Property Rights and Wildlife Utilisation:

generating incentives for conservation and economic development

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Abstract

The accelerating depletion of natural resources (biological diversity) range from degraded ecosystems, endangered species, and loss of genetic resources. This depletion has raised concern over future human economic welfare among other things. Conservation strategies have been implemented to preserve remaining biological diversity. The focus of this thesis is on conservation strategies to halt the loss of wild species. Protected areas and trade bans are the most recognised conservation measures. These strategies have their limitations however. This study will argue that increasingly the preservation of remaining wild species will be through economic incentives, specifically at the local community level.

If we are serious about saving wild species, our behaviour towards the utilisation of wildlife must change. A recent innovative idea is to look upon conservation as a form of economic development. Strategies that can lead to the successful implementation of this concept include conservation partnerships that actively involve local communities especially in developing countries, and the commercial and sustainable use of wild species.

Underlying the effectiveness and efficiency of these strategies are property right institutions and markets. Economic theory argues that natural resources will be protected only if direct economic benefits accrue to those most responsible for the care of these resources. In essence, the wise management of biological diversity must generate conservation and economic development benefits. This study examines and discusses the above issues.

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1
Introduction

Many of the world's richest ecosystems and many of the world's poorest people can be found in the tropics **(Randall, 1991, 64).**

The economics of biodiversity conservation probably represent the most profound challenge to environmental economics **(Moran and Pearce, 1997, 109).**

1.1 RESEARCH TOPIC

A recent advancement in conservation theory is the sustainable utilisation of wild species (Webb, 1991; Swanson and Barbier (eds.), 1992; Freese, 1998). It is recognised as an appropriate and viable conservation tool by the World Conservation Union (IUCN), the World Commission on Environment and Development (WCED), the Convention on Biological Diversity (CBD) and *Agenda 21*, as well as in the natural and social sciences. Further, such a strategy has implications for the economic development of local communities in developing countries (Wells *et al.*, 1992; Swanson and Barbier (eds.), 1992; Furze *et al.*, 1996).

Important to this research is human behaviour and decision-making regarding the natural environment. Hence this study examines conservation policies; this includes both traditional and recent innovative strategies. A critique is presented on the limitations of traditional strategies: protected areas and trade restrictions while the advantages of sustainable use and legal commercial trade are discussed. It is hoped that such strategies will benefit both threatened wildlife and the economic welfare of local communities.

The motivation for this study is to understand the linkages and interdependence between conservation of wildlife and local communities. Better understanding is called for because of the concern over the current status of many wild species, concern over future uncertainty and all the possible consequences, and finally concerns for the economic plight of poor people in developing countries. Although the essence of this study is conservation, the heart of this study is property rights. Specifically the study will examine how assigning property rights to local communities combined with sustainable commercial use may augment conservation efforts and increase the wealth of local communities.

The research will identify gaps in knowledge that will require further study. These include (but are not restricted to) the role of local community participation, the impact of sustainable use on local communities, what are the most appropriate institutions for effective conservation, and incomplete information (biological and economic) with regard to our current knowledge of the sustainable use of wildlife.

1.2 THESIS OUTLINE

Chapter 1 introduces the major issues of this study. It will outline the problems and questions that this research examines. Important concepts are defined and the aims of the study are stated. The literature relevant to this study is reviewed in Chapter 2. This second chapter includes sections on threats to biodiversity, traditional conservation strategies, new innovative strategies, and economic development. Property rights are discussed in detail in Chapter 3. The conclusion of Chapter 3 is that before there is investment such as protection in conservation, the relevant parties must change their behaviour towards the use of wildlife. One way this can be achieved is through allocating property rights. A dominant theme of this study is the importance of well-defined property rights and the institutions to enforce the rights and duties associated with that property rights. Specifically discussed will be the advantages of assigning communities property rights to local wildlife. According to the *World Conservation Strategy* and *Caring for the Earth*, people are the best protectors of their own wildlife

(Webb *et al.*, 1993). However, with any change in policies, programs or institutions there are costs, benefits, and trade-offs involved. Hence the importance of economic analysis.

Conservation strategies are examined in Chapters 4 and 5. Chapter 4 discusses the protected area and trade restriction strategies. A critique of these strategies is presented. Chapter 5 analyses recent innovative strategies. These include the involvement of local communities in conservation partnerships plus the utilisation of wild species. Details of the advantages of sustainable use and legal commercial trade are given. Limitations of these strategies will also be discussed. The major conclusions from this study are presented in Chapter 6.

Relevant to this study are the socio-economic effects of conserving wildlife and habitats. Areas for improvement will be identified. This will include, what ownership regimes should prevail for effective resource use? What conditions or incentives will encourage local communities to conserve wildlife? Will sufficient incentives be generated through legal international trade? How will the practice of sustainable use benefit both wildlife and local communities? This affirmative position has some prior support (Swanson, 1994a; OECD, 1997). Finally, comments, discussions, and case studies are presented throughout the text.

Several assumptions are important to this study. First, species diversity is the appropriate measure of biodiversity. Second, maintaining a diverse array of species and their habitats has current and future value. Third, the only way to conserve wildlife is to preserve natural habitat.

1.3 RELEVANCE OF RESEARCH

Economics is the study of choice and the allocation of scarce resources: land, labour, and capital. Conservation is about maintaining natural resources. Historically the two were viewed as separate. Now, there is a growing recognition that economics has an important role to play in the conservation of biodiversity. That is, conservation requires choices to be made regarding the allocation of resources to maintain and protect natural resources and how these natural resources may be optimally utilised (Freese, 1998).

Research is required to identify how both areas of study can be integrated and what gains can be achieved from this integration. According to Meffe and Carroll (1994a, 21), 'a strong cross-disciplinary perspective is desirable and necessary for success in conservation.' In short, such an approach will lead to a better understanding of natural and social processes and how they interact.

1.3.1 Biodiversity: how many species, and the human/biodiversity connection

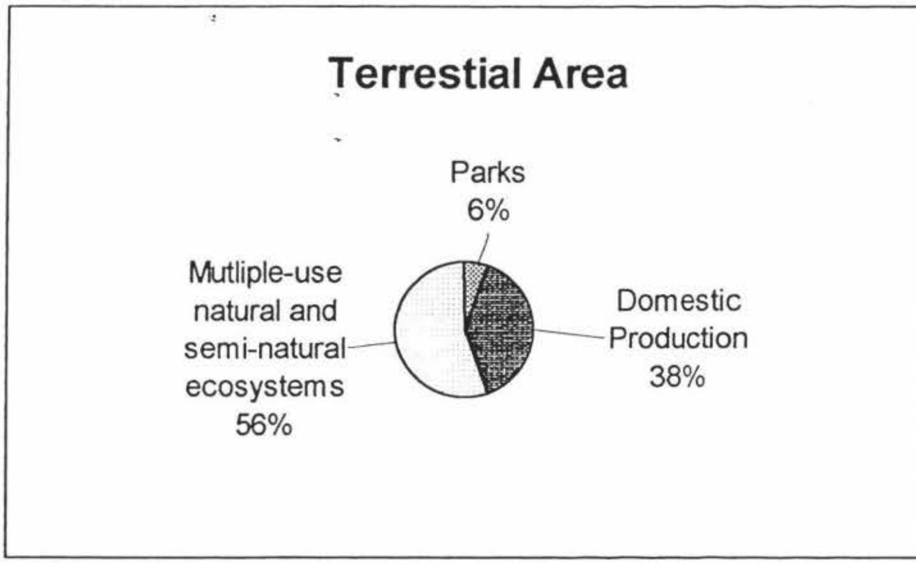
The range of living organisms on Earth is referred to as biodiversity or more correctly biological diversity (Moran and Pearce, 1997). Approximately 1.4 million species have been named and classified (McNeely *et al.*, 1990; Orians, 1994; Raven and McNeely, 1998). The Earth's stock of species is however estimated to be between 10-100 million (UNEP, 1993; Myers, 1994). Therefore, at a maximum we have detailed information regarding the status of only about 10 percent of species (Raven and McNeely, 1998).

