CHAPTER 17

The economics of biodiversity and ecosystem services

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17.1 Introduction

The irreversible loss of genetic information (and the resulting loss of both evolutionary and technological options) caused by the extinction of species involves a global public good, the gene pool. Although important, it is not the only reason to be concerned about biodiversity change. As the Millennium Ecosystem Assessment (2005b) points out, another reason for concern is the role of biodiversity in the loss of ecosystem services. These also involve public goods, but unlike the public good associated with species extinction, they are almost always local or regional in extent. The conservation of species threatened with local extirpation protects a number of provisioning and cultural services, as well as the capacity of the local system to function over a range of environmental and market conditions. The latter may involve, for example, the regulation of specific biogeochemical cycles in different climatic conditions, or the protection of crop yields in the face of an array of pests and pathogens. In almost all cases, however, conservation of the functionality of particular ecosystems provides benefits to specific communities rather than to global society (Perrings and Gadgil 2003). Whether we focus on the gene pool or ecosystem services, however, biodiversity – the composition and relative abundance of species – is important because of its role in supporting the capacity of the system to deliver services over a range of environmental conditions. The economic problem of biodiversity, in this sense, differs from the economic problem of individual biological resources. The question is not at what rate to extract a particular resource, but how to balance the mix of species to assure a flow of benefits over a range of possible conditions.

Biodiversity conservation is frequently a public good. In many cases, nobody can be excluded from the benefits offered by the protection of assemblages, and if one person benefits it does not reduce the benefits to others. Because it is a public good, it will be 'undersupplied' if left to the market. The incentive that people have to free ride on the conservation activities of others means that people will collectively conserve too little biodiversity. At the same time the lack of markets for many of the biodiversity impacts of human activities mean that people are not confronted with the true cost of their decisions. Open access to scarce environmental resources is widely recognized to be a major cause of overexploitation.

Nowhere is this more clearly shown than in the world's fisheries. Worm *et al.* (2006) identified catches from 1950 to 2003 within all 64 large marine ecosystems worldwide: the source of 83 per cent of global catches over the past 50 years. They reported that the rate of fisheries collapses in these areas (catches less than 10 per cent of the recorded maximum) has been accelerating, and that 29 per cent of fished species were in a state of collapse in 2003. Cumulative collapses affected 65 per cent of all species fished. While property rights are generally better developed in terrestrial systems, many of the effects from anthropogenic land use change on biodiversity and ecosystem services are also not reflected by markets. Since habitat loss through land use change is the greatest single source of biodiversity loss, this is a major problem. Both the 'public good nature' of biodiversity conservation and the existence of biodiversity externalities mean that private decision-makers largely ignore the effect of their own behaviour on biodiversity, on ecosystem functioning, and, by and large, on the ecosystem services on which we all depend.

The economics of biodiversity and ecosystem services is largely about the failure of markets to signal the true cost of biodiversity change in terms of ecosystem services, the failure of governance systems to regulate access to the biodiversity embedded in 'common pool' environmental assets, and the failure of communities to invest in biodiversity conservation as an ecological 'public good'. This chapter reviews both the nature of the challenges posed by these failures and the options for addressing them. It requires that we are able to identify correctly both the private and social decision problems, and hence that we are able to value those non-marketed environmental effects that are ignored in many private decisions. Chapters 18 and 19 address the issues associated with the valuation and modelling, respectively, of the non-market effects of private decisions on biological resources. In addition, however, it requires that we are able to identify governance mechanisms, institutions, and instruments that will induce private decision-makers to behave in ways that are consistent with the social interest. This chapter focuses on the institutional and policy options for securing the socially optimal mix of species, given the role of biodiversity in assuring ecosystem services over a range of environmental conditions.

17.2 Biodiversity externalities in ecosystem services

Crocker and Tschirhart (1992) describe externalities of the kind described in the Millennium Ecosystem Assessment (2005b) as 'ecosystem externalities'. They define **ecosystem externalities** as: *market-driven actions that impact the wellbeing of either consumers or producers by altering the ecological functioning on which consumption or production depends, but where the welfare effects of those actions are ignored*. Thus, an ecosystem externality refers to the case where an individual's economic activity generates a real change in ecosystem services that impacts the wellbeing of others, but which is ignored by that individual. This may be because the activity involves the use of a public good (often a common pool environmental resource) where there is scope for free-riding, because of the incompleteness of markets, or because of the system of governance. Where the change in ecosystem services is mediated by a change in biodiversity, we refer to biodiversity externalities in ecosystem services.

For example, nitrogen oxides emitted from coal-fired power plants and mobile sources are a serious air pollutant that can directly impact human health – a traditional externality. Urea, ammonia, or ammonium applied as fertilizer to agricultural lands contributes to nitrate pollution of ground and surface water. In bays and estuaries, nitrate pollution is a serious problem that causes algae blooms and reduces abundances of desirable food species. Nitrogen applied to land has a more direct effect on biodiversity. A twelve year study of Minnesota grasslands (Tilman *et al.* 2001) showed that added nitrogen decreased species diversity and dramatically changed community composition. Species richness declined by 50 per cent and native bunch grasses were replaced by weedy European grasses. In both cases there is an ecosystem externality. The pollutant causes adaptations in the ecosystems affected, which in turn reduce the flow of ecosystem services in the form of reduced fishing, grazing and recreational opportunities.

The effect of biodiversity change on ecological functioning and the provision of ecosystem services has been most convincingly demonstrated for grasslands. In another set of Minnesota grasslands experiments in which biodiversity was deliberately varied across replicate plots, the capacity of the system to function over a range of environmental conditions was lowest where species richness was lowest. These and other experiments suggest that the loss or removal of species that function effectively in specific conditions reduces the range of conditions over which the system as a whole can operate (Tilman and Downing 1994, McGrady-Steed *et al.* 1997, Naaem and Li 1997). Tilman *et al.* (1996) describe this result, using reasoning from economics, as a portfolio effect. Increasing the number of species that fluctuate independently will decrease system volatility, just as increasing the number of independent assets in a financial portfolio will decrease the volatility of returns. Greater species richness is equivalent to greater diversification, which leads to lower variance.

In a closely related line of reasoning, Perrings *et al.* (1995, p. 4) state that 'the importance of biodiversity is argued to lie in its role in preserving ecosystem resilience, by underwriting the provision of key ecosystem functions over a range of environmental conditions'. Conserving biodiversity maintains species that may look unimportant for ecosystem function under current conditions, but which may play a crucial role in a drought, pest infestation or other shock (Walker *et al.* 1999). As above, conserving diversity can increase the probability of maintaining the flow of desired services over a range of potential environmental conditions. Resilience has been defined in two ways in the ecological literature. One, due to Pimm (1984), is the speed with which an ecosystem returns to equilibrium after a shock. A second, due to Holling (1973), is the magnitude of the shock that can be absorbed by an ecosystem without losing functionality – effectively the maximum perturbation that be accommodated without flipping from one state (stability domain) to another. The Holling resilience of a system increases with both the resistance/robustness of that system (the extent to which a perturbation moves it from the equilibrium) and its flexibility/adaptability (its ability to accommodate perturbation without loss of functionality). The economics literature has tended to adopt the second of these two ideas. Under either definition, though, if the reference state is desirable, then greater resilience will increase welfare. If the reference state is undesirable, greater resilience will reduce welfare. The biodiversity problem, in this case, is to choose the mix of species that will maximize some index of human wellbeing, given both the expected range of environmental conditions (shocks) and people's aversion to risk. As Chapter 19 shows, the measure of biodiversity used will depend on the nature of the decision problem – on the aspects of biodiversity that matter for human wellbeing. If that involves maintaining the options open to society in an uncertain world, then the right way to think about biodiversity is as a portfolio of biological assets or a risk-pooling mechanism. That is, biodiversity limits the variability in the supply of provisioning and cultural services.

