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The Impact of Wildlife Trade on Endangered Species

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London Environmental Economics Centre

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ENVIRONMENTAL EFFECTS OF WILDLIFE TRADE ON ENDANGERED SPECIES

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Abbreviations

Names:

CITES = Convention on International Trade in Endangered Species of Wild Flora and Fauna
EEC = European Economic Community
LEEC = London Environmental Economics Centre
USA = The United States of America
WCMC = Wildlife Conservation Monitoring Centre

Units:

ha = hectare
kg = kilogramme
na = not available
tn = tonne

1. Introduction

The extinction of key wildlife[1] species and the decline of biodiversity in general has promoted interest in current levels of exploitation of endangered species and the international trade in these wild plants and animals and their products. Making reliable estimates of the number of endangered wildlife and rates of species extinction is plagued with difficulties. Although the lack of detailed ecological survey data required for such assessments prohibit precise figures, there have been some attempts to estimate the number of species lost in the past and the current number of endangered species (see Table 1.1).

Estimations of future extinction rates are based on projected rates of habitat loss and the relationship between species richness and habitat area. Using this approach, Reid and Miller (1989) estimated that if current tropical deforestation trends continue between 1990-2020 approximately 3-7% of species will be lost in Africa and Madagascar, 7-17% in Asia and Pacific, and 4-9% in Latin America and the Caribbean respectively. The limited data that are available clearly indicate that many wild plants and animals are threatened with extinction and that global biodiversity is declining.

Tropical countries are particularly important for protecting global biodiversity, because the moist forests in these regions contain more than half of all species. Table 1.2 lists the countries with the greatest numbers of species of mammals, birds and reptiles. Past destruction and degradation of wildlife habitats have reduced the remaining habitat area to 40% of the original area in west-central Africa, and to around 36% of the original habitat area in Indomalaysia (see Table 1.3). This has serious implications for the survival of many species.[2] However, the threat of species extinction is not just confined to the tropical forests of Southeast Asia, Africa and Latin America. Figure 1.1 provides an overview of the percentage of threatened mammals, reptiles, birds, amphibians, fish and invertebrates species compared to total species contained in OECD countries. Many countries, especially those in Central and Eastern Europe, record over 40% of their wildlife species under threat.

This paper focuses on the international wildlife trade and its impact on endangered species and the loss of biodiversity (Section 2). Although the trade in wild plants and animals is far from homogeneous across species it is possible to identify a few factors that characterize the wildlife trade in general:

- over-exploitation of wildlife for the international trade generally plays a minor role in species extinction relative to other factors such as habitat destruction and domestic wildlife consumption. However, trade interventions may be warranted for a *few key species* that are threatened with extinction from over-exploitation for the international market and whose extinction would incur high economic costs due to the foregone values of the resource, including biodiversity value, tourism value and non-use/preservation value;

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[1] In this paper wildlife is taken to include wild plants and animals but not commercial timber.

[2] The hypothesis that competition from a population (in this case humans) for essential resources of food and habitat leads to the extinction of other species also dependant on the essential resources is based on Darwin's notion of the survival of the fittest. This is interpreted by Hardin (1960).

- the high levels of *stress and mortality of species* captured in the wild, such as parrots, for the international trade is often unnecessary and needs to be addressed through appropriate policies;

- increasing demand for wildlife and wildlife products is linked to *rising income levels*. In particular, there is a substantial demand for wildlife species and their products emerging in many rapidly developing Asian countries, e.g. for elephant ivory and rhino horn in China, Singapore and Macau. Therefore, the pressure to manage the wildlife trade and protect wildlife is expected to continue in the future;

- publicity campaigns have been relatively successful in *changing consumer preferences* for some products derived from endangered species, e.g. cat skins and ivory. However, the success of such campaigns has been limited to the markets of Europe and the United States, and may not be so effective in the Far East;

- previous attempts to control the international wildlife trade by regulating traditional trading routes has often led to the entrance of new trading countries, highlighting the *fluidity of the wildlife trade* and the prevalence of an illegal trade. Flourishing black markets exist for trade in most highly valued, but banned wildlife species e.g. parrots, rhino horn, elephant ivory etc;

- restricting the trade in one species often leads to the expansion in trade of a *close substitute species*, e.g. banning trade in large cat skins encouraged traders to switch towards marketing small cat skins. Thus trade controls directed at exploitation of an individual wild plant or animal type needs to take into account the impact on the alternative species.

It is clear that in the cases where wildlife trade controls are warranted, they need to be carefully designed in order to be effective. A wide range of *market and policy failures* already exist in both exporting and importing countries that affect the wildlife trade and the incentives for sustainable resource management (Section 3). These need to be taken into account by any policies designed to manage the trade. International trade measures may be required to complement domestic policies aimed at improving incentives to sustainably manage wildlife resources. Section 4 critically appraises the different types of *domestic and multilateral trade policies* that have been, and are being, implemented to achieve wildlife and biodiversity objectives.

2. The Wildlife Trade

2.1 Wildlife Over-Exploitation as a Cause of Extinction

There are three major factors that contribute to the extinction of wildlife species:

- *the loss of natural habitats*, which is widely considered to be the dominant cause of species extinction;

- *the introduction of new species* into natural ecosystems - which may contribute to extinctions through the impact of the new plant or animal competing with or preying on native species, transmitting new pests and diseases for which limited resistance exists in native species, or destroying and degrading the native species habitat; and

- *over-exploitation of species* - species may be exploited in order to supply the domestic and/or international market. However, exploitation of wildlife for the trade is not in itself a cause of concern. Problems arise when the trade produces economic incentives that lead to harvesting levels in excess of the regenerative capacity of wildlife species. If the level of off-take continues to exceed the incremental growth of the population over time then harvesting is unsustainable and depletes the stock of the wildlife population. If these conditions persist, the over-exploitation of a wildlife species to supply the trade may drive the species to extinction.

Table 2.1 shows estimates of the impacts of habitat loss, over-exploitation and other factors on species that have already become extinct and currently threaten species. Over-exploitation was the primary cause of extinction of 23% of all mammal, 32% of all reptile, 11% of all bird and 4% of all fish species lost in the past (Table 2.1a).[3] Over-exploitation continues to pose a significant threat to many wildlife species, especially mammals and reptiles (Table 2.1b). However, in most cases the cause of extinction of wildlife cannot simply be attributed to a single factor, but is due to a complex interaction of several factors over a period of time. For example, although destruction or disruption of a species habitat may threaten the continued existence of 68% of all endangered mammal species, this threat may be further exacerbated by over-exploitation, which threatens 54% of all endangered mammals species. The combined effect of habitat loss and over-exploitation may drive some mammal species to extinction.

2.2 Domestic and International Wildlife Trade Flows

Wild plants and animals are harvested to yield important products (e.g. meat, skins, tusks, horns, feathers, etc.) or captured live for pets, for zoos and research activities. An additional, albeit much smaller, demand for wild animals is for sport hunting. Wildlife resources are exploited for:

- subsistence use;

- domestic commercial use (domestic trade); and

- commercial export (international trade).

It is extremely difficult to estimate levels of domestic wildlife use, as there are few records of subsistence use and domestic trade. The studies that do exist are usually site specific, and cannot be aggregated up to country level. However, the case studies do indicate that the

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[3] The data on the causes of extinction need to be revised, in particular the percentage of species extinct primarily due to species introduction needs to be revised upward. However, no statistics have yet been published containing this information (Luxmoore, WCMC, pers.comm.)

domestic demand for a wide range of wildlife is a major cause of species exploitation. The extent to which the in-country use of these species impacts on the wildlife populations and poses a threat to the continued existence of the species is generally not well-documented.[4]

The exploitation of wildlife for the international trade is much less important than the domestic trade for the vast majority of wildlife species, and consumes only a small fraction of the total species taken from the wild. The international trade is comprised of a diverse range of species, including around 40,000 primates, ivory from at least 90,000 African elephants, over 1 million orchids, 4 million live birds, 10 million reptile skins, 15 million fur pelts, over 350,000 tropical fish and a wide range of other species and products. However, although thousands of wildlife species are involved in the international trade each year, the majority of the trade is concentrated on a few species, for example 6 species make up 75% of reptile leather trade and less than 25 species make up 85% of tropical fish trade (Fitzgerald 1989). In the late 1980s, the annual international trade in exotic wildlife was worth well over US$5 billion wholesale, with a likely retail value closer to US$50 billion (Trexler 1989). Although the total value of the international wildlife trade is small relative to other international trade flows, it may be significant for a few species and regions.

A full assessment of the impacts of the international trade on the traded species and wildlife in general would require a detailed knowledge of (Joint Nature Conservation Committee 1991):

- the abundance and robustness of the traded species both locally and globally. This is affected by a whole range of factors, including the birth and death rates of the species, immigration and emigration, food supply, parasites and predators, habitat loss, seasonal variations in abundance and territorial and other social behaviour affecting abundance;

- the ecological significance of the traded species and its effect on other species;

- levels of exploitation and its composition in terms of age, sex and seasonal variation; and

- the effect of harvesting techniques on the traded species and other wildlife populations and their habitat.

There is clearly insufficient evidence to satisfy these requirements and fully determine the impact of the international wildlife trade on the majority, if not all, wildlife species. One source of data on the international trade in wildlife is customs import and export statistics, although they rarely give detailed statistics on the trade in wildlife. This is partly because the volume and value of the trade is low compared to other internationally traded goods and services, but also because it is often extremely difficult to categorise many of the transactions of individual species and the products of species.[5] Where it is possible to obtain trade data

[4] However, a recent book edited by Robinson and Redford (1991) does provide a wide collection of examples of domestic wildlife use.

[5] See Appendix 3.1, in Barbier *et al.* 1990.

from customs statistics, the information is often limited to the volume of trade (e.g. number of birds, cat skins and live primates) with little data on the value of the trade. Few attempts have been made to extract information from import and export statistics to assess the international trade in wildlife species.

Another source of information on the international wildlife trade is based on data submitted to the Convention on Trade in Endangered Species of Wild Fauna and Flora (CITES - see Section 4) compiled by the Wildlife Conservation Monitoring Centre (WCMC). Table 2.2 provides details on the volume of imports and exports of key wildlife resources - live primates, cat skins, raw ivory, live parrots and reptile skins based on CITES information. It is possible to identify a few dominant trading patterns based on the data presented in Table 2.2 and Figure 2.1. Africa and South America are net exporters of all wildlife commodity groups and are the major suppliers of wildlife species and products to the international trade. North and Central America and Europe are the major wildlife consuming regions, with net imports of all commodities (with the exception of raw ivory in North and Central America). Asia also plays an important role in the international wildlife trade. It is a net exporter of all commodity groups except raw ivory and is also a significant consumer of most wildlife species and products, especially reptile skins and raw ivory.

The total trade volumes may over-estimate the actual number of species supplied to the international market as the figures are not adjusted for re-exports (i.e. double-counting may be present). However, the actual impact of the international trade on wildlife populations may be under-represented by the data because of additional mortalities (of dependants, during capture and transportation), illegal trade and trade between countries not members of CITES.

2.3 International Wildlife Trade: Case Studies

The international trade in four wildlife species and their products (parrots, wild cat skins, rhino horn, elephant ivory) is examined below, outlining the economic factors characterizing the trade, the implications of the trade for wildlife populations and policies designed to regulate the trade. These species have been selected because the trade is known to have deleterious effects on the wildlife populations.

