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The economics of biological invasions

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Abstract

Biological invasions are an economic problem. Invasions are typically the intended or unintended consequence of economic activity. They impose real costs on society, and the risk of invasion depends on human behaviour. Effective control of invasions depends on using the right economic instruments and developing the right institutions. The problem has two special features. The first is that the risks of invasions may be very low, but the potential costs are high. Since they are not reflected in market prices, they are typically ignored. The second is that the control of potentially invasive species is a public good of the 'weakest link' variety. Both features indicate a precautionary approach. To deal with the first, I recommend the use of environmental assurance bonds to cover society against the risks of invasive species whilst providing importers with an incentive to research the consequences of their actions. To deal with the second I recommend the development of an institution similar to the Atlanta Centre for Disease Control to provide the information and technical advice required if governments are to act, and a central organisation (involving UNEP, UNDP and the World Bank) to strengthen eradication, control of mitigation campaigns in those countries least able to deal with invasive species.

Biological invasions: the economic problem

It is curious that a phenomenon that is attracting so much attention from ecologists (Enserink, 1999) and environmental organisations (UNEP, 1999) should have received so little attention from economists. While there are many studies of the costs and benefits of programmes for the control of particular weeds, pests and pathogens, there are no generic studies of the economics of biological invasions. Research under the GISP programme (Perrings *et al.*, 2000) has identified some of the key points at issue, but the agenda is still wide open. This paper proposes preliminary answers to two questions. The first is how does economics alter our perspective on biological invasions. The second is what does economics offer by way of solutions to the problems posed by biological invasions.

There are only two estimates of the aggregate cost of invaders at the national level, both relating to the USA. These are US OTA (1993) and Pimentel *et al.* (1999). The difference between them reflects the growth in concern over the problem. Seven years ago the US OTA estimated the damage costs from 79 particularly harmful species over the preceding 85 years to be $96,994 million. Last year Pimentel *et al.* estimated the annual damage costs from all species to be $122,639 million. While we are probably not much wiser about the true net costs of invasions, we do know that they are important enough for policy makers to take the problem seriously. We also have a better idea of where the costs lie.

It is clear is that invasive species have impacts beyond pests and pathogens in agriculture, forestry or fisheries. They are, for example, thought to be one of the main proximate causes of biodiversity loss worldwide (Glowka, 1994; Wilcove *et al.*, 1998; Czech and Krausman, 1997). This has threatened key ecological functions in many systems, and has had far reaching implications for many other activities (Holling, 1992; Heywood, 1995). We also know that most ecosystem types — terrestrial, fresh-water and marine, animal plant and microbe—have been impacted to a greater or less extent by invasions (Parker *at al.*, 1999; Williamson, 1998, 2000).

To date, relatively little work has been done on the economic implications of these more general impacts. While there are reasonable estimates of the financial costs of herbicide control for plant species in a small number of countries, there are few estimates of damage or control costs for most invasive species (Williamson, 1998). Examples include estimates of the potential control costs of the screw worm fly, *Chrysomya bezziana*, in Australia (Anaman *et al.*, 1994); the benefits from clearing alien species from Fynbos ecosystems in South Africa (Higgins *et al.*, 1997); the impact of Knapweed and Leafy Spurge on the economy of several US states (Bangsund *et al.*, 1999); the damages to North American and European industrial plants from the zebra mussel and other invaders (Khalanski, 1997); and the impact of the green crab, *Carcinus maenas*, on the North Pacific Ocean fisheries (Cohen *et al.*, 1995). But these studies do no more than scratch the surface.

At one level, biological invasions are very obviously an economic problem. They are an example of anthropogenic environmental change. Invasions are typically the intended or unintended consequence of economic activity. They impose real costs on society, and the risk of invasion depends on human behaviour. The regulation or control of invasions should clearly involve economic analysis and economic instruments whether we are concerned with the exploitation of ecosystems — by agriculture, forestry and fisheries — or with their conservation.