17.3 Biodiversity as insurance

To be more precise about this, consider a manager concerned to maintain the flow of some ecosystem service – say food crops in an agro-ecosystem – operating under uncertainty due to stochastic fluctuations in environmental conditions. We can analyze this problem using the approach developed by Baumgärtner (2007), Baumgärtner and Quaas (2006, 2008) and Quaas and Baumgärtner (2008). The manager chooses a level, ν, of agrobiodiversity by selecting a portfolio of different crop varieties. Given this choice the manager realizes a crop yield at level s which is random. For simplicity we may assume that the agro-ecosystem service directly translates into monetary income and that the mean level, $Es = \mu$, of yields is independent of the level of biodiversity and is assumed to be constant. However, the variance of agro-ecosystem service depends on the level of agrobiodiversity:

$$\operatorname{var} s = \sigma^2(v) \quad (17.1)$$

where $\sigma^{2'}(v) < 0$ and $\sigma^{2''}(v) \geq 0$. The farmer's private decision on the level of agrobiodiversity affects not only his private income risk, as expressed by the variance of on-farm agro-ecosystem service, but also causes external effects. Suppose that $B(v)$ defines the sum of external benefits of on-farm agrobiodiversity, and that this takes the form of a reduction in the variance of some public ecosystem service:

$$EB(v) = \Xi \quad (17.2)$$

$$\operatorname{var} B(v) = \sum\nolimits^2 (v) \quad (17.3)$$

where $\sum^{2'}(v) < 0$ and $\sum^{2''}(v) \geq 0$ To see the role of agrobiodiversity, suppose that the manager has the option of buying financial insurance by choosing some level $a \in [0,1]$ of insurance coverage, paying

$$a(Es - s) \quad (17.4)$$

to the insurance company as an actuarially fair premium if the farmer's realized income is below the mean income plus any transaction costs of insurance. The latter are measured by:

$$\delta\, a \operatorname{var} s, \quad (17.5)$$

where the parameter $\delta \geq 0$ describes the 'costs' of insurance. The higher the insurance coverage, a, the lower is the risk premium of the resulting income lottery. The farmer chooses the level of agrobiodiversity, ν, and financial insurance coverage, a. A higher level of agrobiodiversity carries costs $c > 0$ per unit of agrobiodiversity. Hence the manager's (random) income is given by

$$y = (1 - a)s - cv + aEs - \delta\, a \operatorname{var} s. \quad (17.6)$$

Increasing a to one allows the farmer to reduce the uncertain income component to zero. The mean and variance of the farmer's income are determined by the mean and variance of the agro-ecosystem service, which depends on the level of agrobiodiversity:

$$Ey = \mu - cv - \delta a \sigma^2(v) \quad (17.7)$$

$$\operatorname{var} y = (1 - a)^2 \sigma^2(v). \quad (17.8)$$

Mean income is given by the mean level of agro-ecosystem service, μ, minus the costs of agrobiodiversity, cv, and the costs of financial insurance, $\delta a \sigma^2(v)$. For an actuarially fair financial insurance contract ($\delta = 0$), the mean income equals mean net income from agro-ecosystem use, $\mu - cv$. The variance of income vanishes for full financial insurance coverage, $a = 1$, and equals the full variance of agro-ecosystem service, $\sigma^2(v)$, without any financial insurance coverage, $a = 0$.

The farmer is assumed to be risk-averse with respect to his uncertain income y. Specifically, a general form of an expected utility function can be assumed, where $\rho > 0$ is a parameter describing the farmer's degree of risk aversion:

$$U = Ey - \frac{\rho}{2} \operatorname{var} y \quad (17.9)$$

Social welfare is assumed to be the expected welfare stemming from individual income and the public benefits of on-farm biodiversity. Furthermore, it is assumed that the private and the public risks associated with biodiversity are uncorrelated. Specifically, we assume an expected welfare function of the mean-variance type, where the parameter $\Omega > 0$ describes the degree of social risk aversion:

$$W = Ey + EB - \frac{\rho}{2} \operatorname{var} y - \frac{\Omega}{2} \operatorname{var} B. \quad (17.10)$$

In the private optimum, the farmer chooses the level of agrobiodiversity and financial insurance coverage so as to maximize his expected private utility (17.9) subject to constraints (17.7) and (17.8). The resulting allocation has the property that equilibrium levels of both agrobiodiversity and financial insurance coverage increase with the degree of risk-aversion:

$$\frac{dv^*}{d\rho} > 0 \text{ and } \frac{da^*}{d\rho} \geq 0, \quad (17.11)$$

with strict equality at the corner solution $a^* = 1$. The equilibrium level v^* of agrobiodiversity increases, and the equilibrium level a^* of financial insurance coverage decreases, with the costs of financial insurance:

$$\frac{dv^*}{d\delta} > 0 \text{ and } \frac{da^*}{d\delta} \leq 0, \quad (17.12)$$

with strict equality at the corner solution $a^* = 0$. The manager will choose the level of agrobiodiversity so as to equate its marginal benefits and marginal costs, where the marginal benefits comprise both the insurance value of agrobiodiversity and the reduction in payments for financial insurance that results from the reduced variance of agroecosystem service due to a marginal increase in agrobiodiversity. Where financial insurance is available, the manager will choose a level of agrobiodiversity that is below the one he would choose if financial insurance were not available.[1]

The socially optimal allocation ($\hat{v}$, $\hat{a}$) is derived by choosing the level of agrobiodiversity and financial insurance coverage so as to maximize social welfare (17.10), subject to constraints (17.2), (17.3), (17.7), and (17.8). The efficient allocation is such that both agrobiodiversity and financial insurance coverage increase with the degree of individual risk-aversion, i.e.:

$$\frac{d\hat{v}}{d\rho} > 0 \text{ and } \frac{d\hat{a}}{d\rho} \geq 0, \quad (17.13)$$

with strict equality in the corner solution $\hat{a} = 1$. The efficient level of agrobiodiversity increases with the degree of social risk-aversion, but the efficient level of financial insurance coverage is unaffected by the degree of social risk-aversion, i.e.:

$$\frac{d\hat{v}}{d\Omega} > 0 \text{ and } \frac{d\hat{a}}{d\Omega} = 0. \quad (17.14)$$

The efficient level $\hat{v}$ of agrobiodiversity increases with the costs of financial insurance, and the efficient level $\hat{a}$ of financial insurance coverage decreases with the costs of financial insurance:

$$\frac{d\hat{v}}{d\delta} > 0 \text{ and } \frac{d\hat{a}}{d\delta} \leq 0 \quad (17.15)$$

where equality may hold in the corner solution $\hat{a} = 0$.