Humans share resources and have close relationships with the millions of other species that inhabit the biosphere. A number of studies have found a positive relationship between measures of human emotional well-being and natural (non-built) environments (Gowdy, 1997). Many people are members of environmental organisations. For instance, in 1990, US$50 million was donated to WWF-for-Nature, just one of many environmental organisations (Swanson, 1991). Further, outdoor recreation is a popular activity for humans. Biodiversity also has an instrumental value as goods and services used in human economic activities (Callicott, 1994). According to Swanson (1994a, 7),

> *the fundamental avenue to human development has been the conversion of the naturally existing forms of assets to other forms more highly valued by human activities.*

Although this process has advanced human development it has resulted in the decline of biodiversity (WCED, 1987; Swanson, 1994a). According to Freese (1998), 38 percent has either been converted to alternative uses or altered in such a way as to seriously affect native biodiversity. Nevertheless large areas, approximately 62 percent of the biosphere remain in a natural or seminatural state and open for potential multiple uses (Freese, 1998) (Fig 1.1). Hence, opportunities still exist to use these areas and the resources within them in an optimal and sustainable way, both socially and ecologically. In short, future economic sustainability and the ultimate survival of the human species depend on the maintenance and wise use of remaining biological resources (WCED, 1987; Meffe and Carroll, 1994d).

Fig 1.1: Relative proportions of land area uses.



Source: Freese, 1998, 8.

1.3.2 Extinction of biodiversity and related consequences

Biological resources can be classified as renewable resources. Current species richness is the result of millions of years of evolution (Orians, 1994; Raven and McNeely, 1998). Extinction (there have been several mass extinctions of species in the past) and speciation are natural and dynamic phenomena and an integral part in evolution (Swanson, 1994a). Evidence now suggests that current species extinction rates and the number of endangered and threatened species are on a scale greater than at any other time in history (Raven and McNeely, 1998) (Table 1.1, 1.2). Some claim another mass extinction is imminent (Myers, 1994; Raven, 1995).

These rates appear to be higher than the natural or background rate (Myers, 1994; Pimm *et al.*, 1995; Raven, 1995; Moran and Pearce, 1997; Tuxill and Bright, 1998). Scientists have concluded that perhaps a quarter of the Earth's total biological diversity is at serious risk of extinction during the next 20-30 years (McNeely, 1990; Myers, 1994; Jordan, 1995; Pimm *et al.*, 1995; OECD, 1997). The Global 2000 Report states that,

> *between half a million and 2 million species —15 to 20 percent of all species on earth— could be extinguished by the year 2000* (Regenstein, 1994, 266).

According to McNeely *et al.* (1990) the main threat to biodiversity and the cause of this high rate is human mismanagement (Box 1.1). That is, it is human economic decisions, behaviours, and choices that now largely determine which biological resources are maintained and which are driven to extinction not natural processes.

Many of these estimates are calculated using island biogeographic theory (Boecklen and Gotelli, 1984; Myers, 1994). Caution is then needed in applying this theory to mainland species.[1] Further, estimates of biodiversity loss are very uncertain because knowledge about species is limited (Moran and Pearce, 1997).

Box 1.1: Recorded threats from known extinctions and endangerment: animals and birds.

486 animal species extinctions have been documented since 1600: 80 have become extinct from hunting and trapping by humans for food, skin, sport, live trade, or as pests; 114 have become extinct from the effects of purposeful or accidental introductions of exotic species and diseases into new areas by humans; 98 have become extinct from habitat destruction; 5 others have become extinct from other minor or uncertain effects. This leaves 189 for which no cause of extinction is known.

Currently just over 1000 out of 9500 bird species are endangered. Of these, hunting and trapping for food, sport, feathers or the bird trade threaten 122 species; 76 are threatened by the introduction of exotic species into the bird's ranges; 322 species are threatened by habitat destruction; and about 30 species are threatened by a variety of miscellaneous causes.

Source: Smith *et al.*, 1995, 129.

[1] See Box 5B, Myers, 1994, 121-22 that discusses problems of extinction rate estimates due to tropical deforestation.

Table 1.1: Conservation status of mammals, 1996.

Status	Total (number)	Share (percent)
Not currently threatened	2,661	61
Nearing threatened status	598	14
Threatened- vulnerable to extinction	612	14
Threatened- in immediate danger of extinction	484	11

Source: *1996 IUCN Red List of Threatened Animals* cited in Tuxill and Bright, 1998, 47.

Table 1.2: Conservation Status of Amphibians surveyed, 1996.

Status	Total (number)	Share (percent)
Not currently threatened	348	70
Nearing threatened status	25	5
Threatened- vulnerable to extinction	75	15
Threatened- in immediate danger of extinction	49	10

Numbers reflect only the species surveyed for conservation status, not the total number of species known in each group.
Source: *1996 IUCN Red List of Threatened Animals* cited in Tuxill and Bright, 1998, 51.

The loss of biodiversity is of concern. Raven (1995) argues,

of all the global problems that confront us, this [species extinction] is the one that is moving the most rapidly and the one that will have the most serious consequences.

Further, the consequences of this loss are largely unknown (Moran and Pearce, 1997). It is however thought that the degradation and loss of biodiversity will have an adverse impact on human welfare and future economic development. Therefore it is important to analyse why biodiversity is being lost, why it matters, and what can be done about it. Hence, issues dealing with the surmised effects are now major contemporary topics; this includes *inter alia* endangered species. As a result of these concerns there has been an increase in research and analysis on the loss of biodiversity. For example, the *World Conservation Strategy* has the aim to,

provide a comprehensive framework to stimulate urgent, positive, innovative, and co-ordinated action to stem the loss and degradation of the worlds biological resources and enhance the contribution of these resources to human well-being (McNeely *et al.*, 1990, 113).

The United Nations Environmental Programme (UNEP) (1993) perceives, 'its conservation [global biodiversity] has thus come to be viewed as one of the key environmental challenges of the 1990's.' In essence biodiversity is valuable, this does not mean that it should not be used, rather if used wisely this should help generate incentives for conservation and increase human economic welfare.

1.3.3 Wildlife and attitudes to conservation

"Wildlife" can be defined as, 'those species that have been left out of this closely managed [conservation] process' (Swanson, 1992a, 6). The term is also associated with animals of economic value (Naughton-Treves and Sanderson, 1995). In general, people are interested in conserving wild animals (Simon, 1995). Individuals do however conserve wild species for different reasons.

Farve (1993) identifies and describes five categories of western attitude towards wildlife, they are: (1) *Survivor*, who would use a species today, as they may not be alive tomorrow. Human interests and survival are more important than any species. (2) *Exploiter*, who use wildlife for financial gain. It is better to make some money from the species now before anyone else and the future will take care of itself. (3) *Conservationist*, where human interests are still important, and wildlife is to be utilised. However, in the best interests of humans it is better that species are not lost. (4) *Environmentalist,* who are interested in preserving species because they are an important component in natural ecosystems. (5) *Animal protectionist,* who are interested in individual organisms. Humans have no right to inflict pain, suffering or death on nonhuman organisms.

