THE ECONOMICS OF BIODIVERSITY LOSS AND AGRICULTURAL DEVELOPMENT IN LOW INCOME COUNTRIES

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Introduction

Biodiversity conservation has traditionally been seen as problem of protecting genetic diversity. It has had two dimensions: *ex situ* germ plasm preservation in zoos, aquaria and arboreta (and by extension, seed banks, tissue cultures and genomic libraries), and in situ species preservation in refugia, especially in megadiversity areas involving high levels of endemism. Increasingly, however, biodiversity conservation is being taken out of zoos and protected areas. It is recognised that biodiversity is important for the functioning of all ecosystems, and that excessive loss of biodiversity imposes real costs on resource users (Heywood, 1995). It is therefore interesting to consider the problem of biodiversity loss not just in refugia, but in managed ecosystems. These are ecosystems from which some species have been deleted in order to enhance the productivity of others. The problem of biodiversity conservation in such cases does not therefore involve preservation of all existing species. It involves maintenance of sufficient interspecific and intraspecific diversity to protect the productivity of the system. Put another way, the problem of biodiversity conservation in managed systems requires us to think about the optimal or efficient level of species deletion. The main question I want to pose in this paper is whether current rates of biodiversity loss are efficient.

This is not an uncontroversial way to look at the problem. It implies that it is reasonable to apply conventional economic tests to biodiversity loss, and many regard such an approach with repugnance. Wilson (1984, 1993), for example, argues that humans have an inherent inclination to affiliate with life and life-like processes and these innate tendencies form a basis for an ethic of care and conservation of the diversity of life. But to say that there may be an efficient level of biodiversity implies that it may be optimal to drive some species to extinction (if only locally). While most people have little difficulty with this suggestion when the species at issue are, say, the AIDS or smallpox viruses, there is less consensus about endemic agricultural pests or competitors. Nevertheless, this is the approach I want to take: to consider whether current rates of biodiversity loss in agroecosystems are efficient.

The paper does not report original research results, but uses existing literatures in ecology and economics to consider three aspects of the problem. The first, addressed in section 2, is to identify the external costs of biodiversity loss in agroecosystems in developing countries. This section considers the biodiversity implications of agricultural growth. Agricultural growth takes two forms. Extensive growth leads to land conversion. This is associated with both habitat destruction and habitat fragmentation. It is generally seen as the major proximate cause of biodiversity loss. Intensive growth leads to an alteration in the mix of species due to changes in cropping or livestock regimes or to pest management practices. Intensification affects the combination of crops, livestock, symbiotics, competitors and predators. I argue that a reduction in the diversity of species in the system due to intensification may, in some cases, make agroecosystems more susceptible to exogenous shocks or changes in environmental conditions, and that this effect is not captured in market prices (Perrings et al., 1995; Conway, 1993).

The second, addressed in section 3, is the relationship between market failure and income. The costs of biodiversity loss turn out to be sensitive to the distribution of income and assets (Perrings et al, 1994). The Brundtland Report (WCED 1987) may be best known for putting sustainability on to the international policy agenda, but amongst its key arguments was the assertion that poverty induces environmental degradation. Dasgupta, in a number of contributions, has since explored the connection between poverty, population growth and environmental change (Dasgupta, 1993, 1995, 1996). His arguments lend formal support to the Brundtland view. At the same time, however, there is a growing empirical literature on the so-called Environmental Kuznets Curve (reviewed in Barbier, 1997) which seems to suggest exactly the opposite conclusion. It finds that environmental degradation is induced not by poverty but by development, and that indicators of environmental quality tend to be highest in countries where per capita income is lowest. I want to consider both the empirical evidence for a relation between indices of poverty and proxies for biodiversity loss, and the behavioural link between rural poverty and the underlying causes of biodiversity loss. More particularly, I want to ask how rural incomes may be related to the market failures, which drive biodiversity loss in low-income countries.

Finally, Article 11 of the Convention on Biological Diversity calls on the contracting parties to 'adopt economically and social sound measures that act as incentives for the conservation and sustainable use of components of biological diversity'. If market failures are driving biodiversity loss beyond the level that is justified by the gains from extensive and intensive growth of agriculture, what is the scope for developing instruments that will work in a developing country context? In a fourth section I consider what can be done through market based instruments, institutional and property rights reform to address the problem.