Parrots

Whilst it is acknowledged that habitat destruction has historically posed the major threat to parrot populations (and continues to be a problem), the trade in parrots is increasingly significant and may be the primary factor threatening the survival of some species today (Thomsen and Brautigam 1991). A large domestic demand for parrots has existed for centuries, with parrots sought as pets and for subsistence purposes (e.g. meat and feathers for traditional rituals and ornamental purposes). In contrast, a strong international interest in caged birds has only emerged over the past thirty years.

The change in the international demand for parrots is reflected in an increase in the world trade in parrots from over 360,000 in 1980 to nearly 620,000 in 1986 (Table 4.1). This figure is based on CITES data and is expected to underestimate greatly the actual level of world trade due to substantial parrot smuggling. For example, in order to avoid trade restrictions, approximately 150,000 parrots are smuggled across the Mexican boarder into the United States each year (Thomsen and Hemley 1987).

In addition, the numbers of parrots recorded in the international trade is unlikely to represent the true impact of the trade on parrot populations due to the high levels of mortality of wild birds during trade transactions. Table 2.4 indicates the causes and extent of mortality of *psttacines* (the *psttacine* family includes parrots, macaws, cockatoos, conures and parakeets) at each stage during the legal trade process in Mexico. These mortality rates vary with the species, the age and health of the bird and the length of time spent at each stage of the trade process. The highest mortality rate, approximately 30%, occurs during the confinement of the birds by the trapper before being shipped to Mexico city. A great concern is that the loss of birds through the illegal trade is likely to be as much as 50% greater than in the legal trade (Ingio-Elias and Ramos 1991).

Around half of the recorded international trade in live parrots is destined for the pet market in the USA, which imported a minimum of 703,000 neotropical parrots between 1981-85. The major supplies of parrots to the USA come from Latin America, in particular Argentina which exported more than 660,000 parrots over the 1982-86 period (Thomsen and Brautigam 1991). The European markets draws most of its supply of parrots from Africa, whilst Japan imports parrots from Asia and Africa (Fitzgerald 1989).

The financial incentive to trap and trade in parrots is high. Trappers receive an estimated minimum gross income of US$25 per parrot, the middlemen US$84 per parrot and the retailer US$1,200 per parrot (Thomsen and Brautigam 1991). The trapper only receives around 2% of the final value of the bird. However, given that the costs of trapping parrots are low, there are still lucrative profits to be made. Thus, the parrots are generally considered to be an important source of income, but are rarely treated as a valuable asset worth conserving. The bulk of the profits from the parrot trade usually accrue to a small group of traders based in both the exporting and importing commercial centres. Whilst the value of the parrot trade is important for some individuals, the value of the trade to the majority of exporting countries is often relatively insignificant.

Some exporting countries operate *permit systems* to regulate the trapping and sale of birds. However, the cost of the permits are often insignificant relative to the revenues derived from the parrot trade. For example, in Mexico, the cost of a permit to capture song birds and psittacines in 1987/88 was US$4, and the cost for a commercial vendor permit to sell birds was US$66 (Ingio-Elias and Ramos 1991). In addition, because the cost of the permit does not relate to the number of birds captured it has little effect on number of parrots traded.

The environmental costs of supplying parrots to the international trade may include the threat of extinction to some parrot species, a loss in wildlife biodiversity, a reduction in the supply of parrots for the domestic market and an impact on bird watching and tourism. The extinction of parrot species may also lead to the loss of non-use/preservation values, such as existence and bequest values. In addition, one often unaccounted cost of the trade in wild birds is the importation of diseases into the consuming country. For example, a bird smuggled from Mexico into the United States (the Mexican yellow-headed amazon) is believed to have introduced a highly damaging disease to poultry in California and Arizona. Approximately twelve million poultry birds were subsequently destroyed at a cost of US$28 million, and over 110 million doses of vaccine were used to eradicate the disease over a three year period (Ingio-Elias and Ramos 1991). In this case the disease was confined to poultry birds; however, it is possible that other diseases could spread to wild birds, other animals and even humans.

Attempts to control the parrot trade through piecemeal trade bans have not been successful in reducing the demand for parrots. The policies have simply resulted in changes in the source and composition of species involved in the trade. In addition, the illegal trade in parrots has emerged as a serious problem. Although the laws prohibiting illegal parrot trading are being more heavily enforced, parrot smuggling remains an extensive and lucrative business.

Wild Cat Skins

In contrast to the expanding parrot trade, the trade in wild cat skins has been in decline throughout the 1980s. Total gross exports of wild cat skins has dropped from 380,210 in 1981 to 192,402 in 1986 (see Table 2.5), reflecting changing market conditions - new trade controls, changes in consumer tastes, and the increasing difficulty of hunting prized species due to dwindling cat populations. Although there are no reliable figures on the financial returns to trading in cat skins, it is generally considered to be a financially rewarding activity. The majority of the fur trade supplies are used in the fashion clothes and accessory industry of the European markets (especially Germany and France), although Japan and Hong Kong are also important markets.

Following recognition of the threat of extinction of many big cats (e.g. tiger, leopard, jaguar and cheetah) in the late 1960s, a mixture of CITES and national trade controls were implemented in the 1970s to ban the trade in big cat skins. Although these policies were relatively successful in reducing the pressure on the big cats, the demand for small cat skins increased. The composition of the cat skin trade shifted away from being dominated by big cat skins and moved towards the smaller cats (e.g. bobcat, lynx, ocelot, little spotted cat and maragay) (Fitzgerald 1989).

Trade regulations imposed on the supply of small cat skins have been implemented relatively successfully in North America, but in Latin America illegal cat trade has emerged as a serious problem. However, publicity campaigns to inform the public of the plight of cat populations and the implications of purchasing cat skin products have been relatively effective in changing consumer preferences and reducing the demand for these products. For example, in the fur coat industry, following strong publicity campaigns, consumers have generally moved away from demanding 'real' cat skin coats, and have switched towards synthetic substitutes. This suggests that while the use values of wild cats are high, the non-use/preservation value of these species are also significant (Section 3).

Rhinoceros Horn

While the loss of habitat poses a threat to the survival of the rhinoceros population, it is generally considered secondary to the impact of the wildlife trade. Trade in rhinoceros horn has had a devastating impact on the rhino population globally, being largely responsible for the reduction from 73,000 rhinos in the late 1970s to less than 11,000 in the 1990s. In particular, the black rhino population has been reduced from over 65,000 in 1970 to less than 4,000 today. North Yeman has been a major consumer of the valuable rhino horn for dagger handles, with the remaining horn supplying the traditional medicine markets of the Far East (e.g. China, Hong Kong, Macao, S. Korea, Taiwan, Thailand) (Fitzgerald 1989).

Rhino horn is highly valued and fetches a high price. For example, African rhino horn in the markets of the Far East is worth approximately US$450/pound. Based on an average horn weight of 3.5 pounds an African rhino horn is worth around US$1,500. However, Asian horn is even more highly valued, and wholesale prices reach over US$4,000/pound in some markets. This gives an Asian rhino horn a value of around US$14,000. The rhino horn trade throughout the world may be worth as much as US$3 million annually (Fitzgerald 1989).

All rhinos have been on Appendix I of CITES since 1977, and a range of national laws have been implemented in the principle consuming states throughout the 1980s (see Table 2.6). Although trade in rhino horn slowed in the 1980s through a combination of CITES restrictions and national trade bans, the illegal international market in rhino horn continues to be a serious threat to the survival of the species (Milliken *et al* 1991). There have been recent reports that the price of rhino horn has been rising in some regions, which will increase incentives to trade illegally in rhino horn, regardless of the fragility of the population. The environmental costs of the loss os these species are likely to be high, and include a wide range of direct/indirect use values and non-use/preservation values.

<u>Elephant Ivory</u>

During the 1980s the population of elephants in Africa halved in eight years from 1.2 million to just over 600,000. In only a few countries - South Africa, Botswana, Zimbabwe and Namibia - are populations stabilizing or rising. The dramatic decline in elephant numbers in most of Africa has been largely attributed to the illegal harvesting (poaching) of elephants for their ivory. Total exports of unworked ivory leaving Africa between 1979 and 1988 amounted to over 7,500 tonnes. The annual volume of raw ivory exports fluctuated around 900,000 kg in the early 1980s, peaking at 1,031,934 kg in 1983, but then declined sharply thereafter (Table 2.7).

Although the decline in tonnage since the mid 1980s looks encouraging, the statistics hide the real impact of the trade on the elephant populations. In 1979 a tonne of ivory represented approximately 54 dead elephants. These were mainly the bull elephants, valued for their bigger tusks, with an average tusk weight per elephant of 9.3 kg. By 1987 most of the mature bull elephants had been shot, leaving the cows and calves to support the demand for ivory. They have a much lower average tusk size of 4.7 kg, such that one tonne of ivory directly represents about 113 dead elephants. In addition, the high female ratio of the harvested population leads to the death of a further 55 calves who are orphaned or die of starvation. As a consequence, almost the same number of elephants were harvested in 1987 as in 1979 to support the demand for ivory, but from a much reduced and more fragile population.

The price of raw ivory has increased from around US$60 per kilogram in 1979 to US$150/kg in 1988. In early 1989 - in anticipation of an international ban on the ivory trade - the price rose to US$300/kg. The annual value of ivory exports from Africa in the mid 1980s is estimated to be around US$50-60 million. As this figure includes c.i.f. costs and marketing margins in international trade, an estimate of US$35-45 million is probably a more realistic measure of the value of ivory exports accruing to Africa. This amount is a tiny fraction of all African export earnings, and although significant for a few individual countries, the loss of these ivory revenues would not seriously impair African development. In addition, the other non-consumptive values of the African elephant, such as its importance to tourism in some countries may be considerably more significant in terms of foreign exchange earnings.

The flow of ivory in the international trade has traditionally been towards the *hard currency consumers* - the USA, Europe and, more recently, Japan. These three consumers have acquired about 75% of all ivory worked in Hong Kong and Japan (who work approximately 70% of all ivory) this decade. The remainder of the ivory in international trade has gone to the carving industries within various developing countries - for example, India and China - and then re-exported or sold to foreign tourists. The same is true for much African worked ivory. Therefore, it is reasonable to assume that nearly all the ivory in the international trade has ultimately found its way to one of the three hard currency consumers: the United States, Europe and Japan (Barbier *et al.* 1990).

An economic analysis of the demand for raw ivory in Japan and Hong Kong indicated high income elasticities of demand for this product, that is, ivory is considered a luxury good and as real incomes grow the consumption of ivory grows even faster (Barbier *et al.* 1990, ch.4). There appears to be a latent demand for elephant ivory in a number of other Asian states - such as Korea and Taiwan, where demand has increased by 1,000 per cent in the past decade. This demand pressure will remain high so long as rapid economic growth continues in this region. Other rapidly developing countries, such as Singapore, Dubai, and Macau, have been playing an increasingly important role in the movement of raw ivory stocks in their processing and carving. This fluidity is one of the characteristics of both the legal and illegal trade: whenever one channel through which ivory is passing has been blocked by trade controls, another has opened up almost immediately.

The high economic costs associated with the loss of the African elephant - including foregone direct use values (e.g. tourism), indirect use values (e.g. ecological values and biodiversity) and non-use preservation (see Section 3) - led to international pressure for mechanisms to control the ivory trade. At the Seventh Conference to the Parties to CITES in 1989 the African elephant was transferred from Appendix II listing to Appendix I listing. This effectively banned all trade in elephant products, although in future individual countries with healthy elephant populations, effective elephant conservation and management programme and effective ivory trade controls will be able to apply to a 'technical committee' (yet to be established) to have their elephant populations transferred back to Appendix II listing (see Section 4). It was also agreed that the international trade ban should apply to all existing ivory stocks, with no special exemptions for the large ivory stockpiles - in particular in Hong Kong and Burundi.