Conservation of wildlife is equally or more important for people in developing countries (see Newmark *et al.*, 1993, for local peoples attitude to conservation in Tanzania). The main reasons for conservation in developing countries may however differ from those evident in developed countries. For instance, Barnes (1996) suggests that utilitarian values, particularly in the poor rural communities are more important than preservation values. Irrespective of which view is taken, all attach values to wildlife. From an economic point of view and conservation it is optimal then to try and maximise the net benefit from that species.

1.3.4 Biodiversity and developing countries

Biological studies have shown that species diversity increases closer to the equator. Myers (1994) has identified several regions of biodiversity "hot spots". These are countries and habitats with the greatest diversity or the most endangered species (Table 1.3). The richest areas in species numbers are tropical moist forests (McNeely *et al.*, 1990; Pearce 1991; Swanson, 1994a; Blackmore and Reddish, 1996). It is estimated that these forests cover some 7 percent of the Earth's land surface but contain 50-90 percent of all known species (WCED, 1987; Swingland, 1993; Myers, 1994).

Table 1.3: Number of endemic species in some "hot spot" areas.

Region	Area (km^2)	Vascular plants	Mammals	Reptiles	Amphibians
Cape region (South Africa)	134,000	6000	16	43	23
Upland western Amazonia	100,000	5000	—	—	ca.70
Atlantic coast of Brazil	1,000,000	5000	40	92	168
Madagascar	62,000	4900	86	234	142
Philippines	250,000	3700	98	120	41
North Borneo	190,000	3500	42	69	47
Eastern Himalayas	340,000	3500	—	20	25
Southwestern Australia	113,000	2830	10	25	22
Western Ecuador	27,000	2500	9	—	—
Colombian Chocó	100,000	2500	8	137	111
Peninsular Malaysia	120,000	2400	4	25	7
California Floristic Province	324,000	2140	15	25	7
Western Ghats (India)	50,000	1600	7	91	84
Central Chile	140,000	1450	—	—	—
New Caledonia	15,000	1400	2	21	0

Source: Myers and WCMC cited in Orians, 1994, 95.

A large percentage of these areas are located in developing countries. Conservation International has identified these areas as: Latin America; the southern Guianas, southern Venezuela, northernmost Brazil Amazonia, and parts of the western

Amazonian lowlands of Brazil, Colombia, Ecuador, Peru, and Bolivia; in Africa; the central Zaire basin, Gabon, and the Congo Republic; and in Asia; the island of New Guinea (cited in McNeely *et al.*, 1990) (Table 1.4).

Table 1.4: Countries with the highest number of species for selected organisms.

	MAMMALS	BIRDS	AMPHIBIANS	REPTILES
1.	Indonesia ...(515)	Colombia...(1721)	Brazil...(516)	Mexico...(717)
2.	Mexico...(449)	Peru...(1701)	Colombia...(407)	Australia...(686)
3.	Brazil...(428)	Brazil...(1622)	Ecuador...(358)	Indonesia...(600)
4.	Zaire...(409)	Indonesia...(1519)	Mexico...(282)	Brazil...(467)
5.	China...(394)	Ecuador...(1447)	Indonesia...(270)	India...(453)
6.	Peru...(361)	Venezuela...(1275)	China...(265)	Colombia...(383)
7.	Colombia...(359)	Bolivia...(1250)	Peru...(251)	Ecuador...(345)
8.	India...(350)	India...(1200)	Zaire...(216)	Peru...(297)
9.	Uganda...(311)	Malaysia...(1200)	U.S.A...(205)	Malaysia...(294)
10.	Tanzania...(310)	China...(1195)	Venezuela Australia...(197)	Thailand Papua NG...(282)

Source: various sources adopted from McNeely *et al.*, 1990, 89.

Studies have also estimated that these forests contain the largest numbers of threatened species (McNeely *et al.*, 1990). In developing countries there is often a conflict between preserving the range of natural habitats and wild species and the demands for economic growth and development. Table 1.5 presents estimates of habitat loss in selected developing countries in Sub-Saharan Africa.

The challenge then is to develop viable and realistic conservation solutions that are agreeable to all parties but that also take into account the needs of the developing world, especially those of local communities. This will involve measuring all the known and relevant values, economic and otherwise of wildlife.

Table 1.5: Wildlife habitat loss in selected Sub-Saharan African countries.

Country	Original wildlife habitat (sq km)	Amount remaining (sq km)	Habitat loss (%)
Angola	1,246,700	760,847	39
Botswana	585,400	257,576	56
Ethiopia	1,101,003	30,300	70
Gambia	11,300	1,243	89
Kenya	569,500	296,140	48
Madagascar	595,211	148,803	75
Rwanda	25,100	3,263	87
South Africa	1,236,500	531,695	57
Tanzania	886,200	505,134	43
Zambia	752,600	534,346	29
Zimbabwe	390,200	171,688	56

Source: IUCN/UNEP 1986, cited in McNeely, 1990.

1.3.5 Values of biodiversity

Biological resources can be valued for their utilitarian and intrinsic worth (WCED, 1987; Moran and Pearce, 1997). Identifying and quantifying all values is important in conservation. Webb *et al.* (1993, 255), argues that,

> *diversity of use, and thus diversity of value, is perhaps the best practical insurance one can bestow on any animal you want society to conserve. Without a "use", any item, wildlife included, runs the risk of being seen as "useless".*

Utilitarian values refer to the economic applications of that resource. These include direct uses such as consumption, and indirect uses such as ecological services. Biological resources may be exploited for short-term profit or maintained to produce a long-term flow of rents. Intrinsic values refer to the non-economic values of the resource, this also includes cultural and spiritual values. These values are non-tangible and difficult to measure. Therefore any analysis into the value of a given species is likely to be underestimated especially if its intrinsic values are significant.

Relevant to this study are the economic reasons and values important for conservation.[2] Total Economic Value (TEV) is a system for classifying economic values according to the type of benefits they provide (Freese, 1998). Total Economic Value can then be expressed as: TEV = Direct Use Value + Indirect Use Value + Option Value + Bequest Value + Existence Value. However, it is unlikely that there will be a simple additive relationship due to the presence of substitution and complementary effects.

The different types of values are presented in Table 1.6. Note however, that economists are not "valuing the environment" in itself. They are valuing individual human preferences for (or against) the flow of services from the environment (Pearce, 1991). To ensure resources are devoted to biodiversity conservation TEV should be maximised.

Table 1.6: Economic values for environmental goods and services.

Total Economic Value				
Use Values			**Non-Use Values**	
Direct use value	Indirect use value	Option value	Bequest value	Existence value
Outputs directly consumable	Functional benefits	Future direct and indirect values	Use and non-use value of environmental legacy	Value from knowledge of continued existence
Food, biomass, recreation, health	Flood control, storm protection nutrient cycles	Biodiversity, conserved habitats	Habitats, prevention of irreversible change	Habitats, species genetic ecosystem

Sources: Barbier, 1992a, 21; Barbier 1994, 158; Georgiou *et al.*, 1997, 24.

Economic values can be divided into direct and indirect use values and further divided into consumptive, non-consumptive and productive uses (McNeely, 1988;

[2] Also of importance (but not discussed in this study) is the ethical and moral reasons for preserving biodiversity. See Turner (1991); Turner and Pearce (1993); Callicott (1994) and Aldo Leopold "Land Ethic" cited in Tisdell (1994).

Freese, 1998). Biodiversity can be used directly as *inter alia* food, clothes, medicine, energy, tourism, and for research (IUCN, 1987). For instances, studies have estimated that wetlands fish and fuelwood (consumptive use) in Nigeria has a direct use value of US$38-59/ha (Barbier, 1992a). While tourism (non-consumptive) in Cameroon has been valued at US$19/ha (Barbier, 1992a).