A final section offers my conclusions. In summary, these may be stated as follows. Biodiversity loss matters in agroecosystems for a number of reasons, the most important of which are that it reduces the capacity of farmers to cope with external shocks (whether market or environmental). These costs of biodiversity loss are external to the market-they involve market failure. The problem is most severe in low-income countries where mechanisms for private and social insurance against the risks to agricultural incomes are limited. Governments frequently act as insurers of last resort, distributing famine relief when farm incomes fail. Given the limited resources of governments in low-income countries, however, and given the fact that the risks to farm incomes are often highly correlated within such countries, this is seldom an effective solution. In the absence of effective private or social insurance mechanisms, the best way to deal with biodiversity externalities may be through the private costs of different farming systems. Where biodiversity-poor farming systems involve greater social cost, they should also involve greater private cost. My concluding remarks offer some concrete proposals on this.

2. The external costs of biodiversity loss in agroecosystems

The focus of this paper is the local costs of the deletion of species. In the case of endemics the local deletion of some species may well imply global extinction, but even in such cases I want to consider the costs to local resource users. This is rather different from much recent work on biodiversity loss which tends to focus on the global value of local conservation, and the scope for local capture of global values (see, for example, Pearce and Moran, 1994; Pearce, 1999). My concern, however, is with the local efficiency of biodiversity loss, and the scope for developing local incentives for biodiversity conservation.

A local focus draws attention to the role of biodiversity in the provision of locally valuable ecological services. Biodiversity supports a range of ecological services including, at the global level, the maintenance of the gaseous quality of the atmosphere and amelioration of climate. But it also supports a number of much more localised services: the operation of the hydrological cycle including flood control and water supply, waste assimilation, recycling of nutrients, conservation and regeneration of soils, pollination of crops and so on (Daily, 1997). The local value of biodiversity derives from the value of these services. In agroecosystems, for example, the most important ecological services are those influencing the productivity of the system, and its capacity to maintain productivity over a range of environmental conditions. These comprise both on and off-farm services. Watershed protection, for example, offers a range of off-farm services to agriculture including regulation of surface run-off, groundwater recharge, erosion control and localised climatic effects. The conversion of watersheds as a by-product of extensive agricultural growth often means the loss of these services. In what follows I consider the available evidence on value of changes in the mix of species both as a result of habitat conversion and of alternation in the intensity of farming activity.

Habitat conversion and degradation: the extent of the problem

The main proximate cause of biodiversity loss is the habitat loss associated with the processes of deforestation and desertification. Both processes are associated with areas where a high proportion of output and/or employment derives from agriculture. Specifically, biodiversity loss due to agricultural growth at the extensive margin is associated with regions of low population density but high population growth (Sub-Saharan Africa and Latin America). Biodiversity loss due to agricultural growth at the intensive margin is associated with regions with high rural population density and growth (South Asia and South East Asia). This said, as the productivity gains of agricultural intensification have faltered in Asia, so pressure in that region has gone back on to remaining forested areas, and recent rates of deforestation are higher in South and East Asia than elsewhere.

Consider, first, the process of deforestation. Table 1 reports deforestation in selected sub-regions for the period 1981-1990. Not only did deforestation accelerate in regions where the process was already under way at the beginning of the decade, but also afforestation turned to deforestation in other regions. In Sub-Saharan Africa the highest rates of forest loss occurred in West Africa-Ghana and Togo in particular. But the annual rate of loss in these countries, 1.3 to 1.4 per cent, was still low compared to regions where forest stocks are more depleted. Four countries in Latin America-Costa Rica, El Salvador, Honduras and Paraguay-were converting remaining forests at more than 2 per cent a year during 1980s, while Bangladesh, Pakistan, Thailand and the Philippines were all converting remaining forest resources at 2.9 to 3.0 per cent a year.

Table 1 Forests Resources and Deforestation, 1980-1990

	Extent of Natural Forest (1000 Ha)		Annual Deforestation (1981-1990)	
	1980	1990	(1000 ha)	(percent)
West Sahelian Africa	43,720	40,768	295	0.7
East Sahelian Africa	71,395	65,450	595	0.8
West Africa	61,520	55,607	591	1.0
Central Africa	215,503	204,112	1,140	0.5
Tropical Southern Africa	159,322	145,868	1,345	0.8
Insular Africa	17,128	15,782	135	0.8
South Asia	69,442	63,931	551	0.8
Continental South Asia	88,377	75,240	1,314	1.5
Insular South East Asia	154,687	135,426	1,926	1.2
Central America and Mexico	79,216	68,096	1,112	1.4
Caribbean Sub-region	48,333	47,115	122	0.3
Tropical South America	864,639	802,904	6,174	0.7

Sources: World Resource Institute 1994. World Resources 1994-1995. Oxford, Oxford University Press.

Compare this rate of habitat loss/fragmentation in forested areas with rates of change in arid and semi-arid areas. Desertification is the term most frequently used to capture environmental damage in arid, semi-arid and dry sub- humid areas. Like deforestation, desertification implies a reduction in the vegetative cover of land and tends to be associated with the expansion of agricultural output. Desertification currently affects some 3.6 billion hectares and some 900 million people. This makes it a more extensive a problem than that of deforestation (Table 2).

