_{ij}} = \sum_{s=1}^{S} \lambda_s \frac{\partial P_s}{\partial x_{ij}} = \sum_{s=1}^{S} \lambda_s (1 - P_s^{-i}) \frac{\partial q_{is}}{\partial x_{ij}}, \quad \forall i, j ,$$

where λ_s is the shadow value of the sth constraint in (2) and

$$P_s^{-i} = 1 - \prod_{k \neq i} (1 - q_{ks})$$

is the aggregate probability that species s will invade from any vessel other than vessel i. Condition (3) requires that, in the cost-effective solution, the marginal cost of undertaking a particular action (the left-hand side) equals the imputed marginal value of reduced risk stemming from the action (the right-hand side). The marginal value of reduced risk is vessel-specific and will depend on vessel i's actions as well as on the actions of all the other vessels.

Incentive Mechanisms

Using condition (3), it would be straightforward to show that first-best subsidies, i.e., cost-effective subsidies that result in condition (3) being satisfied, could be developed based on reducing the risk of invasion or based on biosecurity investments. We do not derive these optimal subsidies here, but rather describe some of their basic features (see Shortle and Abler [1997] for more on first-best instrument design in the closely related case of nonpoint source pollution). In the case of first-best risk-reduction subsidies, the right-hand side of (3) implies that the subsidy rates would have to be set at vessel-specific rates to account for the distinct marginal environmental impacts that each vessel's risk has on aggregate risk, and the subsidies would have to be applied separately for each species of concern. In the case of first-best biosecurity-based or input-based subsidies, the right-hand side of (3) suggests that the subsidy rates would have to be vessel-specific and applied to each input that might influence risk. Each of these first-best options is administratively complex. We therefore consider two types of "second-best" subsidy programs, where some efficiency is sacrificed for greater administrative ease and, presumably, lower transaction costs.

The first type of subsidy we consider is based on a reduction in the probability that a vessel introduces *any* species, $q_i(\mathbf{x}_i, \mathbf{b}_i, \mathbf{h})$. The restriction of a single probabilistic compliance measure (as opposed to separate compliance measures for each species of concern) reduces efficiency, but such a system would be easier to implement and manage. The risk-reduction subsidy is of the form $\max\{\rho(q_i^0 - q_i), 0\}$, where ρ is the subsidy rate for reducing the risk of an invasion, and q_i^0 is a baseline level of risk from which actual risk must be reduced in order for the vessel to receive a payment. Assume that $q_i^0 = q_i(\mathbf{x}_i^0)$ is the initial risk posed by vessel i in the unregulated equilib-

[8] Technically, the IMO is promoting the use of the precautionary principle of minimizing risk (Rigby and Taylor 2001). However, it also realizes that risk cannot be completely eliminated, suggesting that it understands that the costs of attaining such an objective would be too high. Our focus on a safety-first criterion therefore appears to be consistent with their objectives.

rium where it makes input choices $\mathbf{x}_i^0$. The restriction of uniformly applying the subsidy rate ρ across all vessels implies certain inefficiencies due to the fact that each vessel has a different marginal environmental impact.

The risk-reduction subsidy we have described is voluntary. Vessels will evaluate whether they are better off with or without the subsidy, and they will make biosecurity investments only if the subsidies cover the associated costs. If there were no fixed costs and if the marginal cost of risk reduction was non-decreasing, then vessels would generally be willing to invest some effort in biosecurity. For instance, in Figure 1 the marginal cost of risk reduction, *MC*, is increasing. Profit-maximizing vessels choose the level of risk reduction such that $MC = \rho$. Given that ρ is set at a constant level, the total subsidy payment, area $R + TVC$, covers the vessel's biosecurity costs, *TVC*, and also provides surplus rents in the amount of *R*. Now suppose fixed costs are relevant. The rents must exceed these fixed costs if the vessel is to accept the subsidy and undertake the biosecurity investments.

The second type of subsidy that we consider is an input-based subsidy of the form $\varsigma_j x_{ij}$, where ς_j is the subsidy rate for biosecurity choice *j*. The subsidy rate is applied uniformly to all vessels, and subsidies are applied only to a subset of easily observable inputs. Both of these features ease administration costs but decrease efficiency. As above, fixed costs may discourage some vessels from participating in the subsidy program.

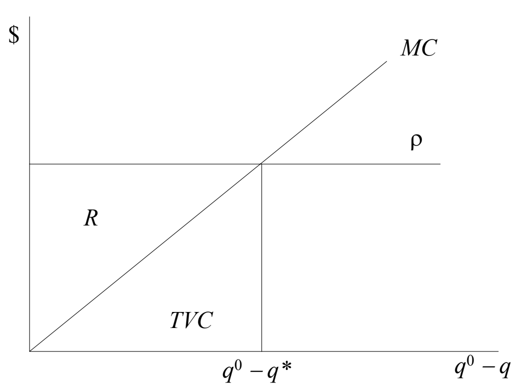

Figure 1. Subsidy Rents, *R*, Must Cover Fixed Costs to Induce Participation

An Application to Great Lakes Shipping

We investigate several incentive policies numerically using data on transoceanic vessels that operate in the Great Lakes. The empirical application builds on the effort of Horan and Lupi (forthcoming), who investigated the potential use of tradable risk permits for preventing IAS. Although we have made every effort to calibrate the model realistically, research on potential Great Lakes invaders is still evolving and is at a fairly early stage, so knowledge of many parameters is somewhat limited. The following analysis is therefore best viewed as a numerical example rather than a true reflection of reality. Nonetheless, the results shed light on the economic issues surrounding the prevention of new invasions in the Great Lakes.