Indirectly biodiversity has option, quasi-option, and existence values (Moran and Pearce, 1997). Option value refers to the decision not to use the resource now, but to preserve it for future use (Moran and Pearce, 1997). The fact that a species does not have any current value does not mean it will have no value in the future (Table 1.7). Option value therefore is a means of assigning a value to risk aversion in the face of uncertainty and asymmetric information (McNeely, 1988; Heal, 1998). That is, it is an insurance premium as no-one knows what significant uses a species may have in the future (Moran and Pearce, 1997).

Quasi-option value refers to the choice of preserving the resource until future information becomes available and if the information suggests a low value for that resource there is then the option of destroying it (Heal, 1998). In short, the informational value of biodiversity is of increasing importance (Swanson, 1994a). This information has accumulated through the evolutionary process, but once a species becomes extinct this information is lost. Option and quasi-option values help preserve this information.

Existence value is the value attached to a species or habitat by individuals just because it exists, even if they may never use it (McNeely, 1988). For example, the kiwi is of national significance to many New Zealanders (Moyle, *in press*).

Biodiversity also has value for the many roles it has in the functioning of ecological services. Many are essential for human survival and support economic activity. For example, forests provide carbon storage (Brown *et al.*, 1993), maintain the water cycle, and regulate the climate (McNeely, 1988). Forests are also a habitat for wild species. It has been estimated that the carbon storage value of tropical forest is US$1300/ha/year (Barbier, 1992a). However, existing economic valuation methods do not easily capture this indirect use value (Aylward, 1992; Moran and Pearce, 1997). Biodiversity therefore, has many different uses and values to humans.

Table1.7: Market value of recently utilised varieties.

Year of first marketing	Species	Origin	Estimated market value
1963	Rosy Periwinkle	Madagascar	$88m. p.a. (1985)
1976	Lycopers. Chmielewskii	Peru	$8m. p.a. (1986)
1981	Wild Hops	....	$15m. p.a. (1981)
1990's	Zea diploperennis (Perennial maize)	Mexico	$ billions

Source: Swanson, 1992a, 3.

In summary, valuation informs us that biodiversity is important and consequently should be used wisely and optimally. In the past biodiversity has not been accurately and correctly valued. The result is a continuing loss at biodiversity at an increasing rate. Valuation however is only a fundamental first step in conservation. According to McNeely *et al.* (1990) the second step is to determine how these species and areas can be best conserved. This is where economic incentives and disincentives have an important role both for conservation and the distribution of the benefits (McNeely, 1988).

1.3.6 Local, national and international dimensions of biodiversity

The value of biodiversity has local, national and international dimensions. For instance, locally, wild species may be valued for their consumptive uses. Whereas internationally option value may be more important. At a national level the value of biodiversity may be expressed in conservation policies such as national parks, and laws governing the use of wild species such as hunting licenses.

Globally, the value of biodiversity is expressed in the many international environmental agreements, treaties, policies, and programs. The Convention on International Trade in Endangered Species (CITES) is a widely ratified treaty. CITES recognises that the exploitation of some species is derived from uncontrolled market pressure and this may overwhelm some species (Swanson, 1994a). CITES works by identifying those species most at risk and then banning or restricting trade between member countries.[3]

Once countries have ratified CITES they must introduce appropriate and relevance domestic conservation policies (OECD, 1997). The level of investment into management and enforcement is directly related to the value that the state places on its biological resources. Hence, at the national level if biodiversity is perceived to have a low value, protection and conservation may not be a high spending priority for the government.

Local, national, and international interests are often in conflict. For example, local communities have valid economic and cultural reasons to hunt and trade in wild species. However, the state may pass laws banning hunting, thus imposing a cost on those communities. At the same time many western advanced nations may place a high preservation value on these resources and demand greater protection. Further, all nations are sovereign and they may not be willing to accept any international intervention into their internal matters, especially with regard to economic and resource issues (Pearce *et al.*, 1990). This may be exacerbated when species protection is a high cost to the state but the state is unable to recover all the benefits. What is needed is a process that takes into account the needs of all the stakeholders and that raises economic welfare at the local, national, and international levels. This will require an understanding of the distribution of costs and benefits and of the underlying reasons why individuals value biodiversity differently.

1.3.7 Underlying causes of biodiversity loss

Many international and domestic biodiversity conservation policies and programs focus on the direct threats to wildlife (Swanson, 1994a). These include poaching, and illegal trade. While these threats are of concern, it is important to understand the underlying causes (many of which are economic) of species decline. According to Moran and Pearce (1997, 83),

> *failure to invest ownership in biodiversity is widely regarded as one of the basic causes of extinction.....The result is a lack of incentive on the part of those using the resource, or using the 'base resource' to conserve biodiversity.*

[3] How CITES works, its limitations and ways to improve CITES are expanded on in Chapters 4 and 5.

Once the underlying causes are known, actions can be taken to change the behaviour of the parties. That is, creating incentives to conserve rather than over-exploit biological resources. A change in behaviour at the local level will probably generate the greatest benefits. This may include assigning local communities property rights to local wildlife and establishing legal markets. These actions allow communities to benefit from the use of local resources consumptively and/or non-consumptively. In turn, this may generate a sustainable flow of revenue over the long-run. Hence, the protection and conservation of the species becomes an attractive option. In essence, the realised value of the species is increased.

1.3.8 Indigenous Peoples/Local communities

According to Tuxill and Bright (1998, 56) from the Worldwatch Institute, 'conserving threatened species is as much a cultural as a biological endeavour.' Local communities especially those located in developing countries have close links to the natural environment (WCED, 1987). They also live in areas of the greatest biodiversity. (Table 1.8). Further, many cultures practise sustainable management of local resources (Meffe and Carroll, 1994a).

Table 1.8: Countries where high cultural and biodiversity overlap.

Highest Cultural Diversity		**Highest Biological Diversity**
	Indonesia	Colombia
	India	China
Papua New Guinea	Australia	Peru
Nigeria	Mexico	Malaysia
Cameroon	Zaire	Ecuador
	Brazil	Madagascar

Source: IUCN, 1997, 31.

Local communities and wildlife are often adversely affected by centrally planned economic development or conservation policies (WCED, 1987; Furze *et al.*, 1996) (Box 1.2). An absolute protectionist approach further weakens traditional local land tenure and resource use rights (McNeely, 1990) and reduces the TEV of protected species. This has affected local communities' relationship with the natural environment and their use of wildlife.

In many developing countries biological diversity is now treated as an open-access resource. This may result is over-exploitation. Over-exploitation threatens resources and the state may respond by increasing controls and stricter restrictions at the local level. However, it is often "outsiders" who by poaching wildlife are the main cause of over-exploitation not locals (Naughton-Treves and Sanderson, 1995). Yet local communities shoulder the costs of protectionist conservation policies.

Box 1.2: Threats to India's wildlife sanctuaries.

In India, politicians reduced the size of the Melghat Tiger Reserve by one third in 1992 to accommodate timber harvesting and dam construction. While more than 40 percent of the Narayan Sarovar Sanctuary was turned over by the Gujarat State Assembly in 1995 to mining companies eager to harvest the coal, bauxite, and limestone deposits found there. Narayan Sarovar was home to a rich assembly of wildlife, including wolves, desert cats, and the largest known population of the Indian gazelle.

Source: Kumar, 1995, 10.

There is now a trend towards a more participatory and decentralised or "bottom-up" approach to conservation and economic development (Furze *et al.*, 1996). In Article 8(j) of the Convention on Biodiversity it is recognised, *that if the knowledge, innovations, and practices of indigenous and local communities are not respected, preserved, and maintained, humanity is losing experience, knowledge, and technology for conserving biodiversity* (cited in Martinez, 1995, 95).