Desertification captures a range of different types of damage. In irrigated lands, for example, desertification is a consequence of the salinisation and alkali of soils and aquifers. Annual losses due to these causes in the early years of this decade were running at about 1.5 million ha. In rain-fed croplands the dominant manifestation of land degradation is soil erosion and the loss of soil organisms which account for at least 3.5 million ha annually. But in rangelands, degradation takes the form of loss or alteration of vegetation, loss of soil moisture and soil organisms and soil erosion. Annual losses at the beginning of the decade were estimated to be between 4.5 and 5.8 million ha (Tolba et al, 1992).

Table 2 Extent of Desertification of Drylands

	Moderately, severely and very severely desertified land (i.e. includes all lands at least moderately damaged in one of the ways encompassed by the term desertification)					
	Irrigated areas		Rain-fed croplands		Rangelands	
	(000 ha)	per cent affected	(000 ha)	per cent affected	(000 ha)	per cent affected
Africa	1902	18	48863	61	995080	74
Asia	31813	35	122284	56	1187610	75
Australia	250	13	14320	34	361350	55
Europe	1905	16	11854	54	80517	72
N. America	5860	28	11611	16	411154	75
S. America.	1517	17	6635	31	297754	76

Sources: Tolba M.K. and El-Kholy O.A. (eds). 1992. The World Environment 1972-1992: Two Decades of Challenge. UNEP. Chapman and Hall, London.: 137-139.

The value of biodiversity

The costs of the biodiversity loss associated with these two processes include the loss of globally important services such as carbon fixation and sequestration, together with the loss of genetic information. There are few estimates of value of these costs but all indicate that the sums involved are not trivial (Heywood, 1995; Pearce and Moran, 1994). The point has already been made, however, that these processes also leads to changes in ecological services that are of primarily local importance. These include watershed protection and the derivative services of flood control, water supply and storage already mentioned. But they also include amelioration of microclimate, soil conservation, nutrient cycling, as well as timber and non-timber forest products. These services have both current use value, and option value-the potential value of such services in the future.

It has been recognised for some time that the local value of such services is relevant to the analysis of investments in natural resource-based sectors. Anderson (1987), for example, argued that inclusion of indirect environmental benefits substantially improved the economic rate of return on forestry investments. Illustratively, an estimate of the indirect benefits of forest conservation in Korup National Park, Cameroon, (Ruitenbeek, 1989) found the net benefits of watershed protection, flood control and soil fertility maintenance to be roughly comparable to the forgone benefits from timber production. Estimates of the net benefits of habitat conservation in Costa Rica suggest a range between $102 and $214 per hectare per year (in 1995 dollars), indicating a net present value for investment in conservation of between $1,278 and $2,871 per hectare at an 8 per cent discount rate (Heywood, 1995).

In some cases, the indirect value of conserved tropical forests has been shown to exceed the private value of the converted land (see, for example, the valuation of the conservation of Khao Yai Park, Thailand, by Kaosard, Patmasiriwat, DeShazo and Panayotou 1994). This is true of marginal steeply sloping lands in watersheds where soil erosion is a major cost of land clearance. In many cases, however, the indirect value of ecological services from tropical forests may not dominate the private net benefits of conversion to commercial arable or livestock production, although it may be sufficient to favour investments which conserve key ecological services over investments which do not.

As a hypothetical illustration, Peters, Gentry and Mendelsohn (1989) used a productivity/earnings method to estimate the value of Peruvian forest at Mishane, Rio Nanay, under alternative uses. They obtained a net present value (in 1989 dollars) of $6,300 per hectare for non-timber forest products (fruit and latex) harvesting, $490 per hectare for sustainable timber production, $1000 for timber clear cutting, $3,184 per hectare for plantation harvesting and $2,960 for cattle ranching. In this case, non-timber forest products clearly dominate other activities. 'Sustainable forestry' is the least desirable option. But if the ecological services of forests were valued at Costa Rican levels it would be sufficient to reverse the rankings of between sustainable forestry, clear cutting, plantation harvesting and cattle ranching. The point here is that the indirect value of forest conservation is generally such that ignoring it leads to inefficient outcomes.

An example is to be found in the use made of the gum arabic tree (*Acacia senegal*) in the Sudano-Sahel region. It has a number of direct uses: the gum is widely used as an emulsifier in confectionery, beverages, photography and pharmaceuticals and the tree provides fodder for livestock, fuelwood and shade. But it also offers a number of indirect benefits. The most important of these are that its extensive lateral root system reduces soil erosion and runoff, and as a leguminous tree it fixes nitrogen which encourages grassy growth for livestock grazing.