In the application, we use official statistics on a subset of 315 transoceanic vessels ("salties") that travel the St. Lawrence Seaway (U.S. Army Corps of Engineers 2002). The data represent the majority of Seaway vessels and are used to develop 31 "classes" of vessels distinguished by their deadweight tonnage (DWT). Costs and probabilities are aggregated within each vessel class using a microparameter approach (Just and Antle 1990). In addition to the transoceanic vessels, a number of other vessels, known as "lakers", operate exclusively on the Great Lakes. While lakers may be responsible for spreading IAS within the Great Lakes, they are not responsible for new introductions into the region. Our focus is on vessels that pose a threat of new introductions.

Since Reeves (1999) reports that vessels carry 15–30 percent of DWT in ballast water, 30 percent of DWT is used as the value of ballast water capacity, denoted b_i for the *i*th vessel, although it is not assumed that each vessel enters the Seaway carrying that much ballast. Instead, this value represents each vessel's potential ballast, and the vessel may enter or leave the Seaway with all of this ballast or a fraction thereof. Because a tank can never be fully emptied (i.e., $b_i > 0$), this value also accounts for the unpumpable sludge in a vessel's tanks. This sludge factor is relevant for the majority of vessels entering the Seaway with no ballast on board (NOBOB) (Reeves 1999).

Although the target species concept has not been adopted in the Great Lakes as it has been in Australia (Rigby and Taylor 2001), some potential invaders of concern have been identified.

Many of these potential invaders are from the Ponto-Caspian region (Ricciardi and Rasmussen 1998, Kolar and Lodge 2002). We focus on three Ponto-Caspian species that have been identified as likely invaders capable of causing extensive damage (Ricciardi and Rasmussen 1998, Kolar and Lodge 2002): *Corophium spp.* (a small amphipod), mysids (a small shrimp), and *Clupeonella caspia* (a small fish). These three species also span a range of organism "types" and sizes of likely invaders; some details of their life histories are presented in Table 1.

Denote k_{is} as the base probability that vessel i transports species s into the Great Lakes when the vessel adopts no biosecurity measures. In the model, this value is directly proportional to the ballast (or sludge) the vessel carries, $k_{is} = \alpha_{is} b_i$, where $\alpha_{is} > 0$ is a parameter: larger vessels are more likely to bring in species, *ceteris paribus*. In general, α_{is} may vary according to the vessel's trade route. However, since most vessels follow the triangle trade route and since we lack detailed information on ports visited and the risks associated with specific ports, this value is set equal for all vessels. Assuming species s is introduced into the environment, the likelihood that the species establishes is denoted β_s. Without any biosecurity efforts, $\beta_s \alpha_{is} b_i$ represents the likelihood of an invasion of species s by vessel i.

Vessels can adopt biosecurity techniques to reduce the chances of an invasion. Filtration reduces the likelihood that species will enter or exit a vessel's ballast tanks. The effectiveness of filtration on species s is denoted by the function $\phi_{Fs}(x_{iF})$, with $x_{iF} \in [0,1]$ and $\phi_{Fs}(0) = 0$, $\phi_{Fs}(1) = \phi^U_{Fs}$, where ϕ^U_{Fs} is an upper bound on ϕ_{Fs}. Here, x_{iF} is an index representing the effort applied to the filtration technology, e.g., by choice of filter mesh sizes. The subscript F, here and elsewhere, represents filtration.

The survival of species in ballast water during transit is affected by in-transit ballast management practices. The most promising in-transit practices are ballast exchange via continuous flushing, reballasting, heat, chemical treatments, and ultraviolet radiation (Rigby and Taylor 2001, Pollutech Environmental Limited [Pollutech] 1996). Reballasting is often considered dangerous; ballast exchange via continuous flushing has been shown to be safer and as effective (Rigby and Taylor 2001). Chemical treatments are usually discouraged due to their high cost and the safety and environmental hazards associated with their use (NRC 1996, Rigby and Taylor 2001, Pollutech 1996). Ultraviolet radiation is not considered a stand-alone technology—it is not usually considered effective unless it is combined with a filtering technology, but, even in combination with filtering, experts disagree on its potential (NRC 1996). Thus, the two in-transit practices we consider are heat and ballast exchange via continuous flushing (henceforth, ballast exchange), which Perakis and Yang (2001) suggest as the most promising practices (along with filtering) for the Great Lakes.