The reasons are quite straightforward. We need to understand the incentive effects of existing institutional, regulatory and market conditions if we are either to make predictions about invasive species or to develop policies for their control. We need to understand the cost effectiveness or efficiency of publicly funded eradication, control or mitigation options if we are to avoid waste. But we also need to understand the limitations of economic instruments for dealing with the class of problems to which biological invasions belong. More than most other environmental problems, biological invasions expose the limitations of economics. The nature of the risks of invasions and the impact of invasions on ecosystem functioning and economic production limit the effectiveness of many tools of economic analysis and instruments of economic control.

Having said that biological invasions are an economic problem, let me add immediately that this does not mean that the problem can be analysed in terms of economics alone — any more than it can be analysed in terms of ecology alone. To understand the implications of invasions for human welfare we need to specify an appropriate 'production function' describing the relationship between inputs and outputs (Perrings, Folke and Mäler, 1992). Individual species in an ecosystem are conceptualised as inputs to the production of useful goods and services via a set of ecological functions and processes. Their value, like the value of other inputs, derives from the value of the goods and services they support. This implies an analysis that is as much ecological as economic.

Invasions are an outcome of a very complex set of processes. The causes of invasions include the use of exotic species in a wide range of economic activities, the conversion and fragmentation of habitat, the liberalisation/deregulation of markets, the expansion in the trade of goods and services and the increasing mobility of people (Dalmazzone, 2000). Many of these processes typically confer significant local benefits. The potential costs of invasions should be assessed against such benefits. The difficulty for economics stems from the fact that invasions typically involve low probability events potentially leading to fundamental change in ecosystem functions, and that this may impose very high costs on local resource users. Technically, this implies that the relevant production sets are non-convex and that the expected utility approach to decision-making under uncertainty — the conventional approach — fails to predict the decisions that people actually make in such circumstances.

One of the major 'risks' associated with invasions is that by altering biodiversity, ecosystems may be transformed into new configurations with unknown consequences for human welfare. Biodiversity helps to maintain ecological services in the face of variable environmental conditions (Holling *et al*, 1995). In this sense it has indirect value: the value of ecological resilience. Invasions have the potential to cause loss of resilience. Moreover, like the spread of infectious disease, invasions also require control that is effective everywhere. Because the control of invasions is in the nature of a public good, it will in any event be under-provided by the market. Its similarity to the spread of infectious disease means that the protection provided to all will only be as good as the protection provided by the least effective member of the community. The paper discusses the implications of both features for the economics of eradication, control or mitigation, and suggests mechanisms for dealing with them.

Vulnerability to invasion: economic aspects

Ecosystems vary in their susceptibility to invasion. Deserts, semi-deserts, tropical dry forests and woodlands, temperate systems and pelagic marine systems appear to be least susceptible, while mixed island systems, lake, river and near-shore marine systems appear to be most susceptible (Heywood, 1995). Similarly, systems with low diversity — especially if they are without existing predators or competitors — appear to be more susceptible than systems with high diversity (Rejmánek, 1989). But susceptibility also depends on human behaviour, land use, demographic, market and institutional structures, the regulatory framework and the control strategies adopted. Habitat fragmentation, habitat conversion and agricultural disturbance are all argued to have increased the susceptibility to invasion (Williamson, 1996, 1999).

Indeed, the key to understanding the problem in any given country involves understanding the way that human behaviour and invasive species interact. Human behaviour reflects the incentives, institutions and regulatory framework – usually quarantine and the prohibition of blacklisted species – within which people make decisions (Perrings *et al.*, 2000). It makes it difficult to identify the probabilities attaching to different outcomes on the basis of the characteristics of species or their habitat alone. The spread of potentially invasive species depends on the use that people make of invasive species, on their predators and competitors, on demographic patterns, on transport networks and the like.