The CITES ban received support from the majority of African nations and consuming countries, some of whom had already banned ivory trading domestically - for example, Kenya, the USA and Europe. However, some elephant range states and ivory consuming countries voiced dissatisfaction at the proposal. Zimbabwe, Botswana, South Africa, Malawi and Zambia (some of whom have healthy elephant populations) refused to support the CITES resolution to move the African elephant from Appendix II to Appendix I. Of the major consuming countries, China and Hong Kong have also declared reservations against the ban and have continued trading in elephant ivory. The fact that not all countries have agreed to the ban leaves the elephant populations vulnerable to ongoing illegal trade. Six southern African countries (South Africa, Zimbabwe, Botswana, Malawi, Zambia and Namibia) are seeking a downlisting of their elephant populations from Appendix I to Appendix II at the next Conference of the Parties to CITES in March 1992. Acceptance of their proposals would mean a resumption of the international trade in ivory just two years after the initial implementation of the ban.

3. Market and Policy Failures Affecting the Wildlife Trade

Individuals harvest and trade in wildlife species in order to make money. So long as it remains in the direct economic interest of an individual to harvest and trade in wildlife, s/he will continue to do so. By investing time and effort in harvesting wild resources, individuals are essentially comparing the current returns from this activity to alternative sources of income. Individuals will continue to harvest and trade endangered species so long as they perceive that the relative returns to this activity remain high. Unfortunately, whilst wildlife resources may be regarded as an important source of *income* they are not necessarily treated as a valuable *asset* to be sustainably managed and maintained (Burgess 1991).

If wildlife species are considered to be a valuable asset then it may be in the interest of the *individual to maintain these resources into the future as they increase in value over time.* The decision to sustain wildlife resources depends on the relative returns from these assets compared to the returns from other assets available to the individual in the economy (reflected in the real interest, or discount, rate) and the individual's preference for receiving the returns today rather than in the future (reflected by the individual's time preference).

The relationship between the price of the traded wildlife resource, the costs of harvesting and the return on comparable investments (the discount rate) determines the rate at which individuals decide to use the resource. However, from the standpoint of the individual exploiter, it may be optimal to harvest a species to extinction, if there is a combination of:

- a high price of the resource relative to the cost of harvesting; and
- a high discount rate by users relative to the species growth rate.[6]

Given that the costs of harvesting plants and animals from the wild are often extremely low, in comparison to the price of the traded resource, lucrative profits can be derived from exploiting the resource. The price of the wildlife species or product may reflect the scarcity value of the species; for example, in the case of rhino horn, elephant ivory and cat skins, consumers are willing to pay a 'scarcity premium' for goods they perceive to be unique or

[6] The economics of an 'optimal extinction' scenario is discussed in a bio-economic model of elephant decline, which is applicable to other wildlife species, presented in Appendix 1.1 of Barbier *et al.* (1990).

rare. High economic rents (the total value of selling a commodity minus the costs of supplying it) create an incentive for individuals to harvest endangered species. The existence of economic rents alone does not necessarily imply over-exploitation of wildlife species. However, if this is combined with a net effective discount rate (actual discount rate adjusted for any real price increase) that exceeds the growth rate of the harvested population then it will be in the interest of the individual to deplete the resource as quickly as possible, even to extinction, and in order to reinvest the profits in a higher yielding asset.

The growth rate of a species population is chiefly determined by the species' biological reproductive capacity. Although all wildlife species are potentially renewable resources, some are much less capable of responding favourably to harvesting pressures than others. Of the extensive list of species threatened principally by trade, a great proportion are slow-maturing and produce only a few offspring per year, e.g. many primates, elephants, rhinos, larger cats, and other fur-bearing carnivores.[7]

The low reproductive rate of a parrot species, the hyacinth macaw, typifies the problem facing many wild species. Of the 500 hyacinth macaws in Pantanal in Brazil, only 15% to 30% of the adult population attempt to breed in any one year. What is more, not all breeding pairs of macaws are successful in fledging young and those that are successful rarely fledge more than one offspring. Thus, 100 mated pairs of breeding macaws may produce between seven and twenty-five young each year. This is a very low reproductive rate, and given the size of the current hyacinth macaw population in this region, it is doubtful whether any substantial, long-term harvesting of the population is viable (Thomsen and Brautigam 1991).

There are two reasons why the harvester may face a high private discount rate. First, if the harvester is located in a rural area of a developing country (the source of the majority of traded endangered species) then capital is likely to be scarce. Where it is available, the cost may be extremely high, with informal lending rates typically around 50-100 per cent. In addition, the harvester is likely to prefer to receive income today rather than waiting until the future (i.e. high rate of time preference) because of uncertainty over the future, poverty and risk of death. The prevalence of high economic rents and the combination of low species growth rates and high discount rates characterizes the situation facing many threatened and endangered species today.

The extinction of species is not considered to be an economic problem if the level of exploitation is 'socially optimum'. However, there are several domestic market and policy failures that may distort the incentives for conserving wildlife and drive a wedge between the private and socially optimum rates of species exploitation. *Market failures* exist when markets fail to reflect fully environmental values. The presence of open access resource exploitation, public environmental goods, externalities, incomplete information and markets, and imperfect competition all contribute to market failure. *Policy failure* occurs when the public policies required to correct for market failures over- or under-correct for the problem. They also occur when government decisions or policies - in areas where there are no market failures - are themselves responsible for excessive exploitation of endangered species. The

[7] See Chapter 8 on international trade and endangered species in Oldfield (1989).

result of market and policy failures is a distortion of economic incentives. That is, the private costs of exploiting wildlife do not reflect the full social costs of the damage resulting from the extinction of, or increasing threats to, wildlife populations.

3.1 Market Failures

Externalities

An *externality* is said to arise when the market price of a resource does not reflect the full costs of resource use. Although the price of a wildlife resource may reflect the direct costs of harvesting the resource, it is unlikely to reflect the *wider social costs* of exploiting the wild plant or animal. These wider social values may include (Barbier 1989):

- use values, which comprise direct and indirect use values;
- option (and bequest) values; and
- non-use/preservation values.

The African elephant is a good example of a wildlife species which is exploited to supply a product to the international market, in this case ivory, without taking into account the wider costs of elephant exploitation. Some of these lost values may be significant and reflect a large 'willingness-to-pay' for conservation. For example, an alternative non-consumptive use value of elephants is tourism. A recent study by Brown and Henry (1990) estimated the viewing value of elephants by tourists in Kenya, which is visited by approximately 250,000 to 300,000 foreign adult tourists annually. Based on survey responses filled in by safari tourists and tour operators in Kenya, the non-transportation cost of safaris averaged US$1,400. This represents potential expenditures of around US$375 million, of which about US$200 million is spent in Kenya. The results of the travel cost and contingent valuation techniques are reasonably comparable, and suggest that the value of viewing elephants in Kenya is US$25 million per year.

Given that special interest tourism is estimated to be growing at 10-15% each year, the future value of viewing elephants is also anticipated to increase. However, respondents to the survey reported that if the elephant population in Kenya was to decline by 50% (or decline by 25%), at least 50% (or 31%) would no longer find Kenya an attractive destination for themselves, their family or to recommend to friends. In contrast, tour operators believed that there would only be a decline of 10% in safari activity if the elephant population decreases by 50% in the next decade. Although expectations of the impact of declines in the number of elephants on tourism differ between tourists and tour operators, it is clear that the loss of elephants will directly lead to a significant reduction in tourist visits and the revenue derived from tourism.

The elephant also has an important *indirect use value* which is derived from its natural ecological functions and role as a 'keystone species' (Western, 1989). The elephant plays a major part in maintaining biodiversity by diversifying savanna and forest ecosystems, acting as seed disperser, reducing bush-lands, expanding grasslands and so on. The ecological benefits of elephants are dependant on their population density being neither too low nor too high. For example, in the protected areas of Amboseli National Park in Kenya where elephants crowd in, or in the areas of non-protected lands that are abandoned, wildlife impoverishment results. The most equitable mix, and high relative abundance, of species occurs at the park boundaries where elephant densities are moderate.

Additional benefits derived from conserving elephants include *option values*. Option values are related to use values, in that they arise from individuals valuing the option of using the elephants in the future. Thus there is an additional 'premium' placed on preserving elephants for future use, particularly if one is uncertain about this future value but believe it may be high and if current rates of exploitation are expected to result in the irreversible loss of the species. *Bequest values* are similar to option values, in that individuals may place a high value on the conservation of elephants for future generations to use. For example, some individuals wish elephants to be preserved in order to ensure that future progeny will have the opportunity for 'elephant viewing'. Furthermore, there are individuals who do not wish to make use of elephants but derive satisfaction from knowing that the elephants will be preserved. That is, they place an *existence value* on the African elephant. Given the widespread public concern over the plight of the African elephants, it is reasonable to suggest that these non-use and preservation values are highly significant.

Although it is extremely difficult to place a monetary value on the wider social values of the African elephant, it is important to recognise that these values exist, are often substantial and need to be taken into account. For example, the study by Brown and Henry (1990) shows that the non-consumptive direct use value of elephants from tourism in Kenya may be as much as ten times the domestic value of harvesting elephants for ivory. The failure of the market to reflect these lost values provides an economic rationale for intervention in the elephant ivory trade and investments in elephant conservation.

Open Access Exploitation

Although many wildlife resources have traditionally been sustainably managed through common property regimes, these may be undermined by increasing populations, policy failures and the expropriation of resource ownership and rights, for example through state establishment of 'protected areas'. The break down of these systems can lead to *open access exploitation*, where it is difficult to exclude individuals from using the resources. This creates incentives for an individual to harvest the wild resources now, rather than wait until the future, because s/he cannot prevent other individuals from exploiting the resource in the meantime. Each individual will therefore ignore any *user costs* of plant and animal exploitation, encouraging over-use of the wild resources.

For example, the nests of certain species of swiftlets have been harvested for hundreds of years. The nests, which contain salvia, have been esteemed by the Chinese as a food tonic, although there is little scientific evidence to support this. The major producer countries include China, Indonesia, Malaysia, Singapore, Thailand, Vietnam and Burma, and the major importing country is Hong Kong (net imports of around 140,000 kg of nests at a total value of around HK$350 million in 1990) (Lau and Melville 1991). In the past, harvesting have been restricted to certain seasons - for example, in Niah Cave in Sarawak (one of the largest swiftlet nest colony's in the world) the nests were collected twice a year in December and June, with the first and second nest taken, but the third nest left so the birds could rear their chicks. However, in recent years, harvesting pressure in many areas has increased and the traditional harvesting cycles are no longer being implemented. Harvests are often continuous throughout the year, with eggs and chicks being destroyed at the time of collection. This has resulted in the marked decline in some populations of swiftlets and the extinction of others. In April 1989, the Sarawak government announced a three-year ban on nest collecting at Niah Cave, in an attempt to let populations recover.

Imperfect Information

Although there are often high economic rents to be made from wildlife exploitation, these rents are often dispersed across a wide range of individuals, including harvesters, local traders, domestic officials, foreign traders etc. The individual who actually makes the harvesting decision typically receives a relatively insignificant proportion of the additional revenue. The price that the harvester receives for exploiting the wildlife resource is not a reliable indicator of the true value of the resource. Thus, the harvester has *imperfect information* upon which to base his consumption/investment decision.