A study of the economics of gum arabic in the Sudan (Barbier, 1992) found that because of a decline in the real producer price of gum arabic relative to other crops (sesame, groundnuts, sorghum and millet), the private profitability of gum arabic was lower than that of other crops except in the Tendelti system of the White Nile where field crop damage occurs frequently. But it also concluded that inclusion of the indirect benefits of gum arabic (control of erosion/runoff, wind breaks, dune fixation, nitrogen fixation) and the role of the gum belt in controlling desertification meant that its social profitability was much higher than its private profitability. In other words, the estimated private rate of return understated the value of *Acacia senegal* precisely because it does not take into account the indirect value of the resource in the provision of a range of ecological functions.

More recently, the link between the diversity of species and the disruption of local ecological services has been clarified. Many ecological functions are supported at any one moment in time by a relatively small number of species (Holling 1992). The removal of these species can cause a fundamental transformation of the ecosystem-whether from forest to grassland, or grassland to a shrubby semi-desert. Where a forest or savanna is transformed in this way, most of the main ecosystem functions are disturbed. Examples of such species include the grass species that maintain productivity in savannas or the insectivorous birds that mediate budworm outbreaks in the eastern boreal forest (Holling 1973, 1986; Walker and Noy Meir, 1982).

In both forests and savannas there is an element of redundancy in the system under given environmental conditions. Species redundancy, in this context, means that their removal has relatively few implications for the functioning and productivity of the system-at least under those environmental conditions (Walker, 1993). There is always a threshold of diversity below which their various functions cannot be maintained. Any self-organising living systems require a minimum diversity of species to capture solar energy and to develop the cyclic relation of fundamental compounds between producers, consumers and decomposers on which biological productivity depends. If the level of biodiversity drops below such a threshold the productivity of the system will fall. However, since not all species play a vital part at all times, it is often possible to delete some species without immediately affecting the functioning of the system.

The important point, though, is that even if the biodiversity needed to maintain ecological services under one set of environmental conditions is low relative to the actual level of biodiversity, a change in environmental conditions can alter the mix of organisms, populations and communities needed to maintain those services. That is to say, biodiversity has value both in supporting the flow of ecological services under given environmental conditions, and in assuring those services over a range of environmental conditions. It has an option or insurance value (Perrings 1995).

The level of biodiversity in an agroecosystem determines its capacity to respond to external shocks, whether market or environmental. This is measured by its resilience. From an ecological perspective, biodiversity protects ecosystem resilience by underwriting the provision of ecosystem services over a range of environmental conditions (Holling et al 1995). Certain species have greater ecological value under one state of nature than others, but species that are 'passengers' under one state of nature may have a key structuring role to play under other states of nature. The ecological impact of biodiversity loss depends on the link between the species and the functions of the system. Whether the deletion of some species affects a given function depends on the number of alternative species that can support the function if the ecosystem is perturbed (Schindler, 1990).

Take the example of livestock farming. Increasing livestock density has led to two major changes in the biodiversity of savanna rangelands. The first involves the loss of perennial grasses and their replacement by annuals, which vary far more in response to fluctuations in rainfall. The second involves the reduction in phenological diversity in the grass sward that moderates inter-annual variation in production. Rangelands subject to light grazing pressure tend to have roughly even amounts of early-, mid- and late-season growing grasses implying that there is roughly same amount of grass to respond to rainfall whenever it occurs in the season (Walker 1988). But in rangelands subject to heavy grazing pressure, the loss of early-season palatable species implies an increase in the relative amount of later growing species (Silva 1987). There is a reduction in the functional diversity of the range, and this leads to an increase in inter-annual variation in fodder production. McNaughton (1985) showed that

Serengeti grasslands in which there was greatest variation to biomass also varied least in annual production.

The insurance value of biodiversity has accordingly been argued to lie in its role in protecting ecosystem resilience (Perrings et al., 1995; Heywood, 1995). In the rangelands example, the market value of the liveweight gains secured by having a combination of grass species is a measure of the insurance value of that combination. From an economic perspective, biodiversity has insurance value because of its role in protecting the productivity of agroecosystems over a range of both environmental and market conditions.