The effectiveness of each biosecurity practice will depend on the level of effort devoted to the practice. For instance, the quantity of ballast exchange depends on the duration of the exchange. The effectiveness of heat depends on temperatures being high enough and applied for sufficient duration to kill all target organisms, which can be difficult and costly (Rigby and Taylor 2001, Pollutech 1996, NRC 1996). As defined above for filtering, define the effectiveness of in-transit practice j on species s by $\phi_{js}(x_{ij})$ (with $x_{ij} \in [0,1]$ and $\phi_{js}(0) = 0$, $\phi_{js}(1) = \phi^U_{js}$, where ϕ^U_{js} is an upper bound on ϕ_{js}).[9]

With this specification for the effectiveness of the three control technologies, the probability that species s invades due to the activities of vessel i is given by

$$q_{is} = \beta_s[1-\phi_{Bs}(x_{iB})][1-\phi_{Hs}(x_{iH})][1-\phi_{Fs}(x_{iF})]k_{is},$$

where the indices B, H, and F represent ballast exchange, heat, and filtering, respectively. For each technology, let $\phi_{js}(x_{ij}) = \mu_{ij} x_{ij}^{\delta_{js}}$ ($j = F,B,H$). The parameters μ_{ij} and δ_{js} were calibrated from reports on the effectiveness of ballast water management practices (Rigby and Taylor 2001) (see Table 1). Although the effectiveness data were not directly developed for our three target species, the data do relate to life history and physiological characteristics possessed by our target species. The parameter β_s is set equal to 0.1 $\forall s$ based on the observation of Perrings et al. (2002),

[9] It is necessary for the regulatory agency to have perfect knowledge of each vessel's actual effort levels in order to accurately gauge whether the vessel is in compliance with its subsidized effort levels. We assume here that it is possible to perfectly monitor effort levels, although in reality monitoring may be difficult and vessels may have incentives to misrepresent their actual effort levels.

Table 1. Effectiveness of Various Ballast Water Management Technologies for Great Lakes Target Species

Target Species	Control Technology		
	Ballast Exchange	Heating	Filtration
Clupeonella caspia	Not generally effective at killing organisms (Rigby and Taylor 2001). Somewhat effective at removing individual organisms from the tanks as the exchange occurs. We assume efficiency equals the proportion of exchange that occurs.	The species *clupeonella cultriventris caspia* naturally occur in temperatures up to 26°C (Aseinova 2003), although this may not be the upper bound for survival. We assume 99 percent efficiency for 40°C ($x_H = 1$) and 90 percent efficiency for 35°C ($x_H = 0.5$).	For *clupeonella cultriventris caspia*, eggs are 1 mm, larvae are 1.3–1.8 mm, and fingerlings are 50–55 mm. Adults average 7.8 cm—much too large to fit through any filter. However, population structures are weighted heavily by newer recruits (Aseinova 2003). Sizes of these younger fish are similar to rotifers and small copepods. We adopt Ribgy and Taylor's (2001) reported removal efficiencies for copepods: we assume 95 percent effectiveness for a 100 μm filter ($x_F = 0.1$) and 99 percent effectiveness for a 25 μm filter ($x_F = 1$).
Corophium spp.		*Corophium curvispinum* have been known to naturally occur in warm lakes up to 31°C (Rajagopal et al. 1999), although this may not be the upper bound for survival. Mortality rates will depend on the ballast water temperature achieved, the time to achieve it, and the duration of heating. Temperatures in excess of 40°C are hard to achieve and maintain in colder waters such as the northern Atlantic. We assume 90 percent efficiency for 40°C ($x_H = 1$) and 50 percent efficiency for 35°C ($x_H = 0.5$).	*Corophium* are marsupial-like amphipods that carry their young in pouches until the eggs hatch. There are many related species. For *Corophium curvispinum*, juveniles are 550 μm in length (Rajagopal et al. 1999) but possibly narrow enough to fit through mesh. Juveniles are up to 1.8 mm and adults average 3.75 mm (Rajagopal et al. 1999). Rigby and Taylor (2001) report removal efficiency of 50 μm to 25 μm filters to be from 80 percent for small rotifers (rotifers usually range in length from 1 mm to 250 mm) and 95 percent for bivalve vetigers. Given the size of juveniles, we assume 60 percent efficiency for the 50 μm filter ($x_F = 0.5$) and 95 percent for the 25 μm filter ($x_F = 1$).
Mysids		The species *Paramysis lacustris* have been known to survive *in situ* in temperatures up to 28°C (Baychorov 1980), although this may not be the upper bound for survival. We assume 95 percent efficiency for 40°C ($x_H = 1$) and 60 percent efficiency for 35°C ($x_H = 0.5$).	Mysids are marsupial-like shrimp that carry their young in pouches until the juvenile stage. There are many related species. For the species *Paramysis lacustris*, adult females range in size from 10 to 14 mm (Baychorov 1980). Sizes of newly released juveniles were not reported, but for the related species *Neomysis Americana* this size averaged 710 μm (Pezzack and Corey 1979). Given that mysids are generally larger than *corophium* and that egg deposition is not a concern for mysids, we use slightly larger removal efficiencies than for *corophium*: 80 percent for $x_F = 0.5$ and 98 percent for $x_F = 1$.

Source: Table adapted from Horan and Lupi (2003).

who note that introduced species often have about a 10 percent chance of establishing a viable population in the new ecosystem. The parameter α_s is calibrated to ensure that, in the absence of any ballast controls, each species has a moderate chance of invading in any given year. Specifically, each species was assumed to have a 10 percent chance of invasion in any specific year in the unregulated base case. Given the limited data, the assumption of a 10 percent annual chance of invasion was made because it corresponds to a 65 percent chance of invasion over the next decade—consistent with the view of scientists who believe an invasion by each of these species is somewhat likely in the near future (Ricciardi and Rasmussen 1998, Kolar and Lodge 2002).