Local communities should be consulted, and allowed to participate in any policies and programs with respect to the use of wildlife. According to the World Bank (1992) participatory approaches offer three main advantages. First, they give planners a better understanding of local values, knowledge, and experience; second, they win community backing for project objectives and community help with local implementation; and third, they can help resolve conflicts over resource use.

1.3.9 Conservation and local communities

There are costs and benefits with any conservation program. For instance, wildlife and land returned to local communities may need several years to recovery before commercial use is viable. During this time local communities have to devote resources into conservation, which is a cost to them. External parties may however receive a disproportional share of the benefits.

Local communities will need funding and assistance to initially rectify these problems and costs and to implement ongoing conservation measures. Assistance could come from developed countries. For example, industrialised countries contribute to the Brazilian Tropical Rainforest Fund to conserve rainforests (World Bank, 1992). The Global Environment Facility (GEF) assists among other things funding for biodiversity conservation. The GEF is a scheme set up and funded by the World Bank to channel investment and technical assistance from the West to help solve environmental problems in the developing world (Brown *et al.*, 1993).

Once conservation is underway and species numbers are recovering, the sustainable use of wildlife could become self-funding and self-sustaining. Sources of revenue include international trade, tourism, and hunting licences. For long-term conservation this revenue must be returned back to the local community. Once communities can see and receive some of the benefits from conservation then this is an incentive for them to continue such practices (Brown *et al.*, 1993).

Of interest to this study is the need for research and information on local communities' use of wildlife. According to Altman *et al.* (1996), *[a]dditional information is urgently needed on indigenous participation in wildlife use.... There is an absence of comprehensive data on indigenous people's long-term use of species which makes it difficult to provide definite statements about either wildlife management or sustainability.*

In essence, it is not known whether local communities will use the resource "wisely". Property rights may offer a solution to this dilemma. By assigning property rights to local communities this will help assist in the successful and effective implementation of the above mentioned strategies.

Experience has demonstrated that equitable access to resources can provide a powerful incentive for conservation, provided that this is made possible within a structure of community or private responsibility for the continued productivity of the resource (McNeely, 1988, 67).

This could be achieved through restoring traditional land tenure arrangements and strengthening local management responsibility (McNeely, 1988; Saunier and Meganck (eds.), 1995).

1.3.10 Property rights

McNeely (1995) identified land tenure as an important issue. That is, if property rights (land tenure) are secure this offers security to the owner to use the resource in the most valuable way and receive any benefits. Property rights are an entitlement on the part of an owner to a resource or good and where the entitlement is socially enforced (Pearce, 1994). McNeely (1988, 12) argues that,

the more well-defined, secure, and exclusive are the property rights to biological resources, the more effectively can the use of these resources be allocated by markets.

National laws, regulations, and/or traditional customs usually dictate and enforce how resources can be used.

There are four types of property right regimes[4] (Bromley, 1991, 1997; Barbier, 1992a; Pearce, 1994). They are: common property, state property, private property, and non-property or open-access. Common, state, and private property regimes can all be effectively enforced, given certain conditions. Only with open-access are property rights not enforceable.

Briefly, open-access operates when there are no institutions that can enforce ownership to an individual or a group. As it is impossible to secure all the returns from

[4] Property rights are discussed in detail in Chapter 3.

using this resource there is little incentive to invest in conserving this resource (Barbier, 1992a; Bromley, 1997; Moran and Pearce, 1997). Common property resources are owned or managed by a well-defined group of people. Outsiders can be excluded from using common property resources. Many traditional and indigenous property right regimes are of this type. State property is where ownership is held by the state. Private property is where owners receive most of the benefits from utilising the resource and can exclude others from using the resources.

There are challenges with the allocation of property rights. For instance, according to Tisdell (1995, 3),

> *there is still much to be done especially in relation to shared rights. In practice many issues involved in biodiversity conservation involve shared rights.*

The challenges include equity issues, informational constraints and enforcement problems. For instance, with common property has every member of the group fair and equal access to all shared resources and benefits. Owners may lack information on the best value of the resources they own, especially if markets are thin. If information is not available or incorrect wrong decisions may be made. If property rights are ill-defined or poorly enforced this could lead to degradation, over-exploitation and possible extinction for species (OECD, 1994). These issues may need to be resolved before an effective property rights regime can be installed. Further, non-attenuated property rights do not guarantee protection and effective maintenance of biodiversity (Moran and Pearce, 1997) However, combined with commercial use such as international trade increases the value of species and in turn generates incentives for conservation.

1.3.11 Commercialisation of wildlife: trade

Internationally, there is significant demand for wildlife and wildlife products. Developing countries are major suppliers and exporters of wildlife. Wild species are harvested or captured for the international market and many are also over-exploited. This has threatened the survival of some species (Burgess, 1994). Trade bans or quantity restrictions are traditional solutions to protect wildlife and halt the decline in endangered species. For certain species such measures are warranted (Burgess, 1994). Nevertheless, many species are still illegally traded. A recent proposal is to allow legal trade and markets for wild species to operate. That is, allowing regulated international commercial trade may help in wildlife conservation. Trade creates incentives and can provide additional funds for investment in species conservation.

Swanson (1994a, 202) argues that, 'mechanism (in the sense of the creation of incentives for wildlife conservation) must be the creation of rents in the wildlife resource.' However, the use of wildlife will only supply a flow of long-term rents if there are well-defined property rights to those resources. Property rights are therefore a key determinate for the success of any sustainable use policy (Sedjo and Simpson, 1995). If property rights are weak or absent an open-access regime may exist and the resources is still over-exploited.

Finally, international trade plays a minor role in species extinction relative to other factors (Burgess, 1994). The vast majority of exploited wildlife is consumed locally (Burgess, 1994). There are however relatively few studies that have been carried out on the consumptive use of species (subsistence and domestic) in developing countries (Burgess, 1994; McNeely, 1988; see Fa *et al.*, 1994; and Fa *et al.*, 1995 for aspects of market hunting for bushmeat).

1.3.12 Summary: relevance of research

A large amount of research, discussion, debate, and policy has arisen from the issues mentioned above. Much of the research points towards adverse effects should the loss of biodiversity continue at its current rate. Many of the issues relevant to this topic are in conflict and these conflicts are often over property rights. For instance, local verses global values, conservation verses land conversion. In the past these issues were incompatible.

The challenge now is to develop new and innovative methods to correct these shortcomings and make them compatible. Since biodiversity is valuable, investment in management and conservation efforts should be judged in economic terms and this requires formal economic analysis (McNeely, 1990). Economics can offer analysis and insights into reasons for individuals' behaviour and suggest alternative policies to conserve biological resources. Research in some of the related areas is limited or incomplete which results in gaps in our knowledge and understanding. These gaps need further research before effective conservation policy can be implemented. This study will examine some of these issues. This is what makes this topic interesting.

1.4 DEFINTION OF KEY TERMS

This section defines several key terms and concepts used throughout this study. They are as follows:

Conservation: as defined by the 'World Conservation Strategy' is, 'the management of human use of the biosphere so that it may yield the greatest sustainable benefit to present generations while maintaining its potential to meet the needs and aspirations of future generations. Thus conservation is positive, embracing preservation, maintenance, sustainable utilisation, restoration, and enhancement of the natural environment' (McNeely, 1988, 195).