The genetic diversity of cultivated species and environmental risks

Genetic resources are used in agriculture in two ways: selection of species for domestication and cultivation, and genetic 'improvement' of domesticated species. The vulnerability of agriculture due to the narrow genetic base of most major crop plants is a well-recognised source of risk. More than 90% of the food supply derives from a handful of grasses (wheat, rice, corn and oats), nightshades (tomato and potato), mammals (cattle, sheep and pigs), and birds (chickens and ducks). The narrow genetic base of these species is a 'cause' of disease and pest epidemics, leading to sharp variation in agricultural yields. Wild or traditional genetic stocks have value because they offer scope for breeding or engineering resistance to crop pests and pathogens. Most cultivated crop varieties and many livestock strains already contain genetic material incorporated from wild relatives, or from crops and livestock still used by traditional farmers. Indeed, it has been estimated that at least half of the increase in agricultural productivity this century is directly attributable to artificial selection, recombination and intraspecific gene transfer procedures. Examples include the following. The use of Turkish wheat to develop genetic resistance to disease in western wheat crops is valued in 1995 at US $50million per year. Ethiopian barley has been used to protect Californian barley from dwarf yellow virus, saving damage estimated at $160 million per year. Mexican beans have been used to improve resistance to the Mexican bean weavil, which destroys as much as 25% of stored beans in Africa and 15% in South America. Wild rice strains have improved protection of cultivated varieties against grassy stunt virus. The susceptibility of sugarcane to the mosaic virus was solved by introducing resistance to it from wild relatives (*Sacharum spontaneum*) (Heywood, 1995).

The value of biodiversity in this case lies in the fact that it provides the raw material for desirable genetic traits in crops. Genetic resources have been used to improve productivity, induce better resistance to pests, or to improve adaptation to harsh environments. This value depends on the technology of genetic transfer. Changes in biotechnology have altered both the use value of genetic resources, and the potential indirect consequences or external costs of their use. The most important of these changes is that biotechnology allows gene transfer both within species and between unrelated species, increasing the flexibility of genetic resources. At the same time, however, the introduction of biological control agents or genetically modified organisms may have indirect effects that give rise to external costs/benefits. The best understood indirect ecological effects are those relating to species introductions, and include impacts on the phenotypic characteristics of individuals; the genetics, abundance and dynamics of populations; the structure and functioning of communities and ecosystem processes such nutrient cycling and disturbance regimes (Parker et al, 1998).

These effects include the deletion of indigenous species through predation, browsing or competition; genetic alteration of indigenous species through hybridisation; and the alteration of ecosystem structure and function including biogeochemical, hydrological and nutrient cycles, soil erosion and other geomorphological processes. The associated external costs include the health and health control costs of the introduction of pathogens and their vectors; the productivity costs of changes in population abundance and species compositions; the cost of pesticides in agroecosystems; defensive expenditures and control costs of invasives such the water hyacinth (*Eichhornia crassipes*) (Williamson, 1996).

Potentially more important still are the effects of biotechology on the incentive to conserve wild relatives and traditional crops. Productivity increases associated with the Green Revolution have been associated with the abandonment of traditional varieties that have been bred over thousands of years. These landraces have been a major source of genetic diversity in agriculture, but many have disappeared with the Green Revolution. Not only has the total number of rice varieties planted been substantially reduced, but also the proportion of the total crop accounted for by ten or less varieties has been substantially increased. Current trends in biotechnology threaten to displace even more traditional varieties, and to increase the vulnerability of farmers who choose to adopt genetically uniform crops. There are no realistic estimates of the insurance value of the displaced landraces. This would, in any event, be location and risk-specific. But if assessed by the variance in farm incomes attributable to crop failures, it is reasonable to suppose that it is highest for farmers in low-income countries without access to alternative methods of insuring against crop failure. I turn now to the relation between biodiversity and income.

3 Biodiversity, agriculture and poverty

The Brundtland Report (WCED 1987) argued that one of the main driving forces behind the degradation of agroecosystems was poverty-induced pressure on forests and rangelands in order to meet basic needs. In recent years, however, an empirical relation has been observed between per capita income and certain indicators of environmental quality that appears to tell a different story. First applied to sulphur dioxide (by Grossman and Krueger, 1993), it has since been observed that a number of indicators of environmental quality first deteriorate and later improve as per capita incomes rise. Since this mimics the relation between income distribution and per capita income identified by Kuznets in the 1950s, it has been styled the Environmental Kuznets Curve. A Kuznets relation has been found between per capita income and, *inter alia*, emissions of sulphur dioxide, particulates and dark matter, nitrogen oxides and carbon monoxide, carbon dioxide, CFCs, various indicators of water quality including faecal coliform, biological and chemical oxygen demand and arsenic (see Barbier 1997 for a review).