In addition, the risks and high levels of uncertainty confronting private individuals translate into *high effective rate of time preference*, causing individuals to discount potential returns from conservation investments at a high rate. However, *societies as a whole are risk averse* and must balance immediate returns from resource exploitation against ensuing losses of future productivity and other environmental values. Where resources are *without viable substitutes* are being *irreversibly lost*, or when the future value of the resource is uncertain, society will prefer less consumption of, and greater investment in, conservation than the private decision maker.

3.2 Policy Failure

There are broadly three ways in which policy failure may contribute to the threat of extinction faced by many wildlife species, through:

- inadequate protection of habitat;
- ineffective domestic management regimes;
- inappropriate and/or insufficient international wildlife trade controls.

Habitat Protection

There have been attempts to reduce the threat of extinction by protecting the habitat of species from further destruction and degradation, and the number of protected areas has been rising steadily throughout the world (see Table 3.1). However, although the number and total area of protected areas has been increasing they still only represent a small percentage of total wildlife habitat. In species rich tropical Africa and the Indo-Malayan region around 10-15% of existing wildlife habitat is protected (World Resources Institute 1991).

While conservationists may appreciate the need to protect species habitat in order to conserve wildlife populations, the complexity of having to deal in numerous national political arenas and intervene in national land use and wildlife policies may severely constrain this policy option. Furthermore, habitat protection may incur high costs, which may make this approach unattractive unless the producer countries are fully compensated for their conservation efforts. Finally, the tendency to focus on the conservation of specific species has made it difficult to direct funds towards widespread habitat protection. However, recent concern over a wider range of species, and biodiversity in general, has led to a change in attitude towards wildlife habitat preservation.

Unfortunately, attempts in the past to establish protected areas have not necessarily benefited the conservation of wildlife species. One reason is that the protection of habitat and wildlife has frequently worked against the direct economic interests of local communities and their incentives to conserve wildlife. As noted by Barbier (1992), the 'preservation' of areas has often led to the displacement or enforced relocation of rural communities. Although subsequent use of the land and the wild resources of these areas is prohibited, local communities have received very little real compensation for the loss of traditional livelihoods and resources. In addition, local communities often suffer economic costs from crop loss and other damages resulting from marauding wildlife or wildlife migration. The 'protection' of wild areas has meant that local communities have lost their traditional management and use rights of local wild resources, and perceive little incentive, and often a major cost, to conserving these resources (Swanson and Barbier 1992). What is more, protecting habitat may be necessary, but not sufficient, to sustain wildlife populations if harvesting levels continue to exceed the regenerative capacity of wildlife populations.

Domestic Wildlife Management Regimes

Threats to wildlife from loss of habitat and unregulated harvests appear to leave little room for sustained, regulated harvest. Indeed, many governments view wildlife management primarily as matter of protecting wildlife species from direct harvesting. However, past experience with domestic bans on harvesting have generally proved to be unsuccessful in controlling wildlife exploitation. The regulatory approach is often undermined by inadequate levels of enforcement and ineffective penalties for offenders. However, Kenya's recent aggressive to 'shoot-on-sight' policy for controlling elephant poaching appears to have been effective in reducing elephant poaching. Similarly, increased anti-poaching and enforcement efforts throughout much of east, central and west Africa have resulted in a substantial decline in elephant poaching in this region. In contrast, poaching in Zimbabwe, South Africa and Malawi is reported to have flared up again, while elephant poaching continues to remain intense in Zambia (EIA 1992). Wildlife management programmes receive greater support from local communities when there are tangible incentives for participation. For example, wildlife management based on systems of multiple use - sustainable harvesting of wildlife combined with sport hunting, ecotourism and wildlife ranching - may provide greater incentives for conservation.

Domestic Trade Controls

The role of domestic trade controls in supporting domestic wildlife conservation regimes may be significant for those wildlife resources that are traded on the international market. At best, domestic trade policies many complement domestic wildlife management programmes and thus benefit endangered species and biodiversity. At worst, ineffective or inappropriate trade controls may actually work against domestic wildlife conservation objectives. Similarly, the failure to design and implement policies to control the wildlife trade may undermine attempts for domestic wildlife conservation.

4. Policy Options to Manage the Wildlife Trade

A few key points can be drawn from the discussion so far. First, over-exploitation of wildlife is but one of several factors causing the extinction of species, and it often plays a relatively minor role in species extinction relative to habitat destruction. Second, only a small percentage of total wildlife exploited goes into the international trade, with the vast majority being consumed domestically. Third, the majority of international trade in wildlife is drawn from populations that are robust and not in danger of becoming extinct. Given the limited impact of the international trade on wildlife in general, the justification for policies to intervene in the trade needs to be clarified:

- for a small number of wildlife species, over-exploitation to supply the international trade is driving the species to extinction and there are often high economic and ecological costs associated with the loss of the wildlife species and biodiversity;
- the ability to conserve wildlife species through habitat protection in producers countries is constrained. What is more, even if habitats can be preserved, exploitation of wildlife for the domestic and international trade may continue to threaten the wildlife populations and biodiversity;
- decisions over the level of species exploitation for the international trade are distorted by a mixture of domestic market and policy failures, including existing domestic trade policies.

There are broadly two approaches for regulating the trade: first, through a *command and control* approach using trade restrictions and bans, and secondly through the use of *economic instruments* in the form of import and export taxes and subsidies. The majority of wildlife trade policies implemented to date have taken the former approach, although the use of economic instruments has received more attention in recent years. The implications of these policies for the environment and the economy, both domestically and on a wider scale, are examined using case studies where possible. Although there is little detailed analysis of the environmental and economic effects of such trade policies, the major question underlying the following section is 'under what conditions are such trade interventions justified?'

4.1 Domestic Wildlife Trade Policy

For those countries who are party to CITES, domestic wildlife trade policies tend to be shaped within the framework of CITES Appendix listing (see Section 4.2). The trade regulations established by CITES do not necessarily address the specific needs of producer and consumer countries, due to regional and national differences. However, under CITES Appendix III listing countries are able to implement additional domestic trade policies. Alternatively, countries can take out reservations against CITES Appendix listing. Other countries that are not party to CITES are free to establish their own wildlife trade policy agendas. These are based on their own wildlife trade requirements and are often set in light of CITES decisions.

There are three types of domestic trade measures that can be used to regulate the wildlife trade. These categories are not completely distinct, and trade policy measures may belong in several categories at once given that there is often more than one objectives behind a single policy (Stevens 1991):

- *complementary measures* - these are import and export restrictions implemented in conjunction with domestic environment related rules and regulations;
- *coercive measures* - these are import measures designed to encourage other countries to adopt appropriate environmental practices; and
- *countervailing measures* - these are measures to counter foreign environmental practices that are directly harmful to domestic interests.

Complementary Measures

In order to uphold domestic polices controlling the exploitation of wildlife within a region, countries may decide to prohibit the importation and/or exportation of protected wildlife species. Such wildlife trade policies are *complementary* to domestic wildlife management objectives. Although information on national wildlife trade laws has been difficult to obtain in the past, it has been more readily available in recent years. An increasing number of African, Asian and Latin American countries have adopted trade controls on exports of native species to reduce the pressure on national wildlife resources and support domestic conservation programmes. Trade bans fall across a broad spectrum between a complete ban on all wildlife trade (e.g. of the Latin American countries, Brazil banned all commercial wildlife exports in 1967, Paraguay in 1975, Ecuador in 1981, Mexico in 1982 and Bolivia in 1984) to trade bans on specific species (e.g. many countries adopted domestic bans on the export of elephant ivory, including Zaire (1978), Gabon (1981), Sudan (1983), Chad (1984), Liberia (1984) and Senegal (1986)) (see Table 4.1). A range of trade controls lie between these two extremes, for example Colombia, Uruguay and Venezuela severely restrict wildlife trade, Australia bans the exports of most endangered species and the United States restricts trade in native wild birds under its Migratory Bird Treaty Act.

The Lacey Act in the United States is a somewhat unique complementary measure which supports other countries' efforts to protect their native wildlife by making it a federal crime to import animals or animal products taken or exported in violation of foreign law. The Lacey Act commits financial and human resources to enforcing the trade controls and is designed to reduce the black market in species smuggled out of other producer countries attempting to control their wildlife trade. In a similar manner the EEC banned imports of some spotted cat species from Latin America in 1987 to complement the national export restrictions adopted by many Latin American countries (Fitzgerald 1989).

There are additional complementary trade policies which are linked to the *way in which the wildlife species are produced* rather than to the actual product traded. For example, in 1987 New York State banned the importation of birds caught from the wild and only allowed captive bred birds to supply the pet trade. The objectives of this policy included maintaining domestic environmental preferences and regulations, supporting export regulations in producer countries, and encouraging 'environmentally sound' production of birds. However, this type of measure raises several problems, including the verification of production processes and labelling of species produced in an acceptable manner. A further concern is that there is often little capacity for captive breeding of birds in producer countries due to the combination of high investment costs and high risks (associated with the susceptibility of young birds to parasites and disease) attached to intensive breeding programmes. Therefore, restricting imports from the traditional producer countries may be seen as a way for the United States to create and protect the market for domestically produced captive birds. If the trade regulations are perceive to be 'protectionist' this may lead to counter trade measures by trading partners in order to protect their own economies. Increasing trade barriers are usually associated with high economic opportunity costs.

The impact of complementary, and other, trade measures on the economy and the environment depends to a large extent on the level to which these measures are *enforced*. In the majority of developing countries there is often little, if any, resources devoted to upholding such trade legislations and, as a result, they are often ineffective in implementing the controls. For example, although the export of wildlife in many Latin American countries is prohibited, the legislation is widely ignored and many wildlife species continue to be exported from this region. For example, an estimated one million caiman lizard skins are exported from South America annually and between 1980 and 1984 there were over two and a half million crocodile skins exported from South America (Redford and Robinson 1991).

When trade restrictions in producer countries are strictly upheld but surrounding producer countries are not so rigorous in their trade controls, wildlife products are often smuggled across borders between producing countries for re-exportation. An example of this occurred in Burundi, located between Zaire and Tanzania, which for many years had just one elephant. However, in 1986 Burundi exported 23,000 tn of ivory, all carefully documented as originating in the country. However, the effective control of trade of in one producer country may have a detrimental impact on wildlife in other countries as they are placed under greater pressure to satisfy the international demand. In other cases, domestic trade controls imposed on a specific species may have an adverse impact on substitute species (for example, see the trade in cat skins in Section 3). Thus, it is very difficult to predict the overall effect of complementary trade policies on domestic and wider wildlife populations. The impact of an ineffective ban on both wildlife and the economy is examined in more detail below.

If a country is effective in implementing its international wildlife controls, and these are complemented by successful policies to manage exploitation for domestic consumption and habitat protection, then the threat of extinction facing wildlife species may be reduced. This is only true if the international trade is a significant source of the threat to the species existence. Although there will be a cost to the economy in terms of the loss of potential export revenue resulting from the trade controls, this cost is unlikely to be highly significant for the majority of producer countries. For example, in Kenya the reduction in revenue from banning ivory exports may be more than offset by increases in tourist revenue (see Section 3).

The complementary domestic policies of conserving habitats and managing domestic demand for wildlife products may incur significant costs. The international wildlife trade is a *potential source of revenue* which can be tapped to support domestic sustainable wildlife management projects. For example, import taxes may be set to raise revenue to reimburse domestic conservation costs. However, to date there are few, if any, examples of this approach being carried out in practice.

Coercive Measures

Coercive trade measures may be passed in order to encourage producer countries to adopt sound environmental practices. These coercive measures are generally established in consuming countries which have no complementary domestic environmental policies as they do not posses the targeted resource within their territory. Such measures may be justified on the grounds that they are being used to maintain the global environment and to halt excessive depletion of common resources, such as endangered species and international biodiversity. Coercive trade measures may also be tied of some form of revenue raising system, such as import taxes, which can then be used to compensate the producer countries for conservation or losses in earnings. This policy option is examined in more detail below.