The structure and trade dependence of an economy both turn out to be important in determining its vulnerability. In a preliminary study, Dalmazzone (2000) considers the relation between the establishment of alien plant species in 29 countries, trade flows and their composition, arable, pastoral and forested land, and other socio-economic variables. She finds that economic activities can increase the inherent susceptibility of ecosystems. Specifically, the variables affecting the recipient environment (land tenure, level of GDP, population density) are responsible for explaining a high proportion of the variation in the share of alien species in different countries. All ecological communities are susceptible to invasion to some degree. Economic activities can increase the susceptibility of most.

Trade has a weaker though still significant impact on the share of alien species hosted by any given country. Preliminary results also add some insight into the problem of biological invasions on islands. Island ecosystems are generally considered highly susceptible to invasions because of a particularly vulnerable native biodiversity. But island states are also typically, on average, small open economies, often geared to the production of primary products. The average percentage of merchandise imports as a share of the GDP, in the sample considered, is about 43% for island countries, against an average 32% for the whole sample, and 26.8% for continental countries. Island states are not only more susceptible ecologically, they are much more open in terms of the movement of goods and services across ecological boundaries.

Some interactions between human behaviour and biological invasions are quite intuitive. For example, the probability of establishment of intentionally introduced species is higher than that of unintentionally introduced species. One reason is that intentionally introduced species have been selected for their ability to survive in the environment where they are introduced (Smith *et al*., 1999). Another is the link between intentional and repeated introduction. Exotic species that are marketed over a period of time have a greater probability of establishment than those that are marketed once (Enserink, 1999). The probability of both establishment and spread also depends on the way in which the environment is altered by human behaviour. The introduction of specific disease or pest resistant crops, for example, selects in favour of other pests and predators in a way that is well — understood (Heywood, 1995).

The main implication of this is that the risks of invasions depend on human responses to the threat of invasions. Shogren (2000) makes the point that decision-makers protect themselves against the risks of invasive species in two related ways: by mitigation and by adaptation. Mitigation includes the conventional categories of eradication and control, and has the effect of reducing the likelihood that a species will establish or spread. Adaptation, on the other hand, implies some change in behaviour to reduce the impact of the establishment and spread of a species. That is, it works on the value of the effect, rather than the likelihood of the effect. Nonetheless, mitigation and adaptation jointly determine the risks and the costs of invasions. He concludes that if mitigation and adaptation are linked in this way, then economic and environmental systems are jointly determined. Risk assessment and risk management cannot be handled separately (Shogren and Crocker, 1991).

A good example lies in the interaction between human behaviour and the spread of pathogens. Delfino and Simmons (2000) model the spread of tuberculosis in developing economies. They show that while the probability of infection influences human decisions in a way that is quite straightforward, the interactions between the virulence of a disease, infected and susceptible populations, the pattern of settlement and the level of development, may be very complex. The relation between the level of economic development, the epidemiology of an invasive pathogen, and human settlement and migration, may lead to any one of a number of possible states depending upon the parameter values. They also show that while the spread of disease is affected by the private costs and benefits of the options facing people, and so may be influenced by economic incentives, people typically ignore the impact of their decisions on the infection risks to others.

The importance of this last point is that there is a public good element in the control of the risks of disease, just as there is in the control of invasive species generally. If left to the market, disease and pest control will both be 'under-supplied'. In fact the market prices of potentially invasive species seldom reflect the costs they may impose on society. That is, invasions are typically external to the market. Nor is this helped by the fact that many markets have been prevented from operating efficiently by agricultural policies and institutions. Tax, price and incomes policies have all increased the susceptibility of agroecosystems to invasion. For example, subsidies designed to promote the export of cash crops has reduced plant genetic diversity and encouraged the use of farm inputs — especially pesticide regimes — in a way that has laid agroecosystems open to invasion (Perrings, 2000). Property rights regimes have discouraged individuals from taking action to control invading species. Land in common property, for example, requires collective action, and open access land militates against any action at all (Hanna, Folke and Mäler, 1997).