In addition, a Kuznets relation has been found for at least one of the proxies used for biodiversity loss induced by agricultural growth-deforestation. Panayotou (1995), Antle and Heidebrink (1995) and Cropper and Griffith (1994) all identify an inverted-U shaped relation between deforestation rates and per capita income. This suggests that in low-income countries falling average incomes are associated with increasing habitat conservation. Interestingly, the same relation has not been found between habitat loss and more direct measures of poverty. IFAD's Integrated Poverty Index (IPI)[1], for example, has little power to

[1] The Integrated Poverty Index (for 114 developing countries) is based on Sen's composite poverty index. It is increasing in poverty. It is calculated by combining a head count index of poverty, the income gap ratio, life expectancy at birth, and the annual rate of growth of per capita GNP. The head count index is simply the percentage of the population below the poverty line. The income gap ratio is the difference between the highest per capita GNP in the sample and the per capita GNP of the country concerned, expressed as a percentage of the former. Life expectancy at birth is included as a proxy for income distribution below the poverty line. Using this measure countries are classified in three broad groups. An IPI of 40 or less indicates severe poverty; and IPI between 40 and 20 indicates moderate poverty; while an IPI of less than 20 indicates little poverty. The IPI was developed on the basis of data for a number of different years, but notionally describes the situation in 1988.

explain variation in habitat loss (Perrings, 1998). It is worth emphasising, however, that the EKC results on habitat loss are generated by single equation models based on cross-sectional data that ignore feedbacks between the economy and its environment. Furthermore, the chosen measures of environmental quality-rates of deforestation-are not only highly suspect, they are unrelated to the size of the forest stock remaining (see Stern et al 1996; Stern 1998 for an assessment of the data and estimation problems in this literature).

There are in fact two main patterns of deforestation, each involving a rather different relation between habitat conversion and rural poverty. Each has been encouraged by infrastructural development (usually roads), and by government policies on migration and settlement. In one pattern, conversion of forestland for agriculture is due to the actions of large numbers of usually landless individuals. This is the case in Indonesia, for example. In the outer islands of Sumatra, Kalimantan, Maluku and Irian Jaya the resettlement programme known as the 'transmigration', brought in some 3.7 million people from the most populated islands of Java and Bali in the 1960s, 1970s and 1980s. In most cases, migrants were settled in what were termed conversion forests. In countries such as Brazil, on the other hand, deforestation is largely due to the actions of larger scale ranchers. While forest conversion by small pioneer farmers using slash-and-burn techniques has been part of the problem, most habitat loss is due to the development of ranches in response to tax incentives and subsidies (Seroa da Motta, 1997). It is not therefore surprising that broad brush empirical studies of the EKC type should be inconclusive.

To identify the relation between poverty, population growth and environmental change implicit in the Brundtland hypothesis requires a different approach. Dasgupta (1993) argues that poverty drives fertility through the microeconomic decisions of rural households. Where conditions are such that people are locked into particular technologies, population growth then leads directly to pressure on the resource base. This may be exacerbated by the failure of both markets and policies. In Sub-Saharan Africa, for example, a combination of (a) cultural and institutional rigidities and (b) policy-induced declining real producer prices is argued to have led to a positive feedback loop between rural poverty, population growth, landlessness and pressure on soils, vegetation and water (Pearce and Warford, 1993).

The connections between income, asset holdings, population growth and resource use are clearly complex. However, certain linkages stand out. The first is that poverty constrains the information available to decision-makers. This obviously applies to education and training, but it also applies to the evaluation of new options. The poor command less information than do the rich. In the case of new crops or crop varieties in agriculture, information about environmental risks depends on local trials. But local trials are not costless-whether to private farmers or to agricultural departments. The quality of information available to farmers is sensitive to the resources available for trials. Similarly, estimation of the local impacts of other technological innovations including pesticides or biocontrol agents, requires research into the effects of such agents elsewhere. Once again, this is not costless.

A second important linkage lies in the fact that peoples' preferences are sensitive to both their income and assets. More particularly, the rate at which people discount the future costs and benefits of present actions turns out to be sensitive to their income. People in extreme poverty effectively discount the future costs of resource use at high rates. Because what matters is consumption today, they frequently neglect the future effects of their actions. Perrings (1989) showed that rates of time preference vary with the level of real income, and that people in subjective poverty who chose to dissave in order to maintain real consumption levels implicitly discount at very high levels.

Empirical studies of the links between rates of time preference, income and wealth, and investment in conservation technology confirm this. In two studies in rural India Pender and Walker (1990) and Pender (1996) found that rates of time preference to be inversely related to wealth. Holden Shiferaw and Wik (1998) investigated the relation between rates of time preference, poverty and conservation in Indonesia, Zambia and Ethiopia. They found that rates of time preference amongst rural households are generally high, and increase with poverty in both assets and income. They concluded that poverty is a disincentive to invest in environmental protection. Taken together with the problems of landlessness and lack of access to rural credit these factors explain the propensity of the poor in some markets to exploit insecure holdings of marginal land, and to respond in a frequently perverse way to economic incentives.