The difference between the socially and privately optimal allocation is that the positive externality of a private farmer's effort is fully internalized in the socially optimal solution. By contrast, in the private optimum the manager chooses a level of agrobiodiversity that is too low. There are different ways that the social optimal solution can be reached by creating the right conditions for farmers. One possibility is by providing a subsidy, $\hat{\tau}$, on the conservation and utilization of agrobiodiversity. This should align the private decisions with the social optimal agrobiodiversity level, i.e.:

$$\hat{\tau} = -\frac{\Omega}{2}\sum\nolimits^{2'}(\hat{v}) > 0 \quad (17.16)$$

The size of the subsidy is increasing in the degree of social risk aversion, Ω, and decreasing in the degree of individual risk aversion, ρ, and the costs δ per unit of financial insurance:

$$\frac{d\hat{\tau}}{d\Omega} > 0, \quad \frac{d\hat{\tau}}{d\rho} < 0, \quad \frac{d\hat{\tau}}{d\delta} < 0. \quad (17.17)$$

The optimal subsidy, $\hat{\tau}$, can be interpreted as a measure of the financial flow needed to internalize the externality, i.e. to solve the public good problem. Thus it can also be interpreted as a measure of the size of the externality.

Although this problem is posed in the context of an agro-ecological problem, the same insights apply to the management of biodiversity in regulating the supply of the full range of provisioning and cultural services. Indeed, even though the Millennium Assessment (2005b) described the regulating services in terms of a very specific set of buffering functions, they actually summarize the role of the portfolio of biological assets in protecting us against the vagaries of both nature and society.

17.4 Biodiversity markets

In all cases, appropriate policy interventions depend on both a comparison between the privately and socially optimal outcomes, and the development of instruments that will induce private decision-makers to behave in ways that are consistent with the social interest.

In some cases, markets are already developing that allow biodiversity conservation to pay for itself (e.g. by establishing property rights in the effects of biodiversity change). As with the agro-ecological examples discussed in Section 17.3, these are cases where biodiversity supports the production of valuable goods and services that can, under the right circumstances, be sold in the market. Doing so may generate enough revenue to make conservation financially viable. This point is the core thesis behind several recent books (Heal 2000, Daily and

[1] This level can be determined from setting $a = 0$ in Eqn (17.11) and maximizing over v. It is strictly smaller than v^* for all $\delta < p$ and equals v^* for $\delta \geq p$, i.e. in cases where financial insurance is so expensive that an optimizing farmer would not buy it.

Ellison 2002, Pagiola 2002, OECD 2004). Market creation stems from a simple but powerful idea, i.e. markets can be devised to signal the opportunity cost to local land users of agricultural practices that affect biodiversity either positively or negatively. Ideally, such incentives need to address both '*forward*' (or '*downstream*') links from land users' decentralized decisions to biodiversity levels and '*backward*' (or '*upstream*') biodiversity linkages, i.e. from changing the stock of biodiversity level and its functional impacts on productivity to land users such as farmers and, thus, work at the landscape level (Pascual and Perrings 2007). But this implies that such incentives may affect the livelihoods of large numbers of land users. This adds a further layer of responsibility to public agencies to be aware of the distributional implications of alternative incentive measures.

One example in which conservation is currently being made financially attractive is ecotourism. The World Tourism Organization estimates that tourism generated revenues of $463 billion in 2001. One of the fastest growing segments of tourism may be nature-based or ecotourism. Some areas have had a long history of profiting from the richness of the local biodiversity, including Yellowstone National Park in the USA, Krueger National Park in South Africa, and a variety of National Parks in Kenya and Tanzania. Costa Rica has also done well promoting ecotourism, with approximately 1 million tourists spending $1 billion in 2000 (Daily and Ellison 2002, p. 178.). Several economic studies have found that ecotourism can generate significant revenues in a variety of developing country settings (e.g. Aylward *et al*. 1996, Lindberg 2001, Maille and Mendelsohn 1993, Wunder 2000).

A second example is bioprospecting for useful genetic material from plant or animal species that may lead to the development of valuable pharmaceuticals or other products. Pharmaceutical firms actively screen organisms in search of such active compounds as part of their intensive research and development programs. Bioprospecting is the term used to describe the process of testing natural organisms for these biochemically active compounds. If identified as active, a compound can result in the development of a new drug based on the natural compound itself (as in the case of vincristine and vinblastine found within the Rosy Periwinkle (*Vinca rosea*) or based on a synthetic compound developed from the blueprint provided by the natural compound. In either case, access to natural compounds is of great use in the research and development process. It has been estimated that 25 per cent of the drugs sold in developed countries and 75 per cent of those sold in developing countries were developed using natural compounds (Pearce and Puroshothamon 1995), suggesting that extant biodiversity is of value to pharmaceutical firms in their efforts to develop new drugs.

The CBD recommends a structure for bioprospecting agreements to accomplish three main goals: the conservation of biological diversity, the sustainable use of natural products, and the fair and equitable sharing of benefits derived from genetic resources (Article 1). They imply the existence of intellectual property rights in the face of current patent systems by which, for example, the pharmaceutical and seed industries can realize the monopoly benefits of new product development that are guaranteed by the ability to patent discoveries. Simpson, Sedjo, and Reid (1996) model the research and development process as a search through a list of research leads and conclude that the value of the marginal lead is generally insufficient for pharmaceutical firms to play a role in the conservation of biodiversity (see also Costello and Ward 2006). However, in the presence of competition, it is no longer the case that discovery of a single active compound is sufficient to guarantee a monopoly position within the market (Conte 2007). In a competitive search environment, the revenues associated with discovery will depend on the proportion of total successes controlled by the firm and a firm may have the incentive to preemptively exclude its competition from searching a portion of the research leads by signing bioprospecting agreements with host nations.

In recognition of the importance of IPRs to innovation, the Agreement on Trade Related Aspects of Intellectual Property Rights (1995) (TRIPS Agreement), mandates that all member nations of the World Trade Organization enact national legislation to provide minimum standards and scope of IPR protection (Strauss 1996). While all member nations have complied, there is still heterogeneity in the security of these property rights across nations, which might explain the pattern of existing agreements across countries. The importance of IPR security might also explain why some companies have made agreements with botanical gardens in developed countries for access to samples from tropical countries, as there is less uncertainty associated with the IPRs in developed nations (Sampath 2005).

There are unanswered questions about the optimal allocation of rents from a bioprospecting agreement. Consider, for example, the case of the rosy periwinkle mentioned earlier, a plant native to Madagascar that contains vincristine, a powerful cancer-fighting compound. No synthetic substitute for vincristine exists, and one ounce of vincristine requires 15 tons of periwinkle leaves. This has resulted in depletion of nearly the entire native periwinkle habitat in Madagascar (Koo and Wright 1999), though the plant has been extensively cultivated elsewhere. However, if drug companies do not keep a significant fraction of rents from developing new drugs they may not have sufficient incentive to develop new drugs via bioprospecting. Mendelsohn and Balick (1995) found a significant difference between likely social and private returns to development of new drugs. Koo and Wright (1999) also argue that biodiversity will be underprovided by the private sector via bioprospecting on the grounds that although the value of biodiversity is very large, market and social values are grossly misaligned.

It should also be noted that any added value of biological resources is created at each step of the innovation process – through the contributions of the local communities and research laboratories to industrial applications – and not only at the final stage of the innovation process. The existing IPR system only addresses the final stage of the innovation process, thus casting doubt as to whether IPRs are sufficient to induce the socially optimal level of conservation (Goeschl and Swanson, 2002; Dedeurwaerdere *et al*. 2007).