More important still is the nature of the public good. The control of invasive species in agriculture and forestry, like the control of communicable diseases, depends on the least effective member of the community. Put another way, the public good involved in the control of infectious diseases and many other invasive species is of the 'weakest link' variety (Sandler, 1997). If control of an invasive plant involves eradication campaigns in all nations sharing access to the affected system, that control will only be as good as the campaign run by the least effective nation. The risks to all depend on the capacity of the weakest. This makes the fact that capacity is falling in many countries a matter for real concern. Falling barter and income terms of trade for many of the poorest countries have limited the resources available to control the effects of invasions, thus raising the risks to all (Perrings *et al*., 2000).

Assessing the risks of invasions in a ecological-economic system

If vulnerability to biological invasions depends as much on the economic system as it does on the ecological system, then the risks of invasion should be calculated accordingly. However, the inclusion of social as well as natural effects compounds the problems in assessing the risks of invasions. Not only does human behaviour determine the risks of establishment and spread, the social significance of those risks depends on the value placed by society on the consequences. Very few established alien species impose sufficient costs to be defined as serious pests, but those that do can be very serious indeed. The decision problem reflects the fact, as has already been noted, that biological invasions fall into the category of low-probability events but with a high potential cost.

The probability that any one introduced species will establish and become a pest or pathogen is low. Williamson's (1996) 'tens rule' – that 1 in 10 introduced species may be found in native habitats, but that only 1 in 10 of those establishes, and only 1 in 10 of those becomes a pest — indicates that the probability is in the order of 0.01%.

At the same time, the control and/or damage costs of species that do become significant pests or pathogens may be extremely high. Pimentel *et al.*'s (1999) estimate of costs of $122 639 million to the USA per year is high enough. But consider that in relation to just one invasive species that is not typically included in the analysis, HIV, and it does not look disproportionate. The cost of the latter in terms of the social disruption it has caused, output lost from morbidity and mortality amongst the young, research and development into treatment options and so on is not known, but is certainly a significant percentage of global NNP. The cost of most plant pests and pathogens is dwarfed by comparison. Nevertheless, these costs are still substantial. The annual control costs for field bindweed in one state in the US, for example, is estimated to be over $40 million (FICMNEW, 1998). Deliberate introductions that have yielded some very substantial direct benefits have also had incidental costs that may be very high. The introduction of the Nile Perch into Lake Victoria, for example, is thought to have resulted in the elimination of some 200 haplochromine cychlid species (Kasulo, 2000).

A standard economic analysis of control in cases such as these depends on estimation of the expected values for the costs and benefits of control options.[1] This makes it possible to calculate an expected net present value and the options can then be compared according to that (or a number of related) yardsticks. However, the empirical evidence is that decision-makers do not behave in this way. Estimates of the probability of an undesirable outcome and of the costs of that outcome both tend to be biased.

Empirical research has shown that individual decisions in the face of risk are usually based on a reference point — generally the status quo — and that outcomes are assessed relative to that (Kahneman and Tversky, 1979). The weight given to a loss is often greater than the weight given to a gain of the same amount. This implies that utility functions are steeper for losses than for gains. People systematically choose the less risky outcome in a direct choice (reveal a higher willingness to pay for it), but place a higher value on the more risky outcome (reveal a higher willingness to accept compensation for it) if they were to sell it.

An extreme form of this, relevant to invasive pests or pathogens, is the 'dread' effect. This increases the subjective risk assessments of certain types of health risk (McDaniels, Kamlet and Fischer, 1992). It also affects people's valuation of those risks. People are willing to pay much more for risk-reduction measures in infrequent high dread situations, than in frequent low dread situations (Loomis and du Vair, 1993). Moreover, individual attitudes to risk are highly sensitive to income. It has long been known that people tend to be risk averse at low levels of income and risk avid or risk loving at high levels of income (Friedman and Savage, 1948).

[1] This is based on the expected utility hypothesis. It holds that people evaluate a risky prospect in terms of the mathematical expectation of the value or utility to them of the prospect

The expected value approach has less and less predictive power as the probabilities of outcomes tend to zero. People facing a 'very unlikely' event tend either to overestimate the probability or to identify it with zero. For very low probabilities the weighting function is often not defined. In the liability insurance markets for low probability high cost risks, for example, insurers typically demand a risk premium that exceeds the expected losses whilst the insured are willing to pay less than predicted by expected utility calculations (Katzman, 1988).