Conservation should not be regarded as just the maintenance of the *status quo*, zoos, and genetic seed banks can carry out this. Conservation should mean that species have the ability to continue to evolve. Thus the importance of *in-situ* efforts as a high priority (Gowdy, 1997). Conservation also has different meanings and values to different people. For example, a sports hunter and environmentalist may both believe in conservation, but would find it difficult to agree on a common definition.

Biodiversity: 'biological diversity encompasses all species of plants, animals, and mircoorganisms and the ecosystems and ecological processes of which they are parts. It is usually considered at three different levels, genetic diversity, species diversity and ecosystem diversity' (McNeely *et al.*, 1990, 17).

In this study "species diversity" will be used as the measure of biodiversity. Much international support and funds for biodiversity conservation is derived from using charismatic species (Swingland, 1993). Furthermore, current conservation efforts tend to focus or target species-specific programs (Burgess, 1994). However, it should be noted that all three forms of diversity are interrelated. Neglecting one will impact on the others, with uncertainty over the long-term effects. In essence, valid arguments can be made for placing a priority on all three levels of biodiversity.

Sustainable development: the most common and widely recognised definition is from the 'Brundtland Commission.' 'It is development that meets the needs of the present without compromising the ability of future generations to meet their own needs' (WCED, 1987). There are many other definitions of sustainable development but what is more important is creating the conditions needed to achieve this objective. That is, meeting the needs of the present generation, while conserving biological resources and environmental quality over the long-run. Other issues that deserve analysis include the debate over the substitutability of natural capital and human-made capital, time preference and the discount rate (but are not included in this study).

Sustainable utilisation: 'The use of a population or ecosystem at a rate within its capacity for renewal and in a manner compatible with conservation of the diversity and long term viability of the resource and its supporting ecosystems' (from *Caring For The Earth* cited in Favre, 1993, 882). According to Swanson (1994a, 201), 'such a strategy has much to commend itself as a conservation tool.' Freese (1998) differentiates three types of sustainability: sustainable offtake, ecological sustainability, and socioeconomic sustainability. Finally the CBD recognises the right of states to exploit on a sustainable basis their own natural resources (UNEP, 1993; Freese, 1998).

For the long-term sustainability of a species (i.e., its survival) reproductive potential is most important and this will require biological research and analysis. Utilisation however, is an economic matter. Wildlife can be used in a variety of ways, lethal and/or non-lethal. That is, wildlife has consumptive and non-consumptive uses. There are however strong ethical arguments against the consumptive use of wildlife. (but not discussed in this study). All uses have economic values and therefore require economic analysis into the best option (Favre, 1993). What is important is that the net economic benefits of using wildlife sustainably are greater than alternative uses and therefore act as incentives to invest in resources for the conservation of that species and its habitat.

However, gathering all relevant information is difficult. Altman *et al.* (1996, 91), notes that, 'the issue of sustainability is another central concern for policy makers. The question [is] what are sustainable levels of resource use?' Further, species population dynamics are poorly understood (Moran and Pearce, 1997). Urgent research is needed to indicate which resources are under or over-utilised and what resources can be exploited commercially and which need protection. Judgements will have to be made based upon what information is available. Local communities are sources of such information (Swanson, 1994a).

Local communities/indigenous peoples: In this study the two terms are used interchangeably. There are problems defining these terms (see IUCN, 1997, 27-31). One definition is given as: 'Minority ethnic groups generally characterised by distinct language, culture, history, territorial base, and self-government. These groups live in and maintain some of the earth's most fragile ecosystems' (Clay, 1994, 359). Many of these groups have valuable knowledge of resource use and practise sustainable use. 'Indigenous people have the potential to bring a particular type of expertise to [species] management' (Altman *et al.*, 1996, 89). There is a need for these groups to participate in wildlife conservation.

Over-exploitation: (over-harvesting) is, 'the taking of individuals at a higher rate than can be sustained by the natural reproductive capacity of the population being harvested. When species are protected by law, harvesting is called poaching' (McNeely *et al.*, 1990, 38).

Incentives: (for conserving biological diversity). 'An incentive is that which incites or motivates desired behaviour in government, local people, and organisations to conserve biodiversity. More broadly, an incentive is any inducement on the part of government which attempts to temporarily divert resources such as, land, labour, or capital toward conserving biodiversity, and facilitates the participation of certain groups or agents in work which will benefit biodiversity' (McNeely, 1988, 197).

Markets: 'A market is the organisation that co-ordinates the production and consumption of goods and services through voluntary transactions, based upon the free will of buyers and sellers. Any transfer of goods and services against their will (such as stealing, poaching) is not a market transaction. Therefore, if information is perfect, all the participants in market transactions gain, since sellers would not sell and buyers would not buy unless they were to gain. The market is, therefore, the organisation to co-ordinate people's activities in seeking self-interest towards increasing social economic welfare' (Hayami, 1997,198).

1.5 THE AIMS AND SCOPE OF THIS THESIS

1.5.1 Exploring the linkages between the key terms

Research has shown that there is a relationship between environmental degradation, species numbers, population, and poverty (WCED, 1987; World Bank, 1992; Todaro, 1994). Furthermore, there are strong links between wildlife use and local communities. Moreover, for some species international trade threatens their survival. There is a need to research the dynamics of these relationships and formulate solutions that eliminate poverty, reduce environmental degradation, and conserve wildlife. This will have local, national, and international perspective's.

It is also known that well-defined property rights can create the conditions for conservation. In addition, local communities have traditional methods and knowledge of sustainable management of wildlife and ecosystems. A proposed solution is to allocate property rights to local communities, establish legal markets for wildlife and allow them to benefit from the sustainable utilisation of local wild species.

1.5.2 Aims and objectives

The aims of this study are:

(1) to research whether sustainable use of wildlife is an effective conservation tool;

(2) to add to the knowledge on local communities involvement in conservation;

(3) identify issues that require further attention and research.

The long-term objective is the conservation of wildlife and increased economic welfare to local communities.

To achieve these aims will require research into wildlife uses and values. Issues covered include *inter alia* protected areas, trade bans, partnerships in conservation, sustainable use, and the role of markets in conservation. Important to a successful conservation and utilisation strategy is the creation of incentives. This will require well-defined and enforceable property right regimes. These issues can be analysed from an economic perspective. The relevant issues for this study will be discussed in the context of the legal commercialisation of wildlife and property rights.

1.5.3 Limitations of study

This study assumes the decline in biodiversity is a result of over-exploitation and habitat conversion. However there are other paths (not discussed) to species extinction (refer, McNeely, 1990; Brown *et al.*, 1993; Swanson, 1994a). The thesis will largely refer to the plight of wild species, rather than genetic or ecosystem diversity. Furthermore, conservation efforts and success will depend on many other variables, *inter alia* human population growth. This study largely researches property rights and does not cover these other issues. This is because evidence suggest that correcting the underlying causes of biodiversity loss such as a lack of or poorly defined property rights may help solve other problems associated with the loss of biodiversity. For instance, by increasing rural incomes through secure property rights, the standard of living may improve and this may cause population growth to decline. This in turn will relieve pressure to convert natural habitat and over-exploit wild species.

1.6 WHY THIS TOPIC DESERVES ECONOMIC ANALYSIS

Economists generally consider an economy's "endowment" of natural resources as a potential source of wealth (Barbier, 1993). Wild species, as biological resources are part of a country's natural capital. Therefore decisions on whether to protect, use, or deplete domestic natural resources will require economic analysis. According to Swanson, (1994a, 3):

> *The fundamental nature of the biodiversity problem is a human choice problem, where human societies are systematically reshaping the biosphere across the face of the earth. Economic analysis lies at the core of the examination of this problem because it is a study of human choices, especially those choices concerning the human allocation of resources.*

Past concerns over species loss and habitat destruction resulted in a policy of absolute protection. Accordingly large areas were thought best for preserving wildlife as they contain and support large numbers and varieties of species (Leader-Williams and Albon, 1988; McNeely, 1988). Developing countries contain large remaining areas of natural habitat. However, developing countries have limited resources to maintain such areas. Further, economic development demands are pressurising states to make use of existing protected areas by converting them into other uses (Swanson, 1991). To ensure that these habitats remain in a natural state will require a demonstration of their value (Swanson, 1991).