Numerous examples of coercive trade measures, including bans on ivory, bird and cat skin imports, adopted by many consuming countries are discussed in Section 3. There are several additional coercive trade measures that are worth noting. For example, the 1973 Endangered Species Act in the United States allows regulation of trade in species listed under CITES and some species not included under CITES which the United States believes deserve protection. The United States also has a Marine Mammal Protection Act which bans commercial imports of all marine mammals and products, including those from polar bears, whales and sea otters.

Some of the major wildlife consumers are based within the European Economic Community (EEC), and the impact of coercive wildlife trade measures adopted throughout the region can be highly significant. For example, in 1982 the EEC banned imports of whale products, effectively closing one of the biggest markets for sperm whale products. The EEC also operates a wildlife classification system which determines the trade measures imposed on the listed species:

- C1 - which encompasses CITES Appendix I species and some additional species, such as the spur-thighed tortoise and the Caribbean flamingo (Appendix II species); and

- C2 - which encompasses the remaining CITES Appendix II species.

Imports of C1 species into the Community are prohibited. In addition, selling within the EEC, offering for sale, displaying publicly for commercial purposes, or otherwise commercially using C1 species is prohibited, subject to certain exemptions which member states may grant. Imports of C2 species are allowed so long as certain conditions are satisfied. First, the importation of any species into the EEC covered by the regulations requires an import permit. Second, importers must show that the listed C2 species were collected legally and that their capture does not have a harmful effect on conservation of the species. Finally, EEC traders must obtain import permits from the intended country of import of C2 species and prove that these incoming live animals will receive adequate care (Lyster 1983).

There are certain benefits of the EEC wildlife trade regulation system. One advantage is that the regulations force EEC member countries to comply with CITES controls even though they may not be party to the convention, and through this also encourages non-members to join CITES. In addition, the EEC intends to join CITES in its own right when the CITES system allows this. This would effectively make all EEC countries members of CITES. However, the EEC approach may not go far enough because it maintains its central objective of promoting a free flow of trade between member nations. As a result, it does not restrict internal trade in C2 species, or require member states to submit reports on intra-community trade to CITES. This makes it difficult to track wildlife shipments once they have entered the EEC network, and may enable traders to smuggle illegal imports into the community through the 'weakest links' in the system (Lyster 1983).

Countervailing Measures

Countervailing wildlife trade measures may be implemented by a country to account for the costs imposed on it by another country's wildlife trade activities. An example of this would be where a country imposed a tax on its imports to reflect the domestic costs incurred through the loss of an endangered species or the decline in biodiversity in general. The loss of domestic wildlife values may include potential use values, such as tourism, and non-use values such as existence values. The revenue raised may be used to compensate for the loss of values domestically. However, there are no examples to date of countervailing measures in the wildlife trade, and it is not anticipated that such measures will be taken in the near future.

4.2 Multilateral Wildlife Trade Measures

Multilateral environmental agreements to achieve various global environmental objectives, such as the protection of endangered species and maintenance of biodiversity, may incorporate trade measures. CITES is an obvious example of a multilateral agreement that uses trade measures to meet its environmental objectives. There are numerous other multilateral treaties which employ similar principles, including the International Convention for the Regulation of Whaling (1946), the Interim Convention on Conservation of North Pacific Fur Seal (1957) and the International Agreement for the Conservation of Polar Bears (1973). There are generally few conflicts between the different international agreements, mainly because CITES tends not to affect the provisions of other treaties that relate to endangered species.

The Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES)

CITES was established in the early 1970s and provides the most important regulatory body controlling the international trade in wildlife and wildlife products. There are currently 111 countries who are members of CITES (TRAFFIC Bulletin 1991). CITES trade restrictions are based on the biological status of wildlife species and the extent to which they are threatened by trade. Member countries prohibit commercial international trade in currently endangered species that are listed under Appendix I (approximately 600 species). Species that may become endangered in the near future are listed under Appendix II (around 30,000 species). For Appendix II species, trade is limited and monitored closely - export quotas depend on the biological status of the Appendix II species population in each producer countries. Appendix III allows countries to prohibit trade in nationally protected species. However, countries are permitted to take out 'reservations' against the Appendix listing which allows them to continue trading in any species, even those threatened with extinction.

With over 111 member countries, CITES is the most widely accepted conservation agreement in the world, and unlike many other international trade agreements draws its membership from both consumer and producer countries. CITES is often portrayed as a symbol of success because the commitment of government and non-government effort and resources is unparalleled in the field of international environmental cooperation. However, when the success of the Convention is measured in terms of effectiveness in improving the status of endangered species, prevention of the endangerment of additional species and reducing the international trade as a threat to species, then the problems of using trade controls to achieve species conservation emerge (Trexler 1989). The various problems encountered by CITES can be briefly summarized as:

- inadequate biological and trade data on which to set trade standards, thus inappropriate trade restrictions may be established;

- exemptions to CITES rules were originally designed to make the system more effective in the long run, but in practice tend to expose weaknesses in the Convention. Exemptions include travelling exhibits, non-commercial trade, personal items, goods in transit, pre-Convention species, captive bred species and ranched wildlife. In addition, reservations against CITES decisions may be taken by individual countries to exempt themselves from trade restrictions. Allowing some imports and exports creates 'loopholes' which can be exploited for the illegal trade;

- not all countries belong to CITES, and non-members are under no obligation to abide by CITES rules. There may be an incentive for individual countries not to join the international agreement as they benefit from continuing to trade in wildlife whilst CITES signatories adopt trade regulations, i.e. the non-signatories act as *free-riders*. If the non-signatories play an important role in the international trade, either as producers, consumers or re-exporters, their activities may undermine the effectiveness of trade restrictions in member countries;

- there are insufficient funds to enforce and implement adequately CITES requirements. In addition, the CITES Secretariat has limited power to make member countries abide by its decisions. For example, Table 4.2a shows that the level of Party compliance in submitting annual reports has been low - throughout the 1980s, only around 55-65% of members have satisfied this requirement. In addition, the standard and reliability of the reports that are submitted is not high, with little correlation between imports and exports recorded by member countries (Table 4.2b). The inability to enforce the trade restrictions means that some signatories continue to trade in wildlife and do not abide by CITES requirements. Other countries may control the legal trade, are not able to prevent illegal trade in endangered wildlife species and products. Once the wildlife trade is driven underground it is much more difficult to monitor, may actually drive up the price and demand for some species, and removes the possibility for governments to obtain revenue from the trade.

The Impact of an Ineffective Ivory Ban

Some of the problems of the CITES system can be highlighted by examining its recent experience in managing the African elephant ivory trade. Although the African elephant population has been decreasing over the past few decades, previous attempts to increase the controls on the ivory trade have been unsuccessful. However, at the CITES conference in October 1989 the Parties were presented with statistics that showed that the African elephant population had halved in eight years from 1.2 million to just over 600,000 and projected population trends that showed if current levels of exploitation continued, African elephants could be extinct by 2010. The response, by a majority vote of 76 to 91 was to move the African elephant from Appendix II listing to Appendix I listing. This was an indefinite ban on all trade in elephant products, including ivory (Burgess 1989).

Although such a response may have been a necessary and effective short term solution to protect the African elephant population, this policy choice does not ensure the long term survival of the species. That is because it does not address the more complex issues surrounding the ivory trade, nor does it allow sufficient flexibility across African countries or create adequate incentives for conservation. In the long run, unless the ban is adequately adopted and enforced, it may have a detrimental impact on the conservation of elephants.

A simple graph illustrates this situation (Figure 4.1). The ban initially causes demand D_o to drop - as the primary consumers, the United States, European Community and Japan, leave the market. The initial impact could be a fall in price, until demand adjusts by the entrance of new consumers - such as South Korea, Taiwan and Saudi Arabia - or until the demand by the existing consumers picks up again. But while overall demand will fall, the illegal component of demand, D_i, could increase. Similarly, although the ban may cause initial supply, S_o, to decrease, a larger proportion of it will be illegal supplies, S_i. As a result, the price of ivory could end up rising, increasing the incentives of poachers and the illegal trade to supply the market. In sum, an ineffective ben might increase prices and incentives to smuggle, and it will definitely result in the loss of the capacity to monitor the trade or invest its returns in conservation programmes.

This means that an ineffective ban may be worse for the survival of the African elephant than a policy aiming to establish a controlled trade based on a sustainable yield of ivory, however small. There are several reasons for this. First, a modest production of ivory offers some incentive to consuming nations to accept a controlled legal trade and enforce it, whereas they would lose all benefits from a banned trade. Secondly, legalising trading might make it possible to set up a trade mechanism that would allow only those producer countries with healthy and sustainably managed elephant populations to sell ivory. A properly constructed trade mechanism, using economic instruments such as taxes and subsidies to manage the trade, could allow for a larger share of the ivory profits to be channelled back into the producers states to be used for sustainably managing elephant populations. mention clause in agreement to allow this.

Anderson (1991) gives an economic justification for intervention in the international ivory trade, in this case with taxes (see Figure 4.2). Assume the supply of raw ivory from African exports is given by *XS*, and the demand for raw ivory for the major consumers in East Asia is given by *XD*. At this level of demand and supply the optimum level of trade would be at the point *QP* where *XS* intersects *XD*. However, individuals in Western Europe and North America may perceive that this level of trade is adversely impacting on the African elephant population and incurring 'external' costs in the form of the loss of an endangered species and the reduction in biodiversity. The marginal external cost is given by the distance between *XS* and *XS'* in Figure 4.2.

Free trade in elephant ivory would provide the producers (Africa) with a welfare gain of *ceg*, and the consumers (East Asia) with a welfare gain of *cag*. However, the conservation group of Western Europeans and Northern Americans would suffer a cost of *cej*. This cost may be greater or less than the sum of the producer and consumer surpluses (*ceg* + *cag* = *cea*). The impact of a ban on the ivory trade, which may be in the interests of the conservation group, would incur substantial welfare losses upon the producers and consumers (i.e. the loss of *cae*). This welfare loss under a ban would encourage continued trading, to make up some lost consumer/producer surplus. The resulting level of trading would not necessarily equate with socially optimal level of trading, and there exists a high level of uncertainty over who (i.e the conservationists, producers or consumers) would gain and who would loose in this system. However, in practice, given the problems of enforcing trade bans, the conservationists will tend not to be the winners.

A globally optimal outcome could be achieved by setting taxes at a level *bd* which would reduce the trade from *0Q* to *0Q'* and equate the global marginal benefit with the global marginal cost. This is based on the assumption that taxes could be collected and redistributed costlessly and without introducing distortions such as smuggling. The impact of this tax on the African producers would be to reduce their surplus by *cdfg*, and reduce the importing consumers' surplus by *bcgh*. However, the import or export tax would raise revenue by *bdfh*, which could be redistributed to offset any costs that the tax incurrs. It is feasible to consider a situation where both the consumers and producers establish taxes which together equal *bd*. The tax revenue could then be shared between the two groups in proportion with their tax rates. The conservationists would be better off by *cdbj*, and the global gain from such a trade tax would be *cbj*.

In contrast, if conservationists wish to ban completely the trade in ivory they must to be prepared to compensate the total losses incurred by the producers and consumers (i.e. *cea*). While the benefit that the conservationists receive from banning the trade (i.e. *cej*) may exceed the loss to the producers and consumers (i.e. *cea*), it may not be sufficient to overcome free-rider and other administrative costs of collecting the revenue from the conservationists and redistributing it to the producers and consumers.