Turning to the private control costs, vessel *i*'s variable control costs are defined by

$$c_i(\mathbf{x}_i) = \sum_j w_{ij} x_{ij} ,$$

where w_{ij} is the constant per unit cost of practice *j* for vessel *i*. Unit costs have been shown to vary by ballast capacity (Rigby and Taylor 2001). We use Rigby and Taylor's cost data for various vessel sizes (Table 2) to calibrate unit costs by vessel size, $w_{ij} = \gamma_{ij} b_i^{\rho_{ij}}$. There is also a fixed capital cost associated with the use of some technologies. Fixed costs (Table 2) also depend on vessel size, $FC_{ij} = \Gamma_{ij} b_i^{\sigma_{ij}}$.[10]

To determine the cost-effective combination of practices, the cost-minimization problem is solved for several levels for the probabilistic constraint on aggregate invasion risk, Φ_s. Because there are multiple control technologies, each with their own lumpy fixed costs, a constrained, mixed-integer nonlinear programming (MINLP) problem must be solved to determine cost-minimizing allocations of control efforts. Although there are many ways to solve such problems, we adopt a brute-force approach that is equivalent to determining an optimum for each possible combination of technology adoption choices across vessels and then comparing these local optima to find the global optimum. In our case, each of the 31 vessel classes has eight possible combinations of technology choices (to use, or not use, each of up to three technologies). Consequently, there are 8^{31} possible technology combinations for which the locally optimal effort levels must be found. Fortunately, many of these permutations are dominated and can be eliminated from consideration, as Horan and Lupi (forthcoming) have determined that (i) the high fixed costs of technologies implies that it is never optimal for a single vessel to adopt two technologies, and (ii) it is never optimal for a smaller vessel to adopt filtration while a larger vessel adopts ballast exchange, because this would only increase costs and reduce effectiveness.[11] So, such dominated permutations (the bulk of possibilities) are therefore ruled out of the comparisons. This enables us to solve for optimal effort levels and to compare results from a small subset of permutations.

Results

Simulation results for several values of Φ_s are reported in Table 3 for the least-cost or first-best outcome, risk-reduction subsidies, and various uniform technological subsidies. Because we lack specific policy reasons for setting differential risk levels, in what follows we set Φ_s equal to Φ $\forall s \in \hat{S}$, so we drop the subscript *s* from Φ in what follows. In addition to Table 3, costs are also presented in Figure 2 for different levels of risk Φ. Costs are expressed as an index, with the base case being industry-wide costs in the least-cost outcome when $\Phi = 0.05$ (the least stringent case). An index value of 175, for example, would indicate that costs are 75 percent larger than costs in the base case.

Consider the results for the least-cost or first-best scenario. This scenario is characterized by participation by all vessels, with a mix of technologies adopted across vessels. The specific cost-effective mix of technologies used by the vessels depends on the value of Φ. Ballast exchange is used more extensively in the least-cost solution when the allowable risk level (Φ) is larger, as evidenced in Table 3 by the proportion of total control costs in ballast exchange. When the constraint on the overall level of risk (Φ) is reduced (making the constraint more stringent), effective ballast transfers require so much effort that it becomes optimal for some vessels to incur the fixed costs of filtration to take advantage of filtration's low unit cost and higher effectiveness. Heating's high unit costs prevent it from being a cost-effective option for any vessel for any value of Φ. Because the least-cost approach would emerge from the use of first-best subsidies applied to risk reduction, we also present subsidy payments for the least-cost case in Table 3. Subsidy payments for this case are indicated to equal total costs. The actual payments depend critically on the choice of baseline, which along with the subsidy rate represents a lump-sum, non-distortionary component of the subsidy. Because first-best subsidy rates are vessel-specific, we have assumed that the baselines can also be chosen to

[10] After deriving fixed costs, they are annualized using a rate of 8 percent over a 15-year interval to obtain the results in Table 2.

[11] Of course, fixed costs also matter and in theory could affect this result, but random experiments support the assumption.

Table 2. Costs of Ballast Water Management Technologies

		Ballast Capacity (m^3)	
Control Technology	Type of Costs[a]	12,000	60,000
Ballast Exchange (with $x_B = 0.75$)	Variable costs	2.814	2.244
	Fixed costs	2.238	0.540
Filtration (with $x_F = 1$)	Variable costs	0.180	19.050
	Fixed costs	0.480	6.564
Heating (with $x_H = 1$)	Variable costs	2.684	3.355
	Fixed costs	0.432	0.540

[a]All costs are expressed in U.S. cents/m^3.

Source: Table adapted from Rigby and Taylor (2001).

be vessel-specific and set at a level that ensures that the subsidies just cover control costs. This represents the best-case scenario for subsidies since it is the smallest subsidy for which vessels will participate, and since it ensures no rent transfer.

Now consider the results for the subsidy on risk reduction. The relative performance of this subsidy scheme depends on the overall risk level, as can be seen in Table 3 and in Figure 2. When $\Phi = 0.05$, control costs are 28 percent larger under the subsidy than in the least-cost allocation, while there is only a 14 percent difference when $\Phi = 0.005$. As with the least-cost scenario, all vessels participate in the case of risk-reduction subsidies. Under both subsidy scenarios, most vessels adopt filtration when Φ is set at low levels, and they increase their effort levels in this technology as Φ is reduced. But they can increase their effort levels only so much before they hit an upper bound on the effectiveness of the technology. The result is that the inefficiencies of the risk-reduction subsidy are diminished as Φ becomes smaller because there are fewer technological/behavioral options as Φ is reduced. Even in the least-cost outcome, more and more vessels must operate with maximum effort when more stringent goals must be satisfied, leaving less room to exploit vessel-specific cost differences that could otherwise lead to increased savings. Consequently, the least-cost and risk-reduction allocations become more similar when the aggregate risk goal is lowered.