Indeed, both ecotourism and bioprospecting have been subject to criticism that revenues generated by conservation activities have not necessarily resulted in benefits to local communities. Local communities with no financial stake in conservation or that in fact suffer financial losses from conservation activities (e.g. wildlife damage to crops) might resent or actively oppose such activities, leading to a greater probability that conservation will fail. Trying to give local communities a stake in conservation has led to efforts to promote community-based conservation (Western and Wright 1994) and integrated conservation–development projects, or ICDPs (Wells and Brandon 1992). The goal of community-based conservation is to give local communities control over resources, thereby giving the community a stake in conservation. The most well-known community-based conservation program is the Communal Areas Management Program for Indigenous Resources (CAMPFIRE) in Zimbabwe (see Barbier 1992 for an early review and economic assessment). ICDPs try to 'link biodiversity conservation in protected areas with local socio-economic development' (Wells and Brandon 1992). Both approaches arose because of the failure of traditional protected areas conservation strategies that ignored the needs of local communities.

The extent to which conservation and local control over resources, or local economic development, are mutually consistent remains to be seen. Overall, community-based conservation and ICDPs have had mixed success to date. There is no guarantee that once they are given the choice, local communities will in fact choose to conserve. Cultural, social, or political factors may block conservation even when economic factors favour conservation. There is also no guarantee that conservation and local economic development are in fact consistent goals. Certainly in some communities with ecotourism potential, or where ecosystems provide valuable ecosystem services, conservation and development may go hand in hand. In other cases, the conservation of biodiversity and economic development may not be consistent. Because of the pervasive nature of external benefits created by biodiversity conservation, it may require more than just allowing local control and market forces to achieve an efficient level of conservation.

Recognition that the conservation of biodiversity may generate benefits that reach well beyond the local community provides a rationale for governments and non-governmental organizations to provide resources for conservation, and for the institution of national or international conservation policies. At present, though there are a number of policies to promote conservation, there are also a number of policies that have the opposite effect. Agricultural subsidies, subsidies to clearing land, resource extraction, and new development may all contribute to driving a further wedge between private and social returns from actions that conserve biodiversity. Perhaps the first rule for policy should be to 'do no harm'. Beyond doing no harm by eliminating perverse subsidies, however, positive external benefits from conservation require policies that create positive incentives to conserve.

Both governments and non-governmental organizations, such as the Nature Conservancy and World Wildlife Fund, are actively engaged in acquiring land for conservation and in other activities promoting conservation. Buying land is a direct and secure way to promote conservation, but it is often a costly instrument for protecting biodiversity. Boyd *et al.* (1999) find that acquisition is often 'conservation overkill'. Conservation easements that rule out certain incompatible land uses, but not all land uses, are often a far cheaper route to secure conservation objective than acquisition. Recently, interest has shifted away from land acquisition toward conservation easements and other ways of working with landowners to promote both conservation and landowner interests. For example, The Nature Conservancy's approach, once heavily weighted toward acquisition, now incorporates mechanisms such as community development projects to reduce the demand for fuelwood and the purchase of conservation easements to limit development (see http://www.nature.org/ for examples).

Acknowledging that donors from high-income nations invest billions of dollars toward ecosystem protection in low-income nations, a related literature debates the relative merits of direct conservation payments versus indirect mechanisms (e.g. payments to promote ecotourism which generates ecosystem protection as a joint product). Although indirect approaches are the predominant form of intervention in low-income countries, Ferraro and Simpson (2002) argue that

Box 17.1 The impact of market and non-market institutions on forest biodiversity and timber extraction: a study in northern India

The institutions that govern forests affect both the diversity of the forest stock and the mix of products and services that are extracted. A study of timber extraction from forests of the north Indian state of Uttar Pradesh from 1975 to 2000 shows how timber harvest is related to institutional conditions, species richness and the ecological characteristics of the forest, as well as to forest stocks, and harvest effort. Using a modified Gordon–Schaefer production function, and the assumption that forests are managed for 'sustainable timber extraction', the reduced form equations are derived and estimated (Chopra and Kumar 2004). The composition of products extracted is determined by their value, high value products being given priority. The modified model includes a bio-economic diversity index defined as $\sum_i (P_i Y_i / TR)^2$ where Y_i denotes harvest of the ith species, P_i denotes the price of the ith species, and $TR = \sum_i P_i Y_i$. The index is postulated to impact extraction as a shift factor in the extraction function. It is a weighted index of biodiversity in which prices are used as weights for the different Y_i. A loss of biodiversity is reflected in an increase in the value of the biodiversity index, which ranges between 0 and 1. The effect of the bio-economic diversity measure on timber productivity is captured by the introduction of an extra term, B, in the timber production function:

$$Y = qBEX \quad (17.1.1)$$

where E is effort and X the aggregate biomass of all species. Eqn. (17.1.1) implies that $Y/E = F(B, X)$, i.e. that the effort involved in timber extraction is inversely related to B. As B decreases (or as biodiversity increases), the extraction function shifts, resulting in a lower effort per unit effort. In other words, the model with the biodiversity index yields a lower level of Y for the same level of E, since $0 < B < 1$. If the forest manager is primarily interested in timber extraction, this results in a substitution of plantation forests for natural forests, so changing the ecological properties of the forest.

To capture this, Chopra and Kumar introduce W (the share of plantation forest in total area) in the growth function for timber biomass:

$$\dot{X} = rX(1 + eW - X/K) - qBEX \quad (17.1.2)$$

in which e is a coefficient for impact of W on growth of timber stock. They postulate that extraction increases as W increases. They further assume that $B = F(W)$. Since B increases as W increases, forests become less diverse. The estimated harvest equation is:

$$log(U_{bt}/U_{bt-1}) = r + 0.0853\,E_t + 0.00169U_b{}^*{}_t + 8.7454W_t{}^{**} \quad (17.1.3)$$

in which U_{bt} is biodiversity adjusted extraction (per unit effort), * indicates significance of the coefficient at 5 per cent level and ** denotes significance at the 1 per cent level. Effort E_t is not, by itself, a significant determinant of trends in extraction in this formulation, but both U_{bt} and W_t are significant. Extraction increases over time as the plantation area increases. Since W is inversely related to the level of biodiversity in the forest, a decreasing biodiversity due to a larger ratio of plantation to natural forest leads to rising trends in extraction. Extraction per unit effort increases as U_{bt} increases. Further, assuming E to be constant, they show that U_{bt} (defined as Y/BE) may increase under the following conditions with respect to the biodiversity index B.

1. With a rising B (falling biodiversity), if Y rises faster than B (extraction rises faster than biodiversity falls) U_{bt} increases and extraction of timber over time decreases. A rising extraction with decreasing biodiversity of the forest pushes the system towards a state in which increases in extraction take place at an increasing rate.
2. With a falling B (rising biodiversity) U_{bt} could decrease provided Y is not rising faster than B is decreasing, leading to a decreasing trend in extraction in subsequent periods.
3. With a constant B (constant levels of biodiversity) increases in U_{bt} are determined by changes in Y.

direct payments can be far more cost-effective, often requiring no additional institutional infrastructure or donor sophistication.

17.5 Economic instruments

Where biodiversity markets do not exist, and where market-driven behaviour leads people to select a combination of species, ecosystems, and landscapes that is not socially optimal, economists have developed a range of market-like instruments for encouraging socially desirable behaviour. The application of these instruments has been widely endorsed, and a number of countries make use of one or more of them. The OECD (2004), for example, makes the following recommendation to member countries:

1) establish and apply a policy framework aimed at ensuring the efficient long-term conservation and sustainable use of biodiversity and its related resources. The overarching goal of such a framework should be to ensure maximum net benefits, both now and in the future, from the use and conservation of resources stemming from biodiversity – as well as an equitable sharing of these benefits that is consistent with national, and applicable international, legislation;
2) make greater and more consistent use of domestic economic instruments in the application of their biodiversity policy frameworks, while attempting to reach further agreement at the international level on the use of economic-based policy instruments with respect to biodiversity conservation and management;
3) integrate market and non-market (i.e. non-price) instruments – taking account of the respective advantages of each in lowering information and transactions costs, and in addressing the 'public' values of biodiversity – into an effective and efficient mix of policies; and
4) integrate biodiversity policy objectives in a cost-effective manner into government sectoral policies, in order to avoid undue adverse effects on biodiversity and its related resources.