Historically, many evaluations of control options have calculated the ex post benefit-cost ratios for either successful invaders or effective controls. This is tantamount to calculating the ex post value of a winning lottery ticket. It tells nothing about the efficiency of the original decision to buy the ticket. It cannot guide ex ante decisions about when to control and when not to control. Nor can it guide the choice between control options. But even if the calculation were to be based on an ex ante analysis of costs and benefits, then as in the case of lotteries, decision-makers would be found to behave in a way that is simply not rational according to the expected utility approach.

Consider, for example, the method for evaluating screening procedures for potentially invasive species discussed by Smith, Lonsdale and Fortune (1999). They define the value of the costs of control (through screening) to be the product of three factors: (a) the accuracy of the control (the screening process), (b) the probability that a species will be invasive, and (c) the cost of screening error. The accuracy of a screening process is the proportion of species that have, ex post, been found to be appropriately dealt with in the screening process: namely the proportion of invaders assessed that are rejected by the process, and the proportion of non-invaders assessed that are accepted by the process. From these is derived a likelihood ratio for the screening process. The effectiveness of a screening system is then given by the product of the likelihood ratio, the probability that an introduced species will become a pest, and the cost if it does.

The first of these depends on the probability that the introduced species escapes to become a casual species, the probability that a casual species will naturalise, and the probability that the naturalised species will become a pest. In general this is very low — as the tens rule suggests — although it has been noted to be as high as 17% for some weedy species from some bioclimatic zones (Lonsdale, 1994). The probability that an introduced species will become a pest may be expressed as the ratio of the probability that an accepted invasive species will impose economic costs (will be a pest), and the probability that an accepted non-invasive species will confer economic benefits (will not be a pest). Similarly, the expected cost of pests, may be expressed as the ratio of the expected costs of accepting an invasive species that turns out to be a pest and the expected costs of rejecting a non-invasive species that turns out not to be a pest.

In terms of the notation in Smith *et al.* (1999), the test for the screening process to be acceptable is thus:

$$\frac{\frac{I_r}{I_t} \cdot R_i \cdot K_i}{\frac{N_r}{N_t} \cdot R_n \cdot K_n} \geq 1$$

The numerator is the product of the proportion of potential invasives rejected by the control, I_r/I_t, the probability of a potential invasive doing damage, R_i, and the expected cost if it does, K_i. The denominator is the product of the proportion of potential non-invasives rejected by the control, N_r/N_t, the probability of a potentially non-invasive species yielding benefits R_n,[2], and the expected net benefits of non-invasives, K_n.

In expected present value terms the test requires that the expected present value of the benefits of the screening process (the net costs avoided by the screening programme) are no less than the expected present value of the costs of control (the foregone benefits of the screening programme). In other words, under the expected utility hypothesis this ratio should approximately equal the ratio of expected discounted benefits and costs of the screening programme. In the case of low-probability high-cost events such as invasions, however, the ratio is likely to be very different from the expected discounted benefits and costs of control. That is:

$$\frac{\frac{I_r}{I_t}\cdot R_i \cdot K_i}{\frac{N_r}{N_t}\cdot R_n \cdot K_n} \neq E\sum_{t=0}^{T}\rho^t\left(\frac{B_{it}-C_{it}}{B_{nt}-C_{nt}}\right)$$

where $B_{it} - C_{it}$ are the net benefits of the exclusion of potentially invasive species at time t, $B_{nt} - C_{nt}$ are the net costs of excluding non-invasive species, and r is a discount factor. The 'dread' effect is likely to bias upwards private estimates of the costs if a potentially invasive species becomes a pest, while the low probability of outcomes is likely to induce a distortion in the perception of risk.