5. Conclusions and Policy Implications

This paper has examined a number of key issues concerning the environmental implications of the international wildlife trade. Discussion has been given to the impact of the wildlife trade relative to other factors on endangered species, and the wider environmental costs, including the loss of biodiversity, associated with species extinction which may justify policy intervention to control the wildlife trade. The policy options to intervene in the wildlife trade to achieve environmental objectives have also been discussed.

Over-exploitation of wildlife for the international trade generally plays a minor role in species extinction relative to other factors, particularly habitat destruction and domestic wildlife consumption. However, for a few species over-exploitation for the international trade poses a major threat to their continued survival. Incentives to harvest and trade in wildlife are often distorted by a mixture of market and policy failures, which need to be taken into account by any regime designed to conserve wildlife.

Attempts to conserve wildlife species through habitat protection are constrained by having to intervene in national land use and wildlife policies. What is more, while protecting habitat may be a necessary condition to sustaining wildlife populations, it is not sufficient to ensure wildlife conservation. Previous approaches to preserving wildlife habitat have rarely created sufficient incentives to encourage local participation in wildlife conservation. As a result, habitat protection as a means of conserving wildlife is often undermined by continued over-exploitation - both for domestic consumption and for the international trade.

Intervention in the wildlife trade may be required to control the exploitation of wildlife for the international market where this has a significant impact on endangered species and to complement policies aiming at managing domestic wildlife consumption. However, designing appropriate trade policy interventions is complicated by a lack of trade data and information on the biological status of wildlife populations and the ability to enforce trade regulations.

Wildlife trade polices are often implemented in a piecemeal fashion with little analysis of their environmental implications nor their economic costs. For example, decisions to ban *all wildlife exports* from a country may achieve some environmental benefits but at a high economic costs. Similarly, multilateral decisions to ban *all trade in a particular species* may adversely affect domestic wildlife management programmes which involve sustainable management practices and rely on wildlife export revenues as a source of revenue. A more targeted approach to trade interventions for *specific species* that are threatened with extinction from over-exploitation for the international market and whose extinction would incur a high economic cost due to the foregone values of the resource may be needed. However, the greater the level of targeting - whether by specie or by country - the more difficult and expensive it is to regulate and enforce the policy.

CITES, the multilateral organisation that co-ordinates wildlife trade regulations, has relied upon the use of trade bans and quantity restrictions to manage the wildlife trade. Although the success of adopting such trade controls crucially depends upon incentives given to the various nations to cooperate there is little evidence that such incentive requirements have been met. In the cases where nations have implemented trade controls, the level of enforcement has generally been inadequate to prevent the emergence of illegal trade. In the long run, unless a ban is adequately adopted and enforced, it may have adverse effects on the economic incentives to conserve wildlife. Although banning the trade in specific species may be a necessary short term response to the critical situation facing many internationally traded wildlife species, more innovative and effective long term solutions for wildlife conservation and managing the wildlife trade need to be sought.

A more comprehensive trade mechanism that allows a limited amount of exports from those countries whose populations are being sustainably managed offers an incentive to both consumers and producers to accept a controlled legal trade and to enforce it. While the necessary mechanism for issuing permits and controlling the number of species traded already exists in CITES, it is important to improve the scientific and trade data used for establishing quotas for capture and export. Additional emphasis needs to be placed on promoting compliance with export regulations and on cooperation between exporters and regulating agencies. A properly constructed trade mechanism, using economic instruments such as taxes and subsidies to manage the trade, could enable a larger share of the profits from the wildlife trade to be channelled back into the producer states to encourage wildlife conservation and support improved monitoring of harvesting and export activities.

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Table 1.1

Past Extinctions and Current Endangered Species

a. Recorded Extinctions, 1600 to Present

	Mainland[1]	Island[2]	Ocean	Total	Approx. Number of Species
Mammal	30	51	2	83	4,000
Birds	21	92	0	113	9,000
Reptiles	1	20	0	21	6,300
Amphibians	2	0	0	2	4,200
Fish[3]	22	1	0	23	19,100
Invertebrates[3]	49	48	1	98	1,000,000+
Vascular Plants[4]	245	139	0	384	250,000

b. Endangered and Vulnerable Species

	Mainland[1]	Island[2]	Ocean	Total	% of Total Species
Mammals	159	48	9	216	5.4
Birds	91	87	0	178	1.9
Reptiles	41	21	6	68	1.1
Amphibians	14	0	-	14	0.3
Fish[3]	193	21	0	214	1.1
Invertebrates[3]	371	338	2	711	0.0
Vascular Plants[4]	3985	2706	0	6691	2.6

Notes:
1/ Landmasses greater than 1 million square kilometres.
2/ Landmasses less than 1 million square kilometres.
3/ Totals primarily representative of North America and Hawaii.
4/ Includes species, subspecies and varieties.

Source: Adapted from Reid and Miller (1989).

Table 1.2

Species Richness by Countries

Mammals	Birds	Reptiles
Indonesia (515)	Colombia (1721)	Mexico (717)
Mexico 9449)	Peru (1701)	Australia (686)
Brazil (428)	Brazil (1622)	Indonesia (600)
Zaire (409)	Indonesia (1519)	India (383)
China (394)	Ecuador (1447)	Colombia (383)
Peru (361)	Venezuela (1275)	Ecuador (345)
Colombia (359)	Bolivia (1250)	Peru (297)
India (350)	India (1200)	Malaysia (294)
Uganda (311)	Malaysia (1200)	Thailand (282)
Tanzania (310)	China (1195)	Papua N.G.
(282)		

Source: McNeely *et al.* (1990).

Table 1.3

Wildlife Habitats

a. Wildlife Habitats in the Afrotropical Region[1], 1986

	Original Area '000sq.km	Remaining Area '000sq.km	Percent Remaining %
Dry Forests	8217	3416	41.6
Moist Forests	4700	1868	39.7
Savannah/Grassland	6955	2835	40.8
Scrub/Desert	177	172	97.8
Wetland/Marsh	62	44	70.9
Mangroves	88	39	44.6

b. Wildlife Habitats in the Indomalayan Region[2], 1986

	Original Area '000sq.km	Remaining Area '000sq.km	Percent Remaining %
Dry Forests	3414	940	27.5
Moist Forests	3362	1227	36.5
Savannah/Grassland	46	12	36.0
Scrub/Desert	816	119	14.5
Wetland/Marsh	414	160	38.8
Mangroves	95	40	42.4

Notes: 1/ data from Angola, Benin, Botswana, Burkina Faso, Burundi, Cameroon, Central African Republic, Chad, Cote d'Ivoire, Djibouti, Equatorial Guinea, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea Bissau, Kenya, Mozambique, Namibia, Niger, Nigeria, Rwanda, Senegal, Sierra Leone, Somalia, South Africa, Sudan, Swaziland, Tanzania, Togo, Uganda, Zaire, Zambia and Zimbabwe.

2/ data from Bangladesh, Bhutan, Brunei, China, Hong Kong, India, Indonesia, Japan, Cambodia, Laos, Malaysia, Myanmar, Nepal, Pakistan, Philippines, Sri Lanka, Taiwan, Thailand and Vietnam.

Source: World Resources Institute (1991).

Table 2.1

Causes of Extinction

a. Causes of Past Extinctions (percent due to each dominant cause)

	Habitat Loss	Over-Exploitation[1]	Species Introduction	Other[2]
Mammals[3]	19	23	20	38
Birds[3]	20	11	22	39
Reptiles[3]	5	32	42	21
Fish[4]	35	4	30	52

b. Current Threats to Species (percent due to each cause)

	Habitat Loss	Over-Exploitation[1]	Species Introduction	Other[2]
Mammals[4]	68	54	6	20
Birds[4]	58	30	28	2
Reptiles[4]	53	63	17	9
Amphibians[4]	77	29	14	3
Fish[4]	78	12	28	2

Notes: 1/ Includes commercial, subsistence, sport hunting and live animal capture for pet, zoo and research trades.
2/ Includes unknown and predator control.
3/ Value reported represents the percentage of species whose extinction was caused primarily by the factor indicated.
4/ Value reported represents the percentage of species whose endangerment or extinction is influenced by each factor, thus row exceeds 100%.

Source: Reid and Miller (1989).

Table 2.2

Trade in Wildlife and Wildlife Products, 1986

	Live Primates (number)	Cat Skins (number)	Raw Ivory (kg)	Live Parrots (number)	Reptile Skins (number)
World					
imports	51256	192402	429549	696002	10480798
exports	51256	192402	429558	618539	10480798
Africa					
imports	423	753	2254	14255	65980
exports	8879	2272	130391	169237	399256
North and Central America					
imports	23588	38683	5051	318162	1388169
exports	6756	89590	126	26550	600161
South America					
imports	82	11	3	2811	47191
exports	6135	6150	NA	256634	1403942
Asia					
imports	7921	16093	355782	60365	4682360
exports	24622	72969	269655	126538	6878809
Europe					
imports	16063	136736	53197	298973	4070201
exports	4781	18335	28394	37234	1193904

Notes: NA - not available

Source: World Resources Institute (1991).

Table 2.3

Major Importers and Exporters of Live Parrots (1981-86)

a. Imports ('000s)

Country	1981	1982	1983	1984	1985	1986
USA	199	281	301	312	311	305
Germany	35	58	56	54	55	60
UK	9	17	24	16	22	34
Japan	21	45	39	39	23	28

b. Exports ('000)

Country	1981	1982	1983	1984	1985	1986
Argentina	73	89	114	109	179	178
Tanzania	22	74	52	45	70	84
Indonesia	26	33	78	79	44	59
Guyana	29	27	25	38	27	30
Uruguay	8	19	33	39	18	21
Peru	16	39	19	52	34	17
Senegal	7	25	21	27	20	28
World (gross Total)	362	536	555	593	615	619

Source: United Nations Environment Programme (1989).

Table 2.4

Surivorship of Nestling Psittacines[a]
at Various Stages of Legal Trade

Stage	Surivorship[b]	Mortality Rate[c]	Mortality[d]
Capture from nest	1.00	0.10	0.10
Confinement by trapper	0.90	0.30	0.27
Transportation within Mexico	0.63	0.20	0.13
Confinement by exporter	0.50	0.10	0.05
International Transportation	0.45	0.04	0.02
Quarantine in importer country (USA)	0.43	0.06	0.03
Pet store in USA	0.40	0.03	0.01
Pet owner in USA	0.39	--	--

Notes: a/ data from 25,376 Mexican psittacine birds in ninety-four shipments with twelve species.
b/proportion of birds still surviving at a given stage out of the original 25,376 birds.
c/ proportion of birds entering in a particular stage that die during it.
d/ probability of dying during each stage.

Source: Ingio-Elias and Ramos (1991).

Table 2.5

Major Importers and Exporters of Cat Skins (1981-86)

a. Imports ('000s)

Country	1981	1982	1983	1984	1985	1986
USA	26	23	23	19	20	17
Germany	198	148	308	88	77	82
UK	7	2	10	16	11	16
France	3	2	15	46	4	8

b. Exports ('000)

Country	1981	1982	1983	1984	1985	1986
China	8	8	3	43	60	68
Paraguay	126	70	34	3	0	0
Bolivia	0	0	3	30	3	6
Argentina	6	0	0	12	NA	NA
World (gross Total)	380	244	445	227	190	192

Source: United Nations Environment Programme (1989).