The set of instruments proposed by the OECD is amongst those described in Table 17.1. In what follows we highlight those instruments that are currently attracting attention.

Payments for ecosystem services The most direct way to create positive incentives for conservation is to institute a system of payments for the provision of ecosystem services (ES). Payments (or Rewards)

Table 17.1 Instruments for encouraging the socially optimal use of biodiversity.

Category	Type of instrument	Example
Economic incentives	Fees, charges, and environmental taxes Payments for ecosystem services Assignment of well-defined property rights Reform or removal of harmful subsidies	Charges or non-compliance fees related to certain types of forestry activities Liability fees for the maintenance or rehabilitation of ecologically sensitive lands Fishing license fees or taxes (whose objective is resource management) Levies for the abstraction of surface water or groundwater Liability payments for biodiversity damages (including interim losses) Charges for: Use of public lands for grazing in agriculture Use of sensitive lands Hunting or fishing of threatened species Tourism in natural parks Payments to farmers within a watershed for using farming techniques that maintain the quality of water resources Auctioned conservation contracts
Funds	Environmental funds and public financing	Global Environment Facility funding of local biodiversity conservation where there are global benefits
Framework incentives	Information provision, scientific and technical capacity building Economic valuation Market creation Institution-building and stakeholder involvement	Global Biodiversity Observation Systems Development of natural resource accounts Inclusion of biodiversity in Adjusted Net Savings Measures Support for biodiversity-related labelling schemes Strengthening governance of local, regional and global common pool biological resources Transferable development rights

for ecosystems services, P(R)ES, are voluntary transactions, not necessarily of a financial nature, in the form of compensation flows for a well-defined ES, or land use likely to secure it. The notion of 'rewards' is used to acknowledge that transactions from beneficiaries to providers may not need to be based on a financial flow. It can also involve in-kind transactions that may include a myriad of valuable goods and services from the beneficiaries' point of view, which can take intangible forms in diverse situations, such as knowledge transfer. P(R)ES is paid/rewarded by the beneficiaries and shared by the providers of the ES after eventually securing such compensation. P(R)ES are often designed to address problems related to the decline in some environmental services, such as the provision of water, soil conservation, and carbon sequestration by upland farmers who manage forest-lands in upper watersheds. In essence, such compensations are intended to internalize the positive externalities generated by upland farmers who can maintain the flow of valuable services that benefit lowland farmers or urban dwellers. However, a key obstacle in the successful implementation of P(R)ES arises at the 'value demonstration' stage, especially due to the scientific uncertainties underpinning the linkages between alternative land uses and the provision of the targeted environmental services.

The country that has moved furthest in this direction is Costa Rica. The 1996 Forestry Law instituted payments for ecosystem services. The law recognizes four ecosystem services: mitigation of greenhouse gas emissions, watershed protection, biodiversity conservation, and scenic beauty. The National Forestry Financial Fund enters into contracts with landowners who agree to do forest preservation, reforestation, or sustainable timber management. Funds to pay landowners come from taxes on fuel use, sale of carbon credits, payments from industry, and the Global Environment Fund

(GEF). Many developed countries have adopted some form of 'green payments' in which agricultural support payments are targeted to farmers who adopt environmentally friendly management practices or land uses (OECD 2001).

P(R)ES cannot be properly designed or implemented without a clear understanding of the property right regimes. Property rights regimes in natural resource management comprise a structure of rights to resources, rules under which those rights are exercised, and duties binding both those who possess the right(s) and those who do not. As Bromley (1991, pp. 2) puts it, '[p]roperty *is not* an object but rather is a social relation that defines the property holder with respect to something of value ... against all others'. In this context, Costa Rica is one of the few examples where an elaborate nationwide PES program is in place under a clear property rights regime. Under this program, only farmers with property rights to land can be paid for the environmental conservation they provide (Pagiola 2002).

A recent illustrative example of the potential effectiveness and flexibility of P(R)ES programs is that of RUPES: *Rewarding Upland Poor for Environmental Services*. The RUPES partnership comprises the International Fund for Agricultural Development (IFAD), the World Agroforestry Centre (ICRAF) and a group of local, national, and international partners.[2] RUPES aims to conserve environmental services at both global and local levels, while at the same time supporting the livelihoods of the upland poor in Asia. So far, the main focus has been on Nepal, the Philippines, and Indonesia, and the environmental services mostly include water flow and quality, biodiversity protection and carbon sequestration.

A variant of P(R)ES is the approach based on *direct compensation payments* (DCP) for taking private land out of production and into conservation (Swart 2003). Similar to other incentive mechanisms, the identification of the level of the efficient compensation payments to landowners requires the demonstration of an objective measure of its conservation value on both biological and economic grounds. In addition, the change in decentralized behaviour needs to be sustained into the future, which requires longer term political commitment. Asymmetric information between landowners and the compensating government agency is at least potentially problematic (Innes *et al*. 1998). If landowners expect compensation that is lower than the present value of the benefit stream arising from developing the land holding, they have a motive to develop quickly. Furthermore, even when exact compensation is foreseen by landowners, they may still have an incentive to intensify land use before compensation if this augments the market value of their property.

Transferable development rights Another approach to conservation is to institute a system of transferable development rights (TDR). TDR are virtually identical to cap-and-trade schemes to limit pollution emissions. In a TDR system, the conservation planner determines how much land can be developed in a given area. Development rights are then allocated and trades for the right to develop are allowed. Developers can increase density in a growth zone ('receiving area') only by purchasing development rights from the preservation area ('sending area'). The approach was developed and implemented extensively in the 1970s to direct development within urban areas (see Field and Conrad (1975) for what appears to be the first economic model of the supply and demand for development rights; see Mills (1980) for a model of TDR and a discussion of their appropriateness for use in protecting public goods).

Not until relatively recently have economists explicitly considered TDR as a mechanism to conserve biodiversity. Panayotou (1994) developed the TDR approach for conservation. He argued that 'biodiversity conservation is ultimately a development rather than a conservation issue' (Panayotou 1994, p. 91). Given that most biodiversity exists in the developing world, and that the public good nature of biodiversity requires a mechanism for paying developing countries to be stewards of this resource, Panayotou argues that TDR may also be an effective way to protect global (as well as local) biodiversity. Merrifield (1996) proposes use of a similar concept where 'habitat

[2] Some of the insights reflected here come from personal communication with Meine van Noordwijk, Tom Tomich and ICRAF personnel involved in RUPES program in Sumatra, Indonesia.

preservation credits' would be required for development. There is no guarantee that TDR schemes, like cap-and-trade schemes, will result in efficient outcomes unless the planner chooses the correct amount of rights/permits to allocate. An additional problem faced in TDR for conservation is deciding what are appropriate trades. Land units, unlike air emissions, have unique characteristics and may contribute to a number of conservation objectives. What constitutes an equal trade is not obvious. Similar problems over establishing the proper trading ratios exist in mitigation banking schemes for wetlands.