1.6.1 Market and policy failures

The current situation can be partially blamed on market and policy failures (Moran and Pearce, 1997). Market failures include incorrect pricing, imperfect information, and ill-defined property rights (Burgess, 1994). Policy failures occur when public policy to correct market failures over-or-under correct the problem (Burgess, 1994). These include subsidises for alternative land use, such as agriculture (Box 1.3).

Box 1.3: Impact of national policies: Brazil.

Since the 1960's Brazilian governments have introduced a series of economic growth and development policies. The aim was to spur investment in the interior of the country. Roads were built deep into the nation's forests, investors were given generous tax holidays, offered credit with negative interest rates, and given other subsidises to "improve" the interior. The result was millions of hectares (ha) of forest converted to cattle ranches that otherwise would not have been profitable. Further, widespread cattle disease has hindered unprocessed meat sales to the USA and Europe. By 1980, 72 percent of the forest conversion detected by satellite was due to cattle pasture. After 1990, four times as much deforestation came from subsidised ranches as from nonsubsidised ranches, and about a quarter of the pasture was already abandoned.

In addition, property right titles are only granted to colonists, shifted cultivators, and displaced peasants who improve the land by clearing it of forest cover. For example, in the 1970's the government started to sponsor smallholder settlement in Rondonia and later Acre in southern Amazonia. In 1975 only 1200 km^2 of the state's forests had been cleared, but by 1985 it was 28,000 km^2. Brazil lost more than valuable forest. It is estimated that a hectare of Brazilian

Amazonia forest contains between 87-300 different species of trees. Further, should deforestation continue at present rates until the year 2000, and then halted completely about 15 percent of plant species will be lost, together with 69 percent of bird species. In addition, by 1988, the fiscal cost of all 470 subsidised ranches was US$2.5 billion. However, in the 1990's there has been a shift in government policy.

Source: Adopted from WCED, 1987; McNeely, 1988; Sandler, 1993; Mendelsohn, 1994; Barrow, 1995; Myers, 1995; Abramovitz, 1998.

Barriers to trade and inadequate enforcement of regulations are also classified as policy failures. The result of these failures is that there are economic gains to be made from poaching and the illegal trade of wild species. Over-exploitation is a reality, threatening the survival and possible extinction of some species. One solution is to increase laws and regulations. However, economic based incentives offer a viable and effective alternative.

Economic tools should be used to correct these failures. According to McNeely (1988, vii),

> *economic inducements are likely to prove the most effective measures for converting the over-exploitation to sustainable use of biological resources.*

This can include compensating or creating positive incentives that alter the behaviour of those individuals affected. For instance, by changing the harvester's (supply) and/or the consumer's (demand) behaviour to show that it is in their best interests to maintain the species. However, in deciding how to "use" wildlife there is an opportunity cost.

> *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 and the individual's preference for receiving the returns today rather than in the future* (Burgess, 1994, 136).

To maintain an adequate level of wildlife requires investment in land, management and other resources. That is, conservation incurs costs (Moran and Pearce, 1997). To ensure investment is forthcoming the net returns from conservation must exceed that from alternative use of the habitat and species. 'In the matter of conservation wildlife must yield a rate of return that is compatible to or greater than that of other forms of wealth' (Barbier, 1993, 2). Yet those who benefit the most from conservation (the global community) do not pay the full costs, while those who do pay (the local community) receive inadequate benefits. This inequality needs to be reduced.

1.7 METHODOLOGY

At the core of this study is the conservation of biodiversity. This study assumes that humans value and utilise wildlife for their own interests. It also assumes that conservation is preferable to extinction. That is, with uncertainty and irreversibility, as well as option and existence values it is better to try and maintain rather than over-exploit a species to extinction. In consequence, this study follows a methodology of first reviewing and discussing traditional conservation strategies. From this foundation, new innovations are discussed. There is a large and diverse body of literature on biodiversity, conservation, and the use of wildlife. Reference to case studies will be made at each stage from the available literature.

In the past the different disciplines and sciences approached these topics separately and individually, formulating specialised plans of action. To understand and analyse this topic now requires a multi-disciplinary approach. Concepts and methods from biology, ecology, anthropology, other natural and social sciences will all contribute to this study. However, the main method of inquiry will be on economic aspects of wildlife conservation. Reference will be made to microeconomic theory, specifically how institutions and markets can lead to efficient outcomes, and to ecological economics which emphases the importance of an inter-disciplinary approach to conservation. Market and policy failures, market-based incentives, the advantages of markets, and property right regimes will be all discussed.

The topic of this study is relevant to current conservation literature. Conservation of wildlife has evolved through a series of stages. In the past, biology and ecology argued that the best way to preserve wildlife was to protect it. Accordingly large protected areas and trade bans were thought best at minimising the risk of species extinction. This has not been as successful as first thought (McNeely *et al.*, 1990;

Possiel *et al.*, 1995). Economic studies that emphasises sustainable utilisation of wildlife is gaining acceptance as an alternative and effective conservation method (Burgess, 1994; Swanson, 1994a; Martinez, 1995).

1.8 CONCLUSION

For conservation to be successful now requires an analysis of many issues previously not thought relevant. One such issue is sustainable utilisation. To be effective this strategy will require local community involvement. This will require research and analysis of local communities. There appears to be limited information in this area. In the past the literature did not appear to acknowledge local communities.

Economic analysis on the values, uses, and impact that wildlife has on local communities is lacking. Research and analysis on property rights will also contribute to the effectiveness and efficiency of sustainable use strategies. It is intended that this thesis will contribute to the current thinking on sustainable use and conservation and add to the knowledge on the economic development of local communities. The sections outlined in this chapter will be expanded upon in the following chapters.

2

Literature Review

One of the important global challenges we currently face is determining the role of conservation in economic development **(Swanson and Barbier (eds.), 1992, ix).**

2.1 INTRODUCTION

The aim of this chapter is to review literature relevant to this topic. It is a comprehensive but not an exhaustive review. Section 2 outlines perceived threats to biodiversity identified in the literature and explains its economic dimension. In Section 3 existing literature on conservation strategies are reviewed and discussed. A critique of these traditional measures is presented. Recent innovative ideas in conservation: sustainable use and partnerships in conservation are discussed in Section 4. In addition, property right concepts and ecological-economic approaches are outlined. Section 5 presents theories of development and explores the relationship between conservation, economic development, and local communities. Of relevance to this study is the literature on sustainable development and sustainability. This chapter also identifies gaps in the literature that will be explored in detail in further chapters.

2.2 LITERATURE ON BIODIVERSITY

2.2.1 Biodiversity loss

There is general agreement in the literature that the natural environment and biodiversity are undergoing changes and losses, some evolutionary, and some caused by *Homo sapiens* (WCED, 1987; McNeely *et al.*, 1990; Meffe and Carroll, 1994a; Orians, 1994; Myers, 1994; Perrings *et al.*, (eds.), 1995; UNEP, 1997; Guruswamy and McNeely, 1998; Worldwatch Institute, 1998). Evidence cited in the literature now suggests that biodiversity (in its three forms) is of significant value and of critical importance for the health and proper functioning of ecosystems (IUCN, 1987; McNeely, 1988; McNeely *et al.*, 1990; OECD, 1991; Randall, 1991; UNEP, 1993; Blackmore and Reddish, 1996; Tobey, 1996). This has raised concerns as to the future well-being of the human race should the current rate of loss continue. For instance, according to the OECD (1997, 11), 'the decline in biodiversity levels, and implications for the continued habitability of the planet, are widely-recognised phenomena.'