Where inefficiencies do arise under the risk-reduction subsidy, this subsidy results in higher costs for two reasons. First, the subsidy is based on the risk of an introduction by *any* species, and so vessels do not have incentives to differentially consider how their choices affect the likelihood of invasion by each individual species. The result is overcontrol of mysids and *Clupeonella caspia* relative to the least-cost outcome. The risk-reduction subsidy's second and perhaps more important source of inefficiency is that the incentive rate is uniformly applied to all vessels and does not take into account the differential marginal impacts that each vessel's risk has on the aggregate likelihood of an invasion. The result is that vessels with high marginal costs and small marginal risk impacts will over-invest in pollution control measures, while vessels with low marginal costs and large marginal risk impacts will under-invest, reducing the cost-effectiveness of the resulting allocation of controls (Baumol and Oates 1988). Specifically, more of the smaller vessels invest in filtration under the risk-reduction subsidy, significantly increasing the fixed costs incurred relative to the least-cost solution. The uniform subsidy does not affect the larger vessels' choice to adopt filtration, relative to the first-best case, but it does cause them to apply less effort to this technology than in the first-best case. Due to the large fixed costs of filtration and the small variable costs, the net effect is an inefficiently large allocation of costs to filtration. This explains why the risk-reduction subsidy results in a larger proportion of costs in filtration relative to the least-cost solution (see Table 3).

Full participation in the risk-reduction subsidy case implies that the rents created by the subsidies are sufficient for covering fixed costs. As described above, this result could occur if the subsidy rate was set inefficiently high in order to boost rents and encourage participation. But boosting rents in this manner is not required in the present case. The same subsidy rates would

Table 3. Simulation Results

Scenario	Annual costs[a]	Subsidy payments[a]	Participation rate	Proportion of total costs in: Ballast exchange	Proportion of total costs in: Filtration	Probability of invasion: ***Corophium spp.***	Probability of invasion: Mysids	Probability of invasion: ***Clupeonella caspia***
Base case (no biosecurity)	0.0	--	0.0	0.0	0.0	0.1	0.1	0.1
Case I: $\Phi_s \le 0.05\ \forall s$								
Least cost	100.0	100.0	100.0	0.36	0.64	0.05	0.05	0.02
Risk-reduction subsidy	128.0	223.3	100.0	0.33	0.67	0.05	0.04	0.01
Ballast-exchange subsidy	210.3	312.4	73.0	1.0	0.0	0.05	0.05	0.05
Filter subsidy	155.6	193.5	73.0	0.0	1.0	0.05	0.05	0.05
Heat subsidy	1062.1	1891.9	77.3	0.0	0.0	0.05	0.05	0.04
Case II: $\Phi_s \le 0.01\ \forall s$								
Least cost	202.5	202.5	100.0	0.18	0.82	0.01	0.01	0.004
Risk-reduction subsidy	220.5	346.0	100.0	0.09	0.91	0.01	0.008	0.003
Ballast-exchange subsidy	359.0	953.1	97.7	1.0	0.0	0.01	0.01	0.01
Filter subsidy	234.2	424.8	97.7	0.0	1.0	0.01	0.01	0.01
Heat subsidy	2085.4	10708.1	100.0	0.0	0.0	0.007	0.003	0.0005
Case III: $\Phi_s \le 0.005\ \forall s$								
Least cost	221.1	221.1	100.0	0.18	0.82	0.005	0.005	0.002
Risk-reduction subsidy	235.4	473.3	100.0	0.08	0.92	0.005	0.003	0.001
Ballast-exchange subsidy	387.6	1334.2	100.0	1.0	0.0	0.001	0.001	0.001
Filter subsidy	246.3	557.5	100.0	0.0	1.0	0.002	0.001	0.001
Heat subsidy	∞	∞	100.0	0.0	0.0	Tech. limit = 0.007	Tech. Limit = 0.003	Tech. Limit = 0.0005

[a]Costs and subsidies are expressed as a percentage of costs in the least-cost outcome under Case I.

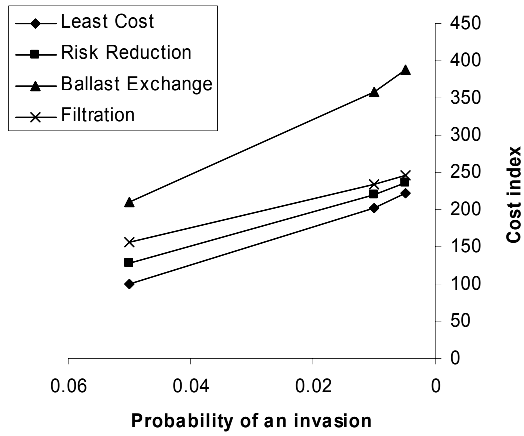

Figure 2. Control Costs by Incentive Program

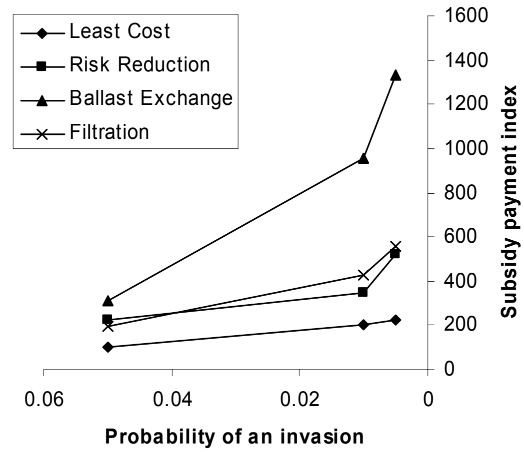

Figure 3. Subsidy Payments by Incentive Program

emerge even if all fixed costs were to vanish, that is, even if any potential participation barriers were removed. In other words, fixed costs are not a binding constraint on participation under the risk-reduction subsidy.