Having said this, TDR appears to be an innovative and cost-effective way to resolve the perverse incentives arising from DCPs. TDR extend the longstanding 'agro-ecological zoning' schemes, which aim to direct development to areas of high productivity potential and to restrict agricultural land use in ecologically significant and sensitive areas. However, such zoning programs do not allow for any substitutability between plots in meeting overall conservation goals. By providing a market-like alternative to the DCPs, flexibility in achieving conservation goals can be introduced. In this vein, the main advantage of TDR is that it can, in principle, encourage conservation on lands with low agricultural opportunity costs, while providing appropriate incentives to the affected landholders (Chomitz 1999).

In contrast to DCP, each landowner is issued tradable development permits by the government agency at an initial period. Subsequently, landowners hold the right to either develop or intensify their land holding. However, to develop that fraction of land a landowner needs to either use of one of the development permits (s)he holds or buy it from other landowners, who upon selling it can no longer develop their land fraction and instead must give it up for conservation. In this case, the government can share the cost of the 'takings', i.e. compulsory government land acquisition, with the landowners themselves.

Two main types of TDR programs exist at the landscape level: single and dual zoning. The former is similar to permit systems such as those used in transferable fishing quotas or pollution control. After the initial allocation of quotas, anyone within the program area may buy or sell the permits. The dual zone system instead explicitly designates both (permit) sending and receiving areas. This allows, for example, for new land use restrictions to be imposed on ecologically sensitive sending zone upon obtaining additional information about its higher conservation value and assigning TDRs to compensate for such additional restrictions. Usually, tight restrictions are also imposed on the receiving zone so as to increase the demand for TDRs (Chomitz 1999).

One of the forerunners of the TDR mechanism is in Brazil. While some initiatives have been proposed, the implementation is still under discussion. The basic idea is to give the opportunity for Brazilian agricultural land owners not complying with the National Forest Code (Law number 4771 approved on 15/09/1965) to buy forest reserves in other areas, normally in close proximity to their property. However, as pointed out by Pascual and Perrings (2007), a fully operational market for forest reserves is still to be implemented. Two examples are the National Provisionary Measure (Medida Provisória, Number 21666-67, approved on 24/08/2001), which amends the Forest Code and in the State of Sao Paulo (State Decree number 50889, approved on 16/06/2006).

Auction Contracts for Conservation (ACCs) One other way to induce private landowners to achieve desired level of supply of biodiversity conservation at the landscape level is by applying a competitive bidding or auction mechanism. An auction is a quasi-market institution with an interesting feature, i.e. it has a 'cost revealing' advantage compared to P(R)ES and DCP and can, in principle, be incorporated into a TDR system. In fact, the cost-revelation feature provides a way of generating important cost savings to governments. This is especially so when significant information asymmetry between farmers and conservation agencies exist regarding (i) the real opportunity cost of conservation and (ii) the ecological significance of the natural assets existing in farmlands. While the former is often better known by farmers themselves, the latter is normally better known by environmental experts. Such information asymmetries one reason for missing agrobiodiversity conservation markets. The idea is to use auctions to reveal the hidden information needed to recreate voluntary conservation contracts between landholders and the government.

In essence, landholders submit bids to win conservation contracts from the government. But while the latter prefers low bids, landowners need to submit bids that at least cover the opportunity cost of carrying out conservation activities. The problem is that information about such opportunity costs is often better known by resource users than by the government and the costs are also likely to be user-specific. Stoneham *et al*. (2007) provide a recent small-scale pilot case study of an auctioning system for biodiversity conservation contracts in Victoria, Australia, known as *BushTender*. The ACC involved 98 farmers, of whom 75 per cent obtained government contracts to conserve remnant vegetation in their farms after all farmers submitted sealed bids associated with their nominated conservation action plans. The selection of the farmers who won the contract was based on ranking the relative cost-effectiveness of each proposed contract. This involved weighting each private bid against the associated potential ecological impacts at the landscape level. Given a public budget of $400,000, contracts with bids that averaged about $4,600 were allocated and specified in management agreements over a three-year period. In total the contracts covered 3,160 ha of habitat on private land. Stoneham *et al*. (2007) have estimated that the *BushTender* mechanism has provided 75 per cent more biodiversity conservation compared to a fixed-price payment scheme (or DCP). In addition, they contend that given the relatively low enforcement costs in their pilot study, this ACC has interesting cost-effective properties. The pilot case study shows that it is possible to recreate the supply side of a market for agrobiodiversity conservation.

P(R)ES, DCP, TDP, and ACC all share an important characteristic for successful market creation for biodiversity conservation. For these mechanisms to be effective, accurate ecological and economic information at the demonstration, capture and sharing stages is needed. If it is not possible, or very costly, to convey clear and credible information about the nature of the services derived from biodiversity, the costs of supplying them, and the benefits derived for society, then the effect of implementing these economic mechanisms would be distorted and would lack precision. Moreover, it would be naïve to champion market creation for biodiversity conservation if other supporting institutions are also lacking, such as property rights to the resources in question (Pascual and Perrings, 2007). Furthermore, if markets for biodiversity are recreated without proper institutional and regulatory backup, then the social costs of such policies may well outweigh the benefits from conservation (Barrett and Lybbert 2000). In a second-best world where information is elusive, most policy initiatives pragmatically focus on ensuring that institutions are developed so as to keep future options open (Tomich *et al*. 2001). In fact, most conservation policies are aiming at developing flexible and open institutions that can mitigate the negative effects of intensification in agroecosystems, without foreclosing future land (de)intensification options.

An important qualification is that many market-like mechanisms have implications for the rights of the poor, particularly in low-income countries where people depend heavily on environmental resources (Dasgupta 2001). Pricing access to ecosystem services can cause the socially disadvantaged and vulnerable to be excluded from those services, and mechanisms need to be developed to address this. For example, in 1991 the Government of Uganda established a national park in the Bwindi forest to protect the mountain gorilla. This park was established with little consultation with the local populations who depended on the forests for their livelihood. As a result, poaching and encroachment were common. In 1995, the Mgahinga and Bwindi Impenetrable Forest Conservation Trust Fund was created, its proceeds being shared with the local communities to encourage sustainable development activities and conservation.

The general problem is that economic interventions that are efficient by the Pareto criterion (which states that an economic intervention is efficient if it benefits at least one person without leaving any other person worse off) may still leave people worse off in relative terms. One approach to this problem is to subject interventions to a second test: that the equity gap between individuals or groups after an economic intervention should be no larger than the gap before the intervention. In this way, if one individual has benefited from the economic instrument, then some transfer will need to take place to ensure that the gap between that individual and others will remain the same. In other words, some form of social redistribution mechanism will need to be institutionalized at the same time the

Box 17.2 The economics of the US Endangered Species Act (ESA)

While market-oriented policies have been of increasing importance in recent years, other important policies directed at the conservation of biodiversity, including the U.S. Endangered Species Act and the Convention on International Trade in Endangered Species, are at their core largely command and control regulatory regimes. The Endangered Species Act (ESA), enacted in 1973, changed conservation policy from a largely voluntary and toothless regime that existed prior to 1973 into a powerful environmental law capable of stopping large government projects and actions of private landowners (Brown and Shogren 1998). Section 7 of the ESA prohibits federal agencies from actions that cause 'jeopardy' (i.e. risk of extinction) to species listed as threatened or endangered. Section 9 prohibits public and private parties from 'taking' listed species. 'Taking' includes causing harm to species through adverse habitat modification from otherwise legal land uses, such as timber harvesting or building, as well as more obvious prohibitions against killing, injuring or capturing a listed species. The way the law is written, the ESA appears to have very limited scope for economic considerations. Sections 7 and 9 are absolute prohibitions. Biological criteria are the basis for listing species. In TVA v. Hill, the US Supreme Court wrote: 'The plain intent of Congress in enacting this statue was to halt or reverse the trend toward species extinction, whatever the cost' [437 U.S. 153, 184 (1978)]. When it looked like a small unremarkable fish (the snail darter) that was previously all but unknown would halt construction of a large dam backed by politically powerful members of Congress, Congress amended the ESA. They authorized the formation of the Endangered Species Committee ('The God Squad') to allow an exemption to the ESA if the benefits of doing so would clearly outweigh the costs. There are high hurdles to be met for convening this Committee and it has been used rarely.