More recently, Perrings and Stern (1999) have sought to estimate the change in long-run productive potential of a semi-arid rangeland where there is prima facie evidence for loss of resilience. The paper argues that a change in productivity may be observable through its effects on the value of economic output, and treats environmental change as analogous to technological change. It estimates a bioeconomic rangeland model using data on the livestock sector in Botswana, and assuming a Box-Cox transformation utility function. This allows estimation of the degree of risk aversion and the farmers' implicit rate of discount. Livestock farmers in Botswana are shown to be both risk averse (the risk aversion coefficient was estimated to be –0.48) and to discount the future heavily (the discount rate was estimated to be16.5%).

4 Incentives for biodiversity conservation

An understanding of the way that rural poverty conditions farmers' responses is crucial to developing incentives to encourage the optimal level of conservation in low income countries. Current rates of biodiversity loss in agroecosystems in developing countries are inefficient. Many of the social costs of biodiversity loss or the benefits of biodiversity conservation are not taken into account by resource-users. The evidence suggests that developing countries should be investing more in conservation. More particularly, they should be developing incentives under Article 11 of the Convention on Biological Diversity to encourage resource users to pay due attention to the social costs of their activities.

The inefficiency of current levels of biodiversity loss implies market failure. In many cases this is due to agricultural policies. Examples include administered prices in agricultural markets that distort the private cost of biological resources; destumping subsidies that encourage clearance of ever more marginal land for agricultural purposes, and stumpage fees or royalties in forestry that encourage excessive harvesting rates in timber concessions (Panayotou and Ashton, 1992). Their effect is exacerbated by the fact that small farmers in most developing countries face credit constraints. Indeed, in the early part of this decade only 5 per cent of farms in Africa and 15 per cent in Asia and Latin America have access to formal credit (Hoff et al, 1993). Pearce (1999) estimates that world-wide subsidies to biodiversity-degrading activities are currently $684-808 billion, and that non-OECD country subsidies to biodiversity-degrading activities are $151 billion.

As a reaction to this, a number of environmental economists have argued the environmental benefits of liberalisation. Munasinghe and Cruz (1995), for example, have claimed that: a) removal of market price distortions such as agricultural or energy subsidies will improve both the efficiency of economic activity and reduce the impact of that activity on the environment; b) enhancing macroeconomic stability will encourage investment and persuades resource users to take a longer term view of their decisions and c) economic liberalisation will reduce poverty and hence pressure on open-access environmental resources. In addition, they argue that improving the security of land tenure by assigning private property or use rights promotes investment in land conservation and environmental stewardship.

The environmental effects of liberalisation are typically more ambiguous than this suggests. On the positive side, liberalisation may lower the costs of environmental protection, but if liberalisation stimulates demand for the products of environmentally damaging activities it will increase environmental damage (Anderson and Blackhurst, 1992). The evidence is that liberalisation has occurred alongside a marked increase in price risk. The coefficient of variation of detrended prices for the major food products rose sharply between the mid 1960s and the mid 1980s. In addition, these prices became positively correlated so reducing the value both of diversification within agriculture, and of export earning stabilisation schemes (Hazell, 1987). The 'Brundtland hypothesis' holds that countries locked in to products for which the terms of trade decline will tend to increase exports of those products just to maintain foreign exchange earnings (Pearce and Warford, 1993). Many primary producing countries have faced a secular fall in world commodity prices. The barter terms of trade of these countries have declined sharply over the last decade. However, the response has not been a reduction in primary commodity production, but an increase in the volume of exports.

The environmental implications of this are at present unclear. Most of the increase in agricultural output in Sub-Saharan Africa, to take the worst affected region, is argued to have derived from intensive rather than extensive growth. The increase in land allocation to crop production in arid and semi-arid areas is smaller than the increase in output. Similarly, the increase in land allocated to pasture is trivial compared to the increase in herd sizes. But data on agricultural inputs does not suggest that increased yield reflects a significant change in agricultural technology in the arid and semi-arid areas, with the possible exception draft power. The impression is that increased agricultural output and exports have been achieved by putting existing agroecosystems under greater pressure with no change to agricultural technology.

Indeed there seem to be five recurring effects of liberalisation and other elements of a structural adjustment package. First, the stimulation of export-oriented primary commodity production increases pressure on the resource base. Second, the real income of consumers tends to fall and there is disemployment in both the public and private sectors. This leads to a reduction in demand for domestically produced goods and services that worsens the condition of the poor. Third, the reduction in public expenditure programmes reduces the budgets of agencies protecting the environment. Fourth, the reduction of credit to small rural investors leads to lower on-farm investments and declining agricultural yields, countering efforts to stabilise the agricultural frontier (particularly in the absence of effective land tenure systems). Finally, deregulation makes it harder to correct price distortions in the forestry, agriculture and energy sectors.