Figure 3 illustrates that, as Φ is reduced, there is an increasing divergence between subsidy payments in the risk-reduction and least-cost cases. Because least-cost subsidies equal control costs in the least-cost case and because we know control costs converge as Φ gets small, this result simply means that risk-reduction subsidy rents are larger when there is more control. This is a standard result in the economics literature on conventional pollution control (Baumol and Oates 1988).

Now consider the case of a uniform filtration subsidy, which is generally a more costly approach than either the least-cost or risk-reduction scenarios. When $\Phi = 0.05$, the filtration subsidy is 55.6 percent more costly than the least-cost approach, and it is 21.6 percent more costly than the risk-reduction subsidy. The reason is that the filtration subsidy is not as well targeted. First, it does not always encourage full participation. For larger values of Φ, the filtration subsidy does not provide sufficient rents to cover fixed costs for all vessels, and so only 73 percent of the vessels participate when $\Phi = 0.05$ (Table 3). This means that Φ is achieved through excess controls by the participating vessels. In addition to this allocative inefficiency, additional inefficiencies arise because the subsidy provides poorly targeted incentives among those vessels that do participate. These inefficiencies arise because (i) the subsidy base, filtration, is less correlated with the externality than the compliance measures of the other two approaches and therefore does not provide incentives for vessels to choose the least-cost options for reducing risk, and (ii) the uniformity of the subsidy does not encourage vessels to consider the marginal environmental impacts of their individual control efforts. In contrast, the risk-reduction subsidy is more efficient because it encourages vessels to choose among multiple technologies to reduce risk in a least-cost fashion. This result is analogous to those results of the pollution control literature that find that performance-based (emissions-based) instruments are more cost-effective than technology-based instruments (Russell and Powell 2000). The inefficiencies associated with participation and correlation with risk are diminished as Φ is reduced because full participation is encouraged to achieve the most stringent risk goals and, as discussed above, because filtration becomes the primary technology for achieving small values of Φ in the least-cost solution. Indeed, Figure 2 illustrates a convergence in costs for the three approaches described so far.

Figure 3 illustrates that aggregate filtration subsidy payments are smaller than risk-reduction subsidy payments when $\Phi = 0.05$. This is because there are fewer vessels to pay in this case under the filtration scenario, although the payment per vessel is larger than in the risk-reduction scenario. As participation is increased under the filtration scenario as Φ gets smaller, aggregate subsidy payments rise, and become larger than those arising under the risk-reduction subsidy.

Consider the last two subsidy scenarios: those based on ballast exchange and heat. When $\Phi = 0.05$, the ballast exchange subsidy results in costs that are 128 percent larger than in the least-cost outcome and 64 percent larger than those arising under risk-reduction subsidies. As Φ is reduced, the absolute cost differences only get larger (although the percentage differences get smaller). As illustrated in Figure 3, subsidy payments under the ballast exchange system are 40 percent larger than risk-reduction payments when $\Phi = 0.05$, and this difference grows to 182 percent for $\Phi = 0.005$. Finally, the uniform heat subsidy is vastly more expensive than all of the other approaches, and is therefore not depicted in Figure 2 or Figure 3.

Conclusion

Alien invasive species transported in ballast water are a form of trade-related biological pollution and, as such, their prevention can be treated as a type of transboundary pollution control problem. Unlike many conventional pollutants, IAS emissions cannot be measured or controlled with certainty, and not every vessel will actually emit a species. These characteristics raise the issue of what compliance measures pollution policy instruments should be based on. As we have described, options include particular pollution prevention technologies as well as a performance proxy consisting of estimates of a vessel's contribution to the risk of an invasion.

To evaluate the potential cost-effectiveness of various second-best subsidies for IAS control, a model of Great Lakes shipping was developed. The simulation results suggest that subsidies based on the performance proxy have the potential to outperform uniform technology subsidies, though the efficiency gains from using a performance-based approach depend on the target level of aggregate invasion risk. At low or intermediate target levels for aggregate invasion risks, cost savings from a performance-based approach do emerge because vessels have the flexibility to choose from a variety of technologies. However, at more stringent levels of aggregate risk reduction, the responses of vessels are limited and potential cost savings from a performance-based approach are smallest. For the lower risk levels, the least-cost solution involves most vessels adopting filtration. When it is efficient for most vessels to use the same control technology, the relative gains from using a performance-based approach instead of a technology-based approach will be small. The findings suggest that, for lower allowable risk levels, a uniform technology subsidy can achieve the desired risk reductions at relatively low costs, provided the right technology is selected for subsidization. Correctly selecting the low-cost technology is key to this finding, especially in light of the high fixed costs associated with some of the technologies.