Biological invasions and economic control

A related problem for the economics of invasive species stems from the effects of invasions on ecosystem dynamics. Invasive species are typically defined to be those whose impacts involve the transformation of a system. Examples already referred to include pines in the South African fynbos, Brazilian pepper and melaleuca in the Florida everglades, cheatgrass and leafy spurge in the western USA, rabbits in Australia, the Nile perch and water hyacinth in the African lakes. Naturalised species that are present in an ecosystem, but are innocuous, tend not to be defined as invasive. The trigger to the recognition of species as invasives is that at some point they bring about a rapid and persistent change in the state of the system.

Most ecological systems are characterised by multiple equilibria. Much is known in particular cases about the factors that cause systems to flip from one equilibrium state to another. Well known examples include the role of the spruce budworm in boreal forests (Jones, 1975; Ludwig, Jones and Holling, 1978), and of herbivores in semi-arid savannas (Walker and Noy-Meir, 1982; Walker, 1988; Westoby *et al.*, 1989). In both cases there is a marked hysteresis effect. The initial change in states tends to be rapid, but the path back to the initial state tends to be very slow. In the spruce budworm case, for example, equilibrium budworm numbers are initially low but at a certain point increase very rapidly with dramatic effects on forest dynamics. The return to lower budworm densities is much slower (Ludwig, Walker and Holling, 1997).

If a system flips from one state to another, it may be said to have lost resilience. Invasive species accordingly tend to be defined as such only if they induce a loss of resilience in the ecosystem concerned. Loss of resilience means both an increase in the time taken to return to equilibrium following some shock (Pimm, 1984), and a narrowing of the range of environmental conditions over which the system can maintain the flow of ecosystem services (Holling, 1973). If changes of state that affect the productivity of a system are either irreversible or only slowly reversible, they impose costs or confer benefits on both present and future users of that system. Resilience, sensu Holling, is a measure of the size of the stability domain corresponding to an attractor or equilibrium, or a measure of the ability of a system to maintain its 'integrity' or 'health' in the face of stresses or shocks (Costanza, Norton and Haskell, 1993). By changing both the function and species diversity of ecosystems, biological invasions change their value to people. The link between resilience, diversity, risk and productivity is still disputed in ecology, but experimental research on grasslands has now shown that long-term ecosystem productivity falls significantly with a reduction in plant biodiversity (Tilman and Downing, 1994; Tilman *et al.*, 1996).

Agroecosystems, including plantation forests, turn out to be especially sensitive to invasions and the consequent loss of native species precisely because they are already simplified by the exclusion of competitor or predator species (Conway, 1993). The simplification of agroecosystems typically leads to a reduction in their resilience. The costs of a loss of resilience in such systems is relatively easy to calculate. It includes, for example, the cost of the herbicides, pesticides, fertilisers, irrigation and other inputs needed to maintain output in the simplified system. In extreme cases it also includes the cost of relief where output fails, relocation where soils or water resources have been irreversibly damaged, rehabilitation where damage is reversible and insurance against crop damage by pest or disease (Brock *et al.*, 2000).

Two things are important here. I have already remarked that if a system can flip from one state to another if sufficiently perturbed it implies the non-convexity of the 'production sets'. This turns out to have far-reaching consequences for the way we think about the economics and management of natural resources. It may imply the non-controllability or non-observability of the system. Decision-makers may not be able to 'see' the system through, for example, market prices, and may not be able to control its dynamics using the instruments available. A corollary of this is that the effects of introducing new species may be fundamentally uncertain. It may also mean either hysteresis or irreversibility in the state dynamics. A system that flips from one state to another at some set of prices may require a very different set of prices to return it to the original state (Mäler, Xepapadeas and de Zeeuw, 1999).

Moreover, the closer the system is brought to the boundaries of the stability domain, the greater the risk that the introduction of an alien species may result in irreversible or persistent change as the system flips from a higher

[2] It is usually assumed that the probability of a potentially non-invasive species yielding benefits is one.

productivity state to a lower productivity state. In most cases, however, the observable level of environmental quality or productivity is a poor indicator of where the environment is relative to the thresholds between states. The fact that bifurcation points may not be seen until they are reached is a problem for both the economics and management of species introductions.