Because of these effects, there is a growing argument that any adjustment package or liberalisation policy should be accompanied by a set of environmental reforms designed to minimise adverse environmental impacts. However, this is not just a matter of setting environmental user fees to match the social opportunity cost of resource use or of creating private property rights. If the aim of a strategy for the conservation of biodiversity is to preserve enough species to guarantee the resilience of local ecosystems, then the system of incentives should encourage appropriate behaviour. Holden et al (1998) conclude that poverty and liquidity constraints in low income rural economies mean that assignment of property rights in land, water or other environmental resources is not sufficient to reduce pressure on environmental resources to sustainable levels. They argue for additional interventions both to alleviate poverty and to regulate resource use.

This brings me back to the conditioning effect of income. The common intuition behind the Brundtland hypothesis is straightforward. If farmers are impoverished by the removal of subsidies and/or the imposition of charges for environmental resources, they will focus their attention on 'free' or open access resources, and their decisions will become increasingly myopic. The economic intuition is equally plain. A change in relative prices induces both substitution and income effects. For the poor, the income effects of price changes may be very strong. This can not only weaken the effectiveness of economic incentives. If income effects are strong enough it can lead to perverse results.

The study by Perrings and Stern (1999) referred to in section 3 identifies the impact of price changes on the resilience of agroecosystems working through the effect on farmers' resource allocation decisions. Changes in cattle prices affect optimal offtake. This in turn affects the growth of the herd, grazing pressure, growth of the range, and therefore the resilience function. In the case of Botswana, Perrings and Stern (1999) find that the elasticity of supply has the opposite sign to that that would be expected in commercial ranching. Raising prices results in less offtake. This is mainly driven by risk aversion. Under risk neutrality the sign of the elasticity depends on whether there are increasing or decreasing returns to the size of the herd. Under risk aversion, however, price increases have a strictly limited effect on offtake.

5 Conclusions

Biodiversity matters in agroecosystems because it determines both the actual and the potential productivity of those systems. Determining the optimal level of biodiversity is, however, a far from trivial exercise. The elimination of pests, competitors and pathogens may increase the productivity of cultivated or husbanded species, but it may also adversely affect the capacity of agroecosystems to respond to environmental stresses and shocks. Experimental studies of the functionality of plant species diversity in pastures have, for example, shown that more diverse grass communities are better able to exploit variation in plant available moisture than less diverse communities (Naeem et al. 1995, Tilman and Downing, 1994). To the extent that these experimental studies can be generalised, they imply that management options that reduce diversity will also decrease long-run productivity and resilience.

A second point is that the long-run costs loss of potential productivity involve market failure. They are external to the market. In many cases biodiversity externalities have been exacerbated by government macroeconomic and microeconomic policies, and by the tendency to use the agricultural pricing, tax and subsidy regime to meet governments' distributional objectives. The net effect of market and policy failure is that farmers may be encouraged to adopt management strategies that yield relatively high returns under average market or environmental conditions, but that are highly susceptible to a change in those conditions. Monocultures supported by heavy use of pesticides and fertilisers provide the best examples of these strategies.

The external costs of such strategies tend to be most severe where mechanisms for private and social insurance against the risks to agricultural incomes are limited. This is often the case in low-income countries. In such countries governments typically act as insurers of last resort, distributing famine relief when farm incomes fail. However, the resources available for famine relief in low-income countries are often limited. Moreover, the risks to farm incomes from market and environmental shocks alike tend to be highly correlated. In small open economies a reduction in producer prices or a drought tend to affect all farmers at the same time. This both restricts the scope for commercial insurance solutions and reduces the effectiveness of government relief.

What are the implications of all this? The first thing to note is that it is not an argument against intensification of agriculture. The alternative to intensification involves encroachment on ever more marginal land and the destruction and fragmentation of ever more scarce habitat. It is, however, an argument against intensification strategies that ignore the costs of a change in the mix of species in the system. It is also an argument for the development of private or social insurance mechanisms that address the change in risks to farm incomes associated with intensification. Given the problems confronting individual governments that act as insurers of last resort, this suggests regional social insurance structures. Finally it is an argument for the appropriate pricing of resources. In some cases farmers are offered inducements to maintain production of drought or disease resistant crops or livestock. If one crop mix is less susceptible to drought or disease than another, then its social value (in terms of averting expenditures on famine relief) should be reflected in its relative private cost. Whether this implies taxation of the high risk components or subsidy of the low risk components depends on local circumstances and the international trading regime.

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