Table 2.6

Legislation Affecting the International Rhino Horn Trade in Principal Consumer States

	CITES Party	Import Ban	Re-Export Ban
Brunei	No	na	na
China	Yes	na	No
Hong Kong	Yes	1979	1986
Japan	Yes	1980	1980
Macao	No	1985	na
Singapore	Yes	1986	1986
Rep. of Korea	No	1986	1984
Taiwan	No	1985	1985
Thailand	Yes	No	No
Yemen Arab Republic	No	1982	1987

Notes: na = not available

Source: Joint Nature Conservation Committee (1991).

Table 2.7

Trade in Raw Ivory, (1979–88)

Raw Ivory Exports by Major African Exporters, 1979–88
(tonnes)

	1979	1980	1981	1982	1983	1984	1985	1986*	1987*	1988*
Botswana	13	5	9	5	6	4	17	.4	.4	.1
Burundi	139	126	61	46	124	104	215	118	50	0
CAR	181	103	107	202	204	127	117	19	2	.1
Chad	27	4	13	20	34	6	.03	0	7	0
Congo	93	175	237	102	53	95	72	12	86	19
Gabon	4	1	1	.2	.9	.2	.4	.5	4	14
Kenya	46	31	6	12	4	13	19	7	.1	0
S. Africa	42	35	33	28	46	36	50	41	18	8
Sudan	124	206	269	270	337	68	22	78	70	0
Tanzania	33	45	27	18	25	45	116	302	56	43
Uganda	25	19	45	12	13	100	206	1	0	.2
Zaire	160	84	45	78	157	90	22	21	12	11
Zambia	16	22	20	34	18	2	14	11	4	2
Zimbabwe	3	2	1	14	14	21	23	6	0	7
All Africa Min. Total	944	952	905	890	1032	798	912	719	331	142
All Africa Max. Total	950	1164	912	900	1040	796	912	605	331	142

Note: These are the 'best estimates' of African exports, disaggregated by country of export, using import and export data from customs sources, and CITES data. Both legal and illegal exports are likely to be included. For starred years (*), because of the adoption of the management quota system by CITES, there may have been an incentive to evade both customs and CITES systems in some countries.

Net Imports of Raw Ivory by Major Consumers
(tonnes)

	1979	1980	1981	1982	1983	1984	1985	1986*	1987*	1988*
USA	6	23	11	7	20	55	24	17	21	9
FR Germany	74	101	32	35	43	-7	16	7	2	1
UK	-5	-26	0	-3	2	3	20	-1	7	3
Hong Kong	366	376	427	318	420	267	109	129	150	133
India	17	19	19	24	23	30	21	0	6	4
France	89	22	7	4	11	21	5	5	4	-2
China	7	10	10	54	20	7	7	19	39	50
Japan	270	240	256	205	174	179	206	29	103	75
Thailand	1	1	2	4	-5	-12	-2	1	0	-3
Belgium	16	-90	-240	-123	-105	-116	0	0	-10	12
Singapore	-7	-4	3	7	0	120	60	324	-140	-129
Macau		0	0	5	16	38	82	57	0	11
Taiwan	11	10	17	18	20	34	21	18	80	5
Total Net Imports, All Countries (Min. estimate)	979	967	895	891	1018	710	749	600	370	153

Note: These 'best estimates' of final demand for raw ivory by the main consuming countries are based on customs trade statistics and CITES documentation.

Minus figures indicate that countries were net exporters in that year, most likely through de-stocking.

For starred years (*), because of the adoption of the management quota system by CITES, there may have been an incentive to evade both customs and CITES systems in some countries.

Source: Barbier *et al.* (1990).

Table 3.1

Parks and Protected Areas Designated

	1930-59 No. Area	1930-59 Area (000ha)	1960-98 No. Area	1960-98 Area (000ha)	Total Parks No. Area	Total Parks Area (000ha)
World	689	60151	2671	363580	4320	486400
Africa	123	29443	282	53910	486	95491
N. & Central America	114	7940	238	137678	587	177584
S. America	53	7199	249	58551	315	66253
Asia	131	6819	733	49506	960	60534
Europe	151	3429	677	20639	1032	27179
USSR	42	2593	109	16529	168	20246
Oceania	75	2728	388	26570	767	38918
Antarctic Treaty Territory	0	0	5	196	5	196

Notes: Areas included are those in IUCN management Categories I through V. The table only includes areas over 1,000 ha, and only those protected by the highest competent authority (i.e. state parks and reserves are not included).

Source: United Nations Environment Programme (1989).

Table 4.1

Selected National Trade Bans and Trade Restrictions Since 1973

1975	Paraguay institutes a ban on trade
1977	Nicaragua bans commercial wildlife trade
1978	Honduras bans most commercial mammal and bird trade
1978	Zaire suspends ivory exports
1980	Ghana bans parrot trade
1980	Belize bans all commercial trade
1980	Chile prohibits exports of several Appendix II species
1981	Gabon suspends raw ivory exports
1981	Argentina prohibits exports of all indigenous felids
1981	Ecuador bans commercial exports
1981	Pakistan bans export of all wild mammals and reptiles
1982	Mexico imposes ban on exports of native species
1983	Sudan bans raw ivory exports
1984	Bolivia imposes one year total export ban
1984	India bans snake exports
1984	Zaire bans grey parrot exports
1984	Malaysia bans long-tailed macaque exports
1984	Liberia bans ivory exports
1986	Argentina bans most trade
1986	Honduras bans mammal and bird exports
1986	Laos bans all exports
1986	Senegal bans all exports
1986	Haiti bans native wildlife exports
1986	Indonesia bans all raw skin exports
1987	Guyana bans all exports
1987	India bans commercial exports of frogs
1987	Papua New Guinea imposes export ban on live vertebrates
1987	Pakistan extends its export ban
1987	New York State bans the import of wild caught birds
1989	United States bans the import of African elephant ivory

Source: Trexler (1989).

Table 4.2

The Success of CITES Implementation

a. Party Compliance with CITES Reporting Requirements

	1980	1981	1982	1983	1984	1985	1986	1987
Number of Parties	61	75	78	88	88	90	93	96
Number of Reports	36	41	44	50	57	68	71	63
Percentage Submitting	59%	55%	56%	61%	55%	66%	76%	65%

b. Correlations Between Reported Imports and Exports

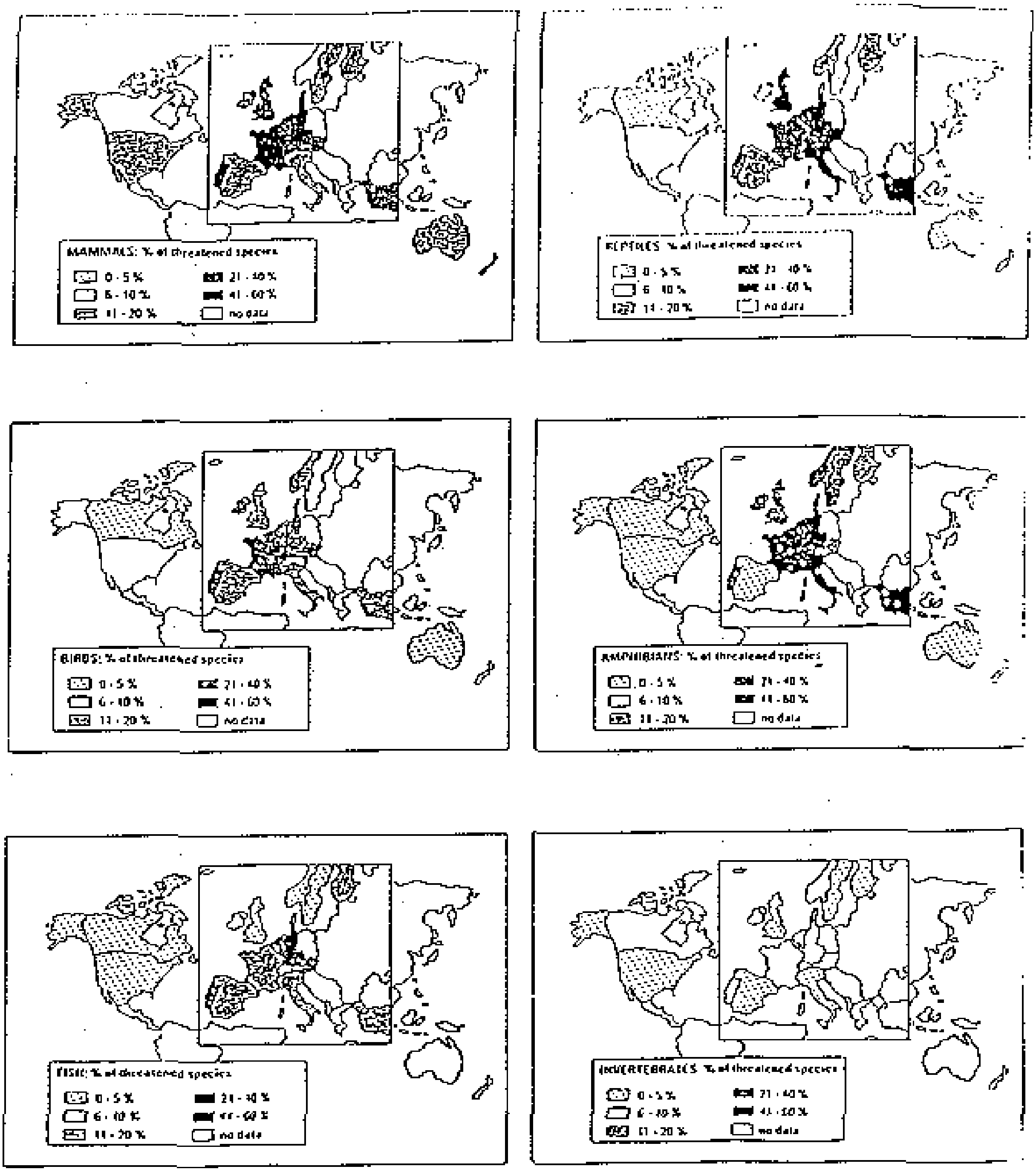
	1979	1980	1981	1982	1983	1984	1985
Non-Party	32%	33%	26%	28%	21%	28%	21%
Non-Reporting Party	4%	9%	9%	11%	18%	5%	8%
Perfect Correlation	<5%	<5%	4%	5%	8%	14%	16%
No Correlation	>75%	>75%	65%	71%	61%	78%	74%

Notes: The first two rows estimate the percentage of transactions that could by definition not correlate due to the causes mentioned. The latter two rows estimate the percentage of transactions, of those that could potentially correlate, which actually do. These statistics are based on analysis of transactions involving several high-profile species and product transactions, and will significantly overstate the average frequency of correlations across all CITES transactions.

Source: Trexler (1989).