Institutions and incentives for the control of biological invasions

Where does this leave the economic control of biological invasions? Note that even if ecological systems are neither observable nor controllable through economic instruments, they may still be 'stabilised' (Perrings, 1991). What this means is that economic instruments may be used to restrict pressure on the ecological system so that it can continue to function over the expected range of environmental conditions. That is, economic instruments can be used to protect the resilience of the ecological system. This involves the use of ecologically sensitive 'safe minimum standards' (Bishop and Ready, 1991) supported by appropriate economic incentives. Instruments that safeguard the range of future options by protecting thresholds of resilience are generally defined as sustainability constraints.

The stabilisation of ecological-economic systems through the application of sustainability constraints implies a precautionary approach. The precautionary principle holds that where the effects of some activity are uncertain, but are potentially both costly and irreversible, society should take action to limit those effects before the uncertainty is resolved. The rationale for the principle is generally that the conjectured costs of not taking action are much greater than the known costs of preventative or anticipatory action (Taylor, 1991). This is partly just the notion that an implicit benefit cost analysis of activities with highly uncertain environmental effects should 'err on the side of caution'. But there is another side to the precautionary approach. Costanza *et al.* (1998) argue that where an activity is potentially damaging, the burden of proof should lie with those whose activities are the source of damage. That is, it is also a principle about who should bear the burden of proof.

Because the general problem of biological invasions, like the more specific problem of the spread of communicable diseases, depends on the independent decisions of millions of individuals, the control of that problem requires instruments and institutions that alter the incentives they face. These should reflect differences in the level of uncertainty, and the potential cost of the establishment and naturalisation of invasive species. Some introductions are entirely accidental. The spread of communicable diseases, for example, is seldom deliberate. However, most introductions involve deliberate imports to support agriculture, forestry and fisheries. Yet market prices for seeds, foods, fibres, pesticides and fertilisers do not generally reflect the ecological risks associated with their use. It follows that farmers have little incentive to take account of costs such as the deletion of indigenous species through predation, browsing or competition; genetic alteration of indigenous species through hybridisation; or the alteration of biogeochemical, hydrological and nutrient cycles, soil erosion and other geomorphological processes.

What are required are instruments that (a) protect key thresholds where the costs of crossing those thresholds are uncertain but are conjectured to be high and/or irreversible, and (b) confront people with the full cost of their behaviour wherever the risks are known. This implies a regulatory regime to protect key species, habitats and ecological services by controlling the introduction of potential invaders; an appropriate set of property rights in natural resources (along with their supporting institutions); a compensation mechanism; and a supporting structure of incentives and disincentives to induce the desired response. The instruments will typically differ for the ex ante control of species introductions and the ex post control of introduced species that have become invasive.

Because of the public good nature of strategies to protect ecological services, environmental authorities might be expected to insist on higher levels of environmental protection than would be provided by the market. But because the introduction of exotic species may also yield substantial benefits, environmental authorities would not be expected to insist on total protection. Although the potential irreversibility of the costs of invasions and uncertainty over damages both indicate a precautionary approach to the control of introductions, this does not imply the exclusion of all non-indigenous species.

Maintaining biodiversity in agroecosystems by precluding the introduction of crop species that displace traditional crops would certainly reduce the risks of biological invasions. It might also reduce the risks of crop failure. Higher levels of genetic diversity in cultivated crops and wild relatives generally means that yields can be maintained over a wider range of environmental conditions. Indeed, the genetic simplification of agriculture and forestry has already reduced the resilience of agroecosystems. More than 90% of world food supply derives from a small number of grasses (wheat, rice, corn and oats), nightshades (tomato and potato), mammals (cattle, sheep and pigs), and birds (chickens and ducks) (Heywood, 1995). The narrow genetic base of the food supply means that it is highly susceptible to disease and pest epidemics.