Despite the fact that the law is written in a way that appears to marginalize economic considerations, it has proved impossible to administer the Act while totally ignoring economics. Several writers have noted that economic and political considerations influence agency actions at all stages of the ESA process including the listing stage, which is supposed to be done strictly on biological grounds (e.g. Bean 1991, Houck 1993). Endangered species whose protection threatens to impose large costs run into political opposition that translates into pressure on the Fish and Wildlife Service. This pressure appears to translate to lower probability of listing (Ando 1999). The benefits side of the equation also seems to affect listing and recovery spending even though the ESA does not base such decisions on the popularity of the species. Metrick and Weitzman (1996) found that more charismatic species were likely to be listed than uncharismatic species, and that once listed 'visceral characteristics play a highly significant role in explaining the observed spending patterns, while the more scientific characteristics appear to have little influence' (Metrick and Weitzman 1996, p. 3).

While much of the early regulatory activity under the ESA targeted government actions under Section 7, the 1990s saw an increase in the emphasis on conservation on private lands under Section 9. More than half of endangered species have over 80 per cent of their habitat on private land (USFWS 1997). Conservation on private lands presents a number of incentive issues (Innes *et al.* 1998). A landowner whose parcel contains an endangered species habitat may face restrictions on what activities may be undertaken. The landowner need not be compensated if restrictions are imposed and losses to the landowner result (though the law on regulatory takings is quite unsettled; see Polasky and Doremus 1998). The potential losses the ESA may impose on a landowner give rise to several perverse incentives. Innes (1997) shows that there can be a race to develop in order to beat the imposition of an ESA ruling. Similarly, there may be an incentive to 'shoot, shovel and shutup' in order to lower the likelihood of imposition of restrictions under the ESA (Stroup 1995). Further, because current law stipulates that acquiring specific information about species is a prerequisite to imposing restrictions on a landowner, there is no incentive for the landowner to cooperate in allowing biological information to be collected (Polasky and Doremus 1998).

There are several possible ways to reform the ESA to cure the worst of the perverse incentives. One method is to provide compensation. When eminent domain is used and there is a physical taking of property, the government is required to provide compensation equal to the market value of the property. The same approach could be taken when the government mandates conservation on private land. There are two potential problems with this approach. First, Blume *et al.* (1984) show that when landowners are fully compensated in the event of a taking, there is an incentive to over-invest. It is socially optimal to take account of the probability of future takings that render the investment worthless. The landowner is, however, fully reimbursed and so ignores this factor. Second, use of

Continues

Box 17.2 *(Continued)*

government funds to pay for compensation may be costly. On the other hand, others point out that there is an advantage to forcing regulators to understand the costs of imposing regulations by paying compensation (e.g. Stroup 1995). Rather than tying compensation to market value, paying compensation tied to the value of conservation along the lines of green payments discussed above, can generate efficient incentives to conserve (Hermalin 1995).

A different approach to reform is to allow landowners to avoid sanctions if they can prove that their proposed actions will not cause harm (Polasky and Doremus 1998). This type of approach is exemplified in the ESA by the provision to allow landowner actions that cause some minor and unintended harm to a listed species for landowners with approved Habitat Conservation Plans. The incentive for filing Habitat Conservation Plans was further sweetened by promises of 'no surprises' and 'safe harbors' that put the burden on the government for costs imposed by future regulatory actions.

economic instruments are being implemented. This however keeps the status quo of the existing equity gaps within society. A third test, which can be considered pro-poor, is that the net benefits accruing from the intervention are distributed according to some ratio whereby the increase in welfare of the worse-off individual is proportionately greater than the welfare increase of the best-off individual (Duraiappah 2006).

17.6 The international dimension

The problem of transboundary externalities resulting from the growth of international trade is the subject of a substantial literature. As with externalities in local markets, one focus has been the consequences of ill-defined property rights in environmental resources (Chichilnisky 1994, Brander and Taylor 1997, 1998, Rauscher 1997). The impact of trade on biodiversity as a specific environmental problem has been evaluated from two main perspectives. One focuses on the link between specialization under trade, habitat conversion, and species loss (Barbier and Schultz 1997, Polasky *et al* 2004). These studies calculate the impact of trade on biodiversity from the proportion of the land area that is converted to the production of primary commodities, the impact on existing species being taken from the species–area relationship (Macarthur and Wilson 1967). Polasky *et al.* (2004) extend the analysis to the two country case. The same mechanism operates in each country. They argue that if there are high levels of endemism in each country, and if consumers are concerned to protect local biodiversity, trade can reduce the level of welfare. But where species are common to both trading partners and consumers are interested in global rather than local levels of biodiversity, trade is unambiguously welfare-enhancing.

A second approach focuses on biological invasions as an externality of trade (Perrings *et al.* 2000, Perrings *et al.* 2002, Kohn and Capen 2002, Costello and McAusland 2003, McAusland and Costello 2004, Knowler and Barbier 2005). This literature considers both the problem of incentives to internalize biodiversity externalities of trade, and the problem of insufficient investment in biodiversity conservation as a public good. Costello and McAusland (2003) explore the use of tariffs on imports to reduce the damage costs from accidental introductions. While they show that import tariffs will always reduce import volumes of potentially invasive species, they find that tariffs can have adverse effects if they alter the composition of imports, or change land use in ways that make ecosystems more vulnerable to invasive species. McAusland and Costello (2004) consider the efficiency of port inspections combined with tariffs on imported goods, and find that the optimal tariff covers inspection costs plus the potential damage costs from outbreaks of pests undetected during inspections. The optimal level of tariffs in each case depends on the risk of biological invasions and the expected level of damage they cause. The public good problem in the case of invasive species involves the protection offered to all by measures to control the introduction of pests and pathogens. Since it is a 'weakest link' public good, the protection to all is frequently only as good as the protection offered by the weakest link in the chain. This has implications for the pattern of international investment in invasive species control (Perrings *et al.* 2002).

The persistence of international biodiversity externalities, like the persistence of other environmental externalities, has much to do with the ways in which international markets and the rules of international trade are structured. Unlike many other environmental resources, however, there does exist a treaty on the trade of biological species. The Convention on International Trade in Endangered Species (CITES) deals specifically with international markets for biological resources (as distinct from markets for the international benefits of local conservation effort). Its role is to reduce the impact of trade on the survival probability of rare and endangered species. It does this by imposing prohibitions. Examples include international bans on the trade in elephant ivory, and on timber obtained from certain endangered tree species (Barbier *et al.* 1994, Swanson 1995).