Figure 1.1

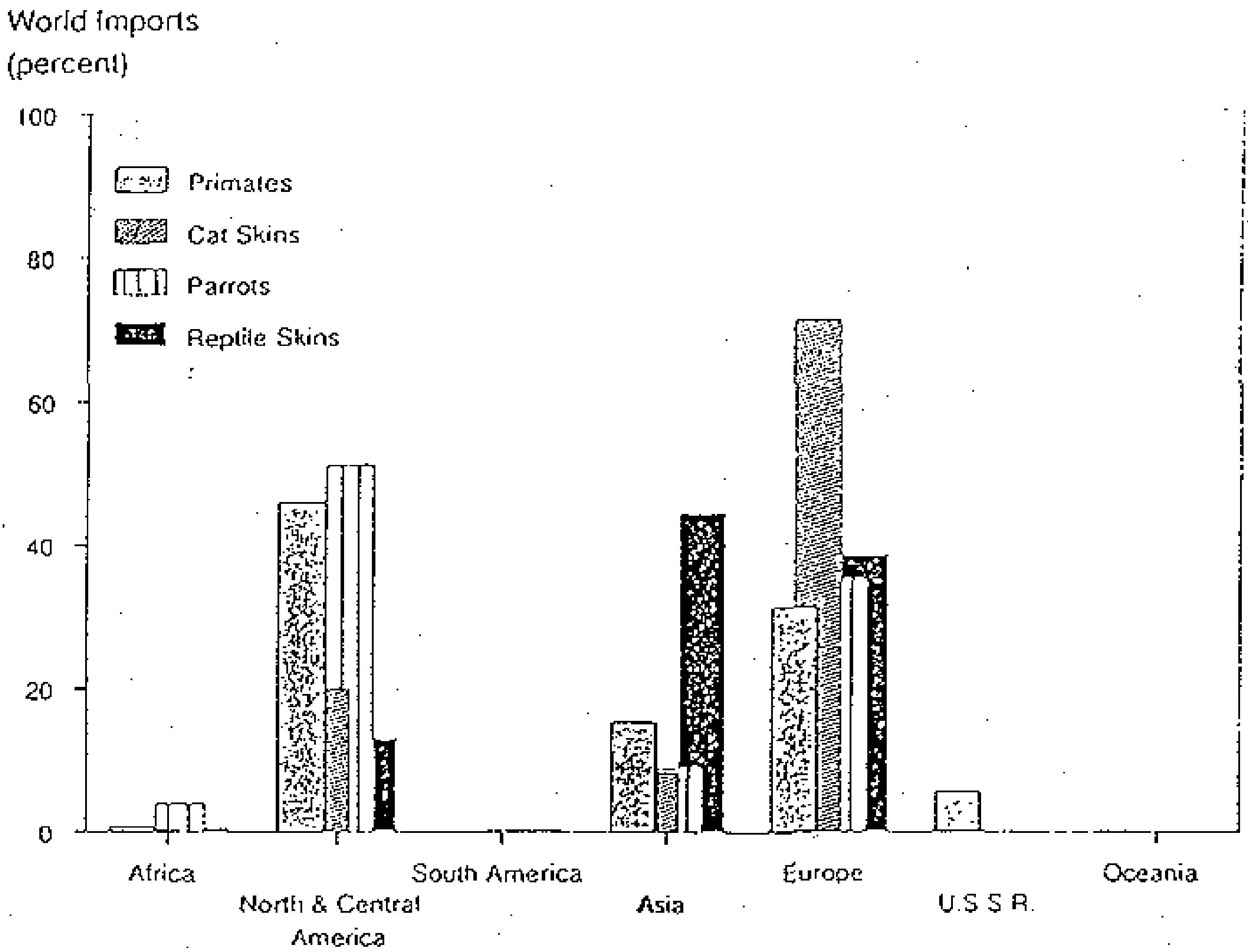
State of Wildlife - OECD Countries, late 1980s



Source: Organisation for Economic Co-operation and Development (1991).

Figure 2.1

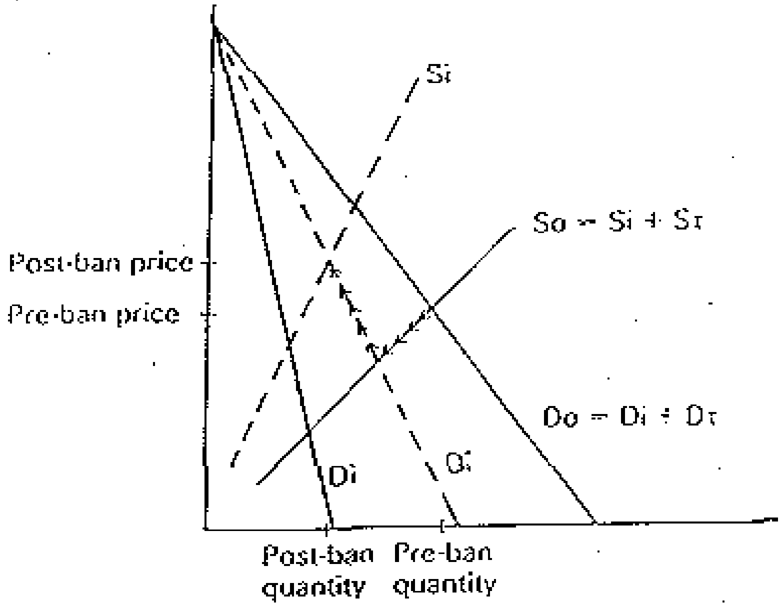
Major Importers of Wild Animals and Skins



Source: World Resource Institute (1991).

Figure 4.1

The Effects of an Ivory Ban



So – Supply curve before ban, illegal and legal supply (Si + Sr)

Si – Supply curve after ban, illegal supply only

Do – Demand curve before ban, illegal and legal demand (Di + Dr)

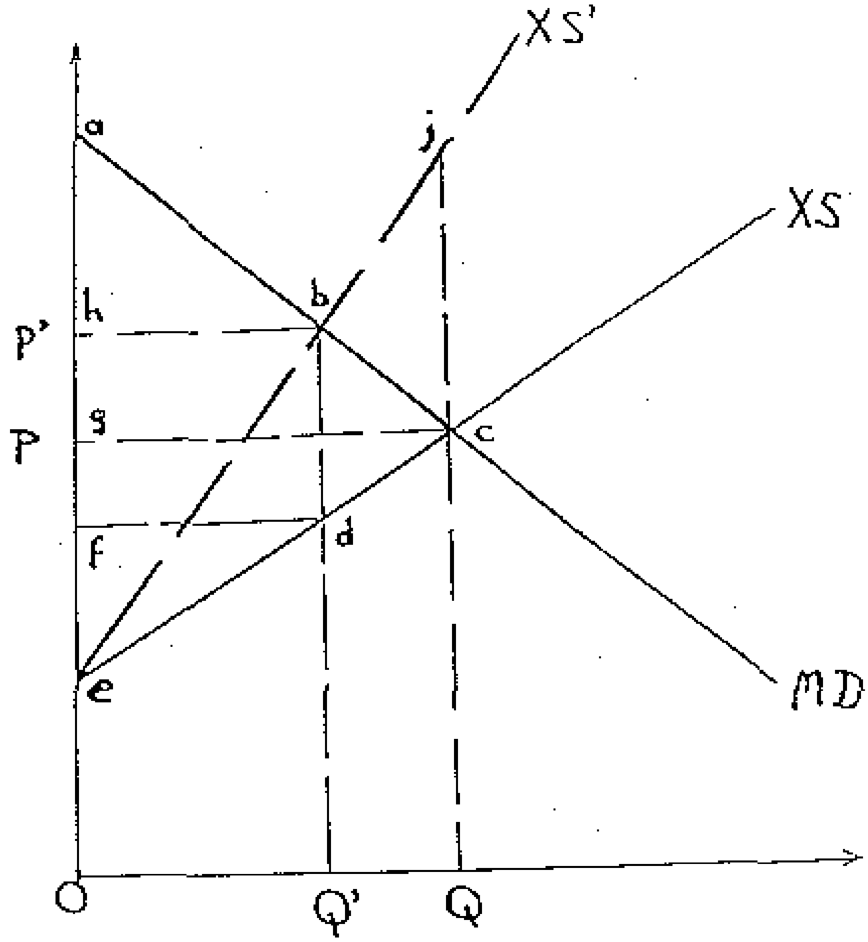
Di – Demand curve before ban, illegal demand only

Di' – Demand curve after ban, illegal demand only

Source: E.B. Barbier *et al.* (1990).

Figure 4.2

Effects of Taxing Production or Consumption to Improve the Global Environment



Source: Anderson (1991).

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BOOKS

Edward B. Barbier

Economics, Natural-Resource Scarcity and Development: Conventional and Alternative Views, Earthscan, London, 1989 (paperback £15.00)

The history of environmental and resource economics is reviewed; then using insights from environmentalism, ecology and thermodynamics, Barbier begins the construction of a new economic approach to the use of natural resources and particularly to the problem of environmental degradation. With examples from the global greenhouse effect, Amazonian deforestation and upland degradation on Java, Barbier develops a major theoretical advance and shows how it can be applied. This book breaks new ground in the search for an economics of sustainable development.

David W. Pearce, Anil Markandya and Edward B. Barbier

Blueprint for a Green Economy, Earthscan, London, 1989 (paperback £8.95)

This book was initially prepared as a report to the Department of Environment, as part of the response by the government of the United Kingdom to the Brundtland Report, *Our Common Future*. The government stated that: '...the UK fully intends to continue building on this approach (environmental improvement) and further to develop policies consistent with the concept of sustainable development.' The book attempts to assist that process.

Edward B. Barbier, Joanne C. Burgess, Timothy M. Swanson and David W. Pearce

Elephants, Economics and Ivory, Earthscan, London, 1990 (paperback £10.95)

The dramatic decline in elephant numbers in most of Africa has been largely attributed to the illegal harvesting of ivory. The recent decision to ban all trade in ivory is intended to save the elephant. This book examines the ivory trade, its regulation and its implications for elephant management from an economic perspective. The authors' preferred option is for a very limited trade in ivory, designed to maintain the incentive for sustainable management in the southern African countries and to encourage other countries to follow suit.

Gordon R. Conway and Edward B. Barbier
After the Green Revolution: Sustainable Agriculture for Development, Earthscan Pub. Ltd., London, 1990 (paperback £10.95)

The Green Revolution has successfully improved agricultural productivity in many parts of the developing world. But these successes may be limited to specific favourable agro-ecological and economic conditions. This book discusses how more sustainable and equitable forms of agricultural development need to be promoted. The key is developing appropriate techniques and participatory approaches at the local level, advocating complementary policy reforms at the national level and working within the constraints imposed by the international economic system.

David W. Pearce, Edward B. Barbier and Anil Markandya
Sustainable Development: Economics and Environment in the Third World, London and Earthscan Pub. Ltd., London, 1990 (paperback £11.95)

The authors elaborate on the concept of sustainable development and illustrate how environmental economics can be applied to the developing world. Beginning with an overview of the concept of sustainable development, the authors indicate its implications for discounting and economic appraisal. Case studies on natural resource economics and management issues are drawn from Indonesia, Sudan, Botswana, Nepal and the Amazon.

David W. Pearce and R. Kerry Turner
** *Economics of Natural Resources and the Environment*, Harvester-Wheatsheaf, London, 1990.

This textbook covers the elements of environmental economics in theory and in application. It is aimed at undergraduates and includes chapters on sustainable development, environmental ethics, pollution taxes and permits, environmental policy in the West and East, recycling, and optimal resource use.

David W. Pearce, Edward B. Barbier, Anil Markandya, Scott Barrett, R. Kerry Turner and Timothy M. Swanson
Blueprint 2: Greening the World Economy, Earthscan Pub. Ltd., London, 1991 (paperback £8.95)

Following the success of *Blueprint for a Green Economy*, LEEC has turned its attention to global environmental threats. The book reviews the role of economics in analyzing global resources such as climate, ozone and biodiversity, and considers economic policy options to address such problems as global climate change, ozone depletion and tropical deforestation.

E.B. Barbier and T.M Swanson (eds.)
Economics for the Wilds: Wildlife Wildlands, Diversity and Development, Earthscan Pub. Ltd., London, 1992 (paperback £12.95).

This collection of essays address the key issues of the economic role of natural habitat and wildlife utilization in development. The book argues that this role is significant, and composes such benefits as wildlife and wildland products, ecotourism, community-based wildlife development, environmental services and the conservation of biodiversity.

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