The findings presented here are based on the limited data currently available for key model parameters. A key gap in the available research, and a subject on which more research is clearly desirable, is the efficacy and costs (fixed and variable) of all possible ballast treatment technologies. As we have shown, such data are vital for establishing the relative risks and cost-effectiveness of policy instruments. We expect that improved information in this area will be forthcoming, in part due to the ballast technology demonstration projects currently underway. Since vessel-specific information on invasion risks is essential to the definition of the performance proxy we propose, the simulations would also benefit from more precise estimates of the invasion risks for existing or new potential invaders. Clearly, better risk information is a precursor to policy implementation of the more cost-effective performance proxy instruments. Given the scientific focus on identifying potential invaders and estimating invasion risks, we expect that the availability of such risk information will improve.

Because the focus of this research was on comparing policy instruments, the empirical model has addressed only the cost-effectiveness of meeting alternative standards for industry-wide invasion risks. No assessment has been made regarding the social desirability of alternative levels for the aggregate risk standards. Determining economically desirable invasion-risk levels requires cost-effectiveness information and information on the potential damage costs associated with invasions. Although more ecological and economic research is required to make the connection between invasions and damages, such research is critical to determine whether it is worth the cost to undertake any level of ballast control. The reason for this is that any non-zero level for the standard on aggregate invasion risk will not permanently prevent an invasion. For example, with a fairly small annual invasion risk of one percent, the risk of invasion over the next decade is 10 percent and becomes 18 percent risk over the next two decades. This suggests that IAS issues are properly framed as intertemporal risk management problems. But such analyses require more information on damages and on potential future investment and innovation opportunities than is currently available. Invasive alien species problems do not simply go away while this information is being collected, nor does the policy process patiently sit and wait for these analyses to be performed. Consequently, our research can best be viewed as a first step toward understanding risk management and policy design issues associated with IAS problems, and can hopefully inform future research and policy debates.

References

Aseinova, A.A. 2003. "Clupeonella cultriventris caspia." Caspian Sea Biodiversity Database, Caspian Environment Programme, Baku, Azerbaijan. Available at http://www.caspianenvironment.org/biodb/eng/main.htm (accessed March 31, 2003).

Baumol, W.J., and W.E. Oates. 1988. *The Theory of Environmental Policy*. New York: Cambridge University Press.

Baychorov, V.M. 1980. "Reproduction of the Mysid *paramysis lacustris* (crustacea) in the Northern and Southern Zones of Its Introduction." *Hydrobiological Journal* 15(6): 29–34.

Beavis, B., and M. Walker. 1983. "Achieving Environmental Standards with Stochastic Discharges." *Journal of Environmental Economics and Management* 10(2): 103–111.

Coscarelli, M., and E. Bankard. 1999. *Aquatic Nuisance Species Handbook for Government Officials*. Office of the Great Lakes, Michigan Department of Environmental Quality, Lansing, MI.

Farley, R.B. 1996. "Analysis of Overseas Vessel Transits into the Great Lakes through Commercial Shipping Ballast Water and Assessment of Control Options, Phase I and Phase II." Unpublished paper, Department of Naval Architecture and Marine Engineering, University of Michigan, Ann Arbor, MI (October).

Great Lakes Commission. 2000. *Ballast Management for the Prevention and Control of Aquatic Nuisance Species*. Great Lakes Commission, Ann Arbor, MI. Great Lakes Commission Resolutions available online at http://www.glc.org/about/resolutions/ballast.html (accessed May 25, 2000).

Hanley, M., J.F. Shogren, and B. White. 1997. *Environmental Economics in Theory and Practice*. New York: Oxford University Press.

Helfand, G.E., and B.W. House. 1995. "Regulating Nonpoint Source Pollution under Heterogeneous Conditions." *American Journal of Agricultural Economics* 77(4): 1024–1032.

Holmes, B. 1998. "Day of the Sparrow." *New Scientist* (June 27): 32–35.

Horan, R.D. 2001. "Differences in Social and Public Risk Perceptions and Conflicting Impacts on Point/Nonpoint Trading Ratios." *American Journal of Agricultural Economics* 83(4): 934–941.

Horan, R.D., and F. Lupi. 2003. "The Economics of Policy Options for Controlling the Introduction and Spread of Aquatic Nuisance Species in the Great Lakes. Interim report, Project No. R/EP-9, submitted to Michigan Sea Grant, April 11.

____. Forthcoming. "Tradeable Risk Permits to Prevent Future Introductions of Invasive Alien Species in the Great Lakes." *Ecological Economics*.

Horan, R.D., C. Perrings, F. Lupi, and E. Bulte. 2002. "Biological Pollution Prevention Strategies Under Ignorance: The Case of Invasive Species." *American Journal of Agricultural Economics* 84(5): 1303–1310.

Just, R., and J. Antle. 1990. "Interactions between Agricultural and Environmental Policies." *American Economic Review* 80(2): 197–202.

Karaminas, L., H.H. Ocakli, K.C. Mazdon, and P.C. Westlake. 2000. "Ballast Water Management." LR Technical Association Paper No. 1, Session 2000–2001, Lloyd's Register of Shipping, London.

Kolar, C.S., and D.M. Lodge. 2002. "Ecological Predictions and Risk Assessments for Alien Species." *Science* 298(5596): 1233–1236.

Lichtenberg, E., and D. Zilberman. 1988. "Efficient Regulation of Environmental Health Risks." *Quarterly Journal of Economics* 103(1): 167–168.