At the same time, preventing new introductions would also entail substantial losses in welfare, especially if forgone development benefits turn out to be high and the importance of existing biodiversity turns out to be low. At least half of the increase in agricultural productivity this century has been attributed to artificial selection, recombination and intraspecific gene transfer (Heywood, 1995). Enhanced disease resistance avoids very substantial damage costs. To take just one example, the use of Ethiopian barley to protect Californian barley from dwarf yellow virus was valued at \$160 million a year in 1995. These are very substantial benefits to forego. The exclusion of all exotics would impose considerable costs on society.

What we should be looking for is a regime that allows the social benefits of new introductions, whilst protecting society from the associated risks. The difficulty with new introductions is that the associated risks are generally uninsurable commercially for the reason that they are fundamentally uncertain and potentially very large. It is impossible to compute such risks actuarially. At present the risks of new introductions are typically born by the state in the receiving country. They are limited only by the quantity and effectiveness of resources committed to screening, and the exclusion policy adopted. The effectiveness of the screening process in turn depends on the resources committed to research into the consequences of the establishment and naturalisation of introduced aliens. The effectiveness of the exclusion policy depends partly on the nature of the constraints (black list or white list), partly on the resources committed to the detection and prosecution of non-compliance, and partly on the incentive effects of the penalty regime. Moreover, just as at the international level, the effectiveness of the national system is only as great as the effectiveness of the weakest link in that system.

It is, however, possible to make more constructive use of incentives in this area. There are precautionary instruments that can be used to protect society by altering the incentives to importers. The environmental assurance bond is an instrument that was developed to address the fact that, without market incentives, experimental research conducted by agents proposing innovative activities will not typically include all relevant potential future costs (Perrings, 1989; Costanza and Perrings, 1992). But because innovative activities are historically unique there is no basis on which to establish ex ante markets in all potential future effects. Sequentially-determined environmental bonds offer incentives to research the socially interesting outcomes of innovative activities.

In the case of potentially invasive species, assurance bonds would work in the following way. Importers of new species or those undertaking high risk activities would be required to post a bond equivalent to the conjectured damage if the species was to establish, naturalise and become a pest. This information might derive from the national screening service. But it might also derive from a central world or regional data/information source that could operate for invasive species generally in the same way as the Atlanta Centre for Disease Control operates for communicable diseases.

The value of the bond would then be reassessed as additional data on the environmental risks emerged. The bond would be refunded if it could be shown that there was no risk, or used to fund an eradication or control campaign if the risks were realised. An environmental assurance bond has the advantage that it both protects society and, by shifting the burden of proof on to those responsible for the introductions, it provides an incentive to research the ecological consequences of introductions. The use of an equivalent to the Atlanta Centre for Disease Control to provide data on the risks of invasive species has the additional advantage that it partly solves the problem of the weakest link.

The most effective ex ante control regimes will still not eliminate the risk of invasions. As in the case of screening and the control of species imports, the effectiveness of eradication, control or mitigation measures for introduced species that do become invasive depends both on the quantity and quality of public resources committed to these measures, and on the structure of incentives. The latter depends on an understanding of the way that human behaviour and invasive species interact. I have already made the point that is difficult to estimate the probabilities attaching to different outcomes of some control strategy solely on the basis of either the characteristics of species or their habitat. The control of invasive species depends on human behaviour, and the key element in any control strategy is likely to be the regulation of human behaviour. This may well imply the use of penalties to deter behaviour that increases the risks of invasions, but it may also imply the use of positive incentives to encourage behaviour that reduces those risks.

In this case, however, even if the incentives are right, the weakest link problem remains. The protection provided to all will only be as good as the protection provided by the weakest. This is a very practical problem with no ready solution. While a central source of information on invasive species may provide the data and technical advice to support eradication, control or mitigation, it would not have the resources to mount campaigns against particular invaders. Nor, at present, would UNEP in isolation. In the absence of a World Environment Organisation commanding sufficient resources to fill this role, there seems to be little alternative to the GEF sponsors, UNEP, UNDP and the World Bank. They should be urged to consider the establishment of a resource with the capability of protecting both global and regional interests from the threat of Biological Invasions by strengthening the weakest links in the chain.

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