CITES has arguably been the international agreement that has had the greatest impact on conservation outcomes (WCMC 1992). CITES authorizes banning international trade in species listed under Appendix I, and regulating trade in species listed under Appendix II. In 1989, CITES initiated a ban on trade in ivory. In the 1970s and 1980s rampant poaching of elephants caused a drop in elephant populations of roughly 50 per cent (Barbier and Swanson 1990). Particularly threatened were elephant populations in east African countries. Elephant populations in southern African countries were less threatened. Imposing the ivory trade ban was controversial. Southern African countries with relatively healthy elephant populations (Botswana, Malawi, Namibia, South Africa, Zimbabwe) objected and did not sign on to the ban. Opponents of a ban argued that it would likely result in high ivory prices as supply was choked off, which would increase the rewards to poaching (Barbier and Swanson 1990). Opponents also argued that by denying rights to sell ivory legally there would be less financial reason to conserve elephant populations and less money available for enforcement efforts against poaching. Proponents of the ban, including east African countries and many developed countries, argued that without the ban elephant populations would continue to decline, as it was too easy to sell illegally harvested ivory and because anti-poaching efforts of impoverished governments were no match for well-organized poaching gangs. Van Kooten and Bulte (2000) summarize economic arguments about the ivory ban and present results from application of several dynamic models.

The argument for trade restrictions of the CITES type is that, in the absence of restrictions, there will be a 'race to the bottom'. Firms will seek to exploit the international advantages offered by relaxed labour and environmental conditions, and countries will use the lack of environmental protection to induce inward investment (Wheeler 2000). By this argument, biodiversity and other environmental externalities are not just an incidental product of market failures. They are the outcome of strategic decisions by governments and firms seeking a competitive advantage. The claim is that where trade agreements make it impossible either to induce inward investment or to protect domestic agriculture or industry through trade policy, countries may be encouraged to use environmental policies to the same effect. Specifically, they may be encouraged either to allow ecological dumping by relaxing environmental protection measures, or to use environmental regulation as trade protection measures.

The empirical evidence for a race to the bottom is mixed. The relocation of polluting industries from high-income to low-income countries is a part of the explanation for changes in environmental indicators observed in the Environmental Kuznets Curve literature (Barbier 1997, Cole *et al* 1997, Arrow *et al.* 1995). However, studies of the incentive effects of environmental regulation have concluded that the costs of compliance with environmental regulations are a sufficiently small proportion of total costs that they do not generally drive location decisions (Jaffe *et al.* 1995, Levinson 1996). Wheeler (2000) argues that the effects of income growth on environmental protection in low-income countries, along with the progressive empowerment of local communities affected by relocation, will be enough to avert a race to the bottom.

In fact, the environmental impacts of trade are one of the few acceptable justifications for imposing trade restrictions under the General Agreement on Tariffs and Trade (GATT). The exceptions allowable under

Article XX of the GATT, along with the Sanitary and Phytosanitary (SPS) Agreement, authorizes countries to impose restrictions on trade in order to protect human, animal and plant life. The evidence on the use of Article XX and the SPS Agreement provides some support for the notion that low-income countries do not, in general, use environmental measures to restrict trade. Both Article XX and the SPS Agreement have been successfully invoked in many circumstances. For example, in 1995–97 there were 724 measures notified under the SPS Agreement. Of these, 55 per cent were notified by high-income countries, 42 per cent by middle-income countries, and only 2 per cent by low-income countries (UNEP, 1999).

Despite Article XX and the SPS agreement, it is generally argued that the WTO is not the place to deal with the environmental effects of trade (Bhagwati 2000, Barrett 2000). This has led to growing pressure for the establishment of an environmental analogue to the WTO – either a World Environment Organization (WEO) (Whalley and Zissimos 2000) or a Global Environment Organization (GEO) (Runge 2001). It is interesting that the case for such an organization rests first and foremost on the fact that there is no other effective way of dealing with global environmental externalities. It is thought that a WEO/GEO could create new international markets for the global environmental benefits of local conservation effort. Existing examples of this, including joint implementation, bioprospecting contracts, debt-for-nature swaps, and transferable development rights, are first steps towards the creation of global markets for environmental benefits. The alternatives to date – multilateral environmental agreements – have been piecemeal, and have generally failed to address the important issues of compensation and penalties for non-compliance. While agreements involving small numbers of parties concerned with specific issues have been reasonably effective, the framework agreements involving much larger numbers of parties have been less effective (Barrett 1994, Barrett 2003).

17.7 Concluding remarks

The socially optimal use of biodiversity requires two problems to be solved. The first is the problem of local market failure, and is associated with the local public goods and biodiversity externalities. The second is the problem of international market failure, and is associated with the global public good protected by the international conservation effort and with the externalities of international trade. Both require the development of incentives to decision-makers to take the full costs of their actions into consideration, institutions for the regulation of access to biological resources, and an appropriate financial mechanism. The incentive problem has two elements. One is the generation of the correct incentives for the socially optimal use of biodiversity. The other is the discouragement of perverse incentives that work against this. The use of incentives to protect local public goods necessarily operates at the local level, where the millions of foresters, farmers, hunters, harvesters, herders, and fishers use environmental resources on a daily basis. It implies a package of direct incentives (taxes, subsidies, grants, compensation payments, user fees, and charges), indirect incentives (via fiscal, social, and environmental policies), and disincentives (prosecution leading to fines and other penalties).

Up to now the newer market-like mechanisms have emerged in areas where the capturable benefits are largest. The most direct attempts to do this involve the widening and deepening of markets for individual biological resources. Amongst the best-known examples concern the markets for forest-based pharmaceutical products. Bioprospecting contracts between individual pharmaceutical companies and developing countries, such as that between MERCK and INBIO in Costa Rica, have received a great deal of publicity. They seek to mobilize investment in biodiversity conservation by offering access to genetic resources, protected by the assignment of intellectual property in genetic ‘discoveries’ (Schulz and Barbier 1997). Although they are very well known, however, such contracts are not at all widespread, and have not generally yielded competitive rates of return (Barbier and Aylward 1996, Simpson *et al.* 1996, Pearce *et al.* 1999, Dedeurwaerdere *et al.* 2007).

A second set of markets offer biodiversity conservation benefits as a side-effect (an externality) of markets for unrelated effects. Joint implementation, or carbon offset arrangements, are promoted by the UN Framework Convention on Climate Change (FCCC). The arrangements allow one country – a high-income country – to meet its carbon emission targets under the Convention by investing in the reduction of carbon emissions or the sequestration of carbon in another country. The high-income country gains from the lower costs of reducing carbon emissions or sequestering carbon in the low-income country. The low-income country gains from the additional investment. Most joint implementation projects refer to improvements in energy efficiency along with investment in renewable energy or fuel switching. They do include some projects involving forest conservation and reforestation. However, the link between joint implementation and biodiversity conservation remains tenuous.

In conclusion, people's ability to maintain critical flows of ecosystem services are being lost because individual decision-makers have an incentive to destroy, or at least not sufficiently to protect, biodiversity. The remedy to this problem is to design governance mechanisms and incentives that encourage individuals to protect the common good, and to implement these incentives at all relevant scales through suitable policies, institutions, and financial mechanisms. A number of working examples of this approach already exist. They demonstrate its potential to improve the state of biodiversity and ecosystem services, and through that, to enhance human wellbeing.

Recent work in the field of ecological economics shows that stability adds additional economic value to ecosystem services in the form of insurance (Chapter 17), further underlining the importance of a thorough understanding of the effect of biodiversity on ecosystem functioning and associated services.