Lichtenberg, E., D. Zilberman, and K. Bogen. 1989. "Regulating Environmental Health Risks Under Uncertainty: Groundwater Contamination in California." *Journal of Environmental Economics and Management* 17(1): 22–34.

Lupi, F., J. Hoehn, and G. Christie. 2003. "Using an Economic Model of Recreational Fishing to Evaluate Benefits of Sea Lamprey (*Petromyzon marinus*) Control." *Journal of Great Lakes Research* 29(S1): 742–754.

MDEQ [see Michigan Department of Environmental Quality].

Michigan Department of Environmental Quality. 1996. *Nonindigenous Aquatic Nuisance Species State Management Plan: A Strategy to Confront Their Spread in Michigan*. Available online at http://www.michigan.gov/deq/0,1607,7-135-3313_3677_8314-16514--,00.html (accessed October 25, 2001).

Mills, E.L., J.H. Leach, J.T. Carlton, and C.L. Secor. 1993. "Exotic Species in the Great Lakes: A History of Biotic

Crises and Anthropogenic Introductions." *Journal of Great Lakes Research* 19(1): 1–54.

National Research Council Committee on Ships' Ballast Operations. 1996. *Stemming the Tide: Controlling Introductions of Nonindigenous Species by Ships' Ballast Water*. Washington, D.C.: National Academies Press.

NRC [see National Research Council Committee on Ships' Ballast Operations].

Perakis, A.N., and Z. Yang. 2001. "An Evaluation of the Economic Impact of Proposed Non-Indigenous Species Legislation for the St. Lawrence Seaway Using Multi-Attribute Decision Theory." Interim Report to the Michigan Sea Grant Program (May).

Perrings, C., M. Williamson, E.B. Barbier, D. Delfino, S. Dalmazzone, J. Shogren, P. Simmons, and A. Watkinson. 2002. "Biological Invasion Risks and the Public Good: An Economic Perspective." *Conservation Ecology* 6(1) (available online at http://www.consecol.org/vol6/iss1/art1).

Perrings, C., M. Williamson, and S. Dalmazzone (eds.). 2000. *The Economics of Biological Invasions*. Cheltenham, UK: Edward Elgar.

Pezzack, D.S., and S. Corey. 1979. "The Life History and Distribution of Neomysis Americana (Smith) (Crustacea, Mysidacea) in Passamaquoddy Bay." *Canadian Journal of Zoology* 57(4): 785–793.

Pollutech [see Pollutech Environmental Limited].

Pollutech Environmental Limited. 1996. "A Review and Evaluation of Ballast Water Management and Treatment Options to Reduce the Potential for the Introduction of Non-Native Species to the Great Lakes." Report to the Canadian Coast Guard (Sarnia, ON).

Rajagopal, S., G. van der Velde, B.G.P. Paffen, F.W.B. van den Brink, and A. Bij de Vaate. 1999. "Life History and Reproductive Biology of the Invasive Amphipod *Corophium Curvispinum* (Crustacea: Amphipoda) in the Lower Rhine." *Archiv fur Hydrobiologie* 144(3): 305–325.

Reeves, E. 1999. "Exotic Policy: An IJC White Paper on Policies for the Prevention of the Invasion of the Great Lakes by Exotic Organisms." Great Lakes Water Quality Forum, Milwaukee, WI.

Reid, D.F., and M.I. Orlova. 2002. "Geological and Evolutionary Underpinnings for the Success of Ponto-Caspian Species Invasions in the Baltic Sea and North American Great Lakes." *Canadian Journal of Fisheries and Aquatic Sciences* 59(7): 1144–1158.

Ricciardi, A., and J.B. Rasmussen. 1998. "Predicting the Identity and Impact of Future Biological Invaders: A Priority for Aquatic Resource Management." *Canadian Journal of Fisheries and Aquatic Sciences* 55(7): 1759–1765.

Rigby, G., and A.H. Taylor. 2001. "Ballast Water Treatment to Minimize the Risks of Introducing Nonindigenous Marine Organisms into Australian Ports." Ballast Water Report Series No. 13, Australian Government Department of Agriculture Fisheries and Forestry, Canberra.

Russell, C.S., and P.T. Powell. 2000. "Practical Considerations and Comparison of Instruments." In J. van den Berg, ed., *Handbook of Environmental and Resource Economics*. Cheltenham, UK: Edward Elgar.

Shackle, G.L.S. 1969. *Decision, Order and Time in Human Affairs*. Cambridge: Cambridge University Press.

Shortle, J.S., and D.G. Abler. 1997. "Nonpoint Pollution." In H. Folmer and T. Tietenberg, eds., *The International Yearbook of Environmental and Resource Economics 1997/98*. Cheltenham, UK: Edward Elgar.

Shortle, J.S., and J.W. Dunn. 1986. "The Relative Efficiency of Agricultural Source Water Pollution Control Policies." *American Journal of Agricultural Economics* 68(3): 668–677.

U.S. Army Corps of Engineers. 2002. *NDC Publications and U.S. Waterway Data CD* (Vol. 8). Institute for Water Resources, Navigation Data Center, U.S. Army Corps of Engineers, Alexandria, VA.

U.S. Environmental Protection Agency. 2001. "Aquatic Nuisance Species in Ballast Water Discharges: Issues and Options." Draft report for public comment, Office of Wetlands, Oceans and Watersheds, Office of Wastewater Management, U.S. Environmental Protection Agency, Washington, D.C.