In Perrings *et al.* (eds.) (1995) natural and social scientists discuss the problem of biodiversity loss with regard to the future of humankind. They concluded that the fundamental goal of biodiversity conservation should be the protection of ecosystem resilience.

> *What is becoming clear, though, is that the sustainability of economic development implies ecological stabilisation: the maintenance of the productive potential of ecosystems supplying essential ecological services either by the containment of stress levels or by the promotion of ecosystem resilience through biodiversity conservation* (Perrings *et al.*, 1995b, 308).

Further, they add that there are still many unanswered questions such as the uncertainty and implications of crossing ecosystem resilience thresholds, and the need for a clearer ecological-economic approach to biodiversity conservation (Perrings *et al.*, 1995b).

What is of concern is that the current rate of biodiversity loss appears to be much higher than the natural rate. This conclusion is supported by extensive studies and research carried out on natural habitat loss and extinction rates. Losses are summarised in McNeely *et al.* (1990) for the status of threatened species, estimated deforestation rates, and wildlife habitat loss (Table 1.5). Studies also include the endangered species lists presented in the *IUCN Red List of Threatened Species* published by the World

Conservation Monitoring Centre (WCMC)[1] (McNeely *et al.*, 1990; UNEP, 1993; Tuxill and Bright, 1998) (Table 1.1 and 1.2). Species are classified under one of eight IUCN threat categories (Box 2.1 and Table 2.1).

Box 2.1: IUCN Red Data Book categories.

Extinct (Ex): species for which there have been no definite reports for 50 years.

Endangered (E): species in danger of extinction and whose survival is considered unlikely as long as the underlying causes continue.

Vulnerable (V): species that are thought likely to become endangered unless the adverse factors that threaten them can be removed.

Rare (R): species at risk through occurring in low numbers, or which have a very restricted range, or are sparsely distributed, but are not in present danger.

Indeterminate (I): species known to be Endangered, Vulnerable, or Rare, but where information is insufficient to know which of these categories is appropriate.

Insufficiently Known (K): species suspected of being eligible for one or other of the preceding categories, but where lack of information makes confirmation impossible.

Threatened (T): species comprising several subspecies that are threatened in different ways and are thus placed in different categories.

Commercially Threatened (CT): species whose survival is threatened by commercial exploitation.

Source: IUCN cited in Simon, 1995, 167.

[1] For current up-to-date lists see IUCN Red List web page: http//www.iucn.org/themes/ssc/iucnredlists/redlist.htm

Table 2.1: Status of threatened species late 1980's.

	Ex	E	V	R	I	Total Globally Threatened Taxa
Plants	384	3325	3022	6749	5598	19078
Fish	23	81	135	83	21	343
Amphibians	2	9	9	20	10	50
Reptiles	21	37	39	41	32	170
Invertebrates	98	221	234	188	614	1355
Birds	113	111	67	122	624	1037
Mammals	83	172	141	37	64	497

Key: Ex =Extinct (post –1600), E = Endangered, V = Vulnerable, R = Rare, I = Indeterminate.
Source: Reid and Miller, 1989; WCMC, unpublished data, cited in McNeely *et al.*, 1990, 42.

In many countries, government and non-government organisations also produce publications and information on indigenous and endemic endangered species. For instance, the Department of Conservation (DoC) in New Zealand have recently published a guide, *Plants of National Conservation Concern in Wellington Conservancy* on nationally threatened plants in the Wellington region (Carman, 1998).

Simon (1995) also provides information on different habitats and species. Vulnerable rainforests, grasslands, wetlands, islands, deserts, mountains, and Antarctica are all described. Characteristics of flora and fauna and threats to specific habitats located around the world are identified. Specific endangered species, including mammals, birds, reptiles, invertebrates, and plants are presented to support the thesis of intensive species loss.

2.2.2 Threats to biodiversity

McNeely (1990); McNeely *et al.* (1990); Brown *et al.* (1993); Burgess (1994); Meffe and Carroll (eds.) (1994); and Blackmore and Reddish (1996) identify or outline the major threats to biodiversity. These range from the direct threats such as introduced species and over-harvesting. Indirect threats include the effects of pollution, climate change, increase in human population, and habitat alteration.[2] While all threats are

[2] It is interesting to note that in Zimbabwe at the turn of the century the elephant population stood at 4,000 while the human population was just 5 percent of its current level. Elephants now number in the mid 50,000 (Sharp, 1997).

important and require research it is over-harvesting and habitat alteration that is the primary focus of this study. To this end the economic reasons for biodiversity loss will be analysed. Perrings *et al.* (1995a) argues that these decisions are made as a result of underlying causes that then manifest into the threats mentioned above. Biodiversity loss is explained as:

> *Most loss of biodiversity is due to the independent decisions of the billions of individuals users of environmental resources, and its underlying causes are to be found in the parameters within which those decisions have been made. These include the objectives that motivate decisions, the preferences that lie behind the demand for goods and services, the property rights that define individual endowments, the set of relative prices that determine the market opportunities associated with those endowments, and the cultural, religious, institutional and legal restrictions on individual behaviour that prescribe the range of admissible actions* (Perrings et al., 1995a, 13).

In essence, individuals exploit (sustainably and unsustainably) biodiversity for their own self-interests and these decisions are most probably rationale given the particular circumstances in which they are made. For example, over-exploitation may occur when there are no controls or when controls breakdown that restrict the harvesting of a wild species. In consequence, the harvesters is acting rationally and in their own self-interest if they decide to harvest the resource as quickly as possible.

This is a similar argument to that proposed by Swanson (1994a; 1995b). Swanson has analysed biodiversity loss and suggested several reasons for its decline. These include the expansion of the human niche, the systematic reallocation of basic resources away from other species, and the underdevelopment of segments of human societies (Swanson, 1994a; 1995b). Underlying these processes is human management and institutions.

> *It [this book] concerns the economic incentives that drive the human-made extinction process. It concerns the institutions (primarily domestic) that exist at present that do not adequately manage this process. Most importantly, it concerns the institutions (primarily international) that do not exist at present, but are necessary for the adequate management of biodiversity* (Swanson, 1994a, 2).

Furthermore, the unnaturally high extinction rates and loss of biodiversity has been attributed to traditional economic development. That is, the processes by which economic, social, cultural, and political institutions change and develop into modern economies. One reason given for the decline is that past economic development and the environment were seen as separate issues (World Bank, 1992). For instance, many developing countries have and are converting, extracting, and exploiting natural resources in response to poverty, population, and development pressures. Species are affected, with many endangered and threatened with extinction. Swanson (1995b) explains the nature of the extinction process. That is, human development occurs because humans' benefit from making the biological environment more uniform. Consequently, the global biodiversity portfolio is narrowed to a few specialised species (Swanson, 1995b).

One reason for this is that developing countries have undervalued the natural environment. For example, tropical rainforests are cleared for the commercial value of the timber, minerals and other market commodities that lie below the soils or for conversion into livestock ranches and roads. Rainforests were not valued for their use and value as habitat for species preservation, carbon sequestration, food production, and medicinal products (MacKerron and Cogan, 1993). Simply there are no markets, or markets are incomplete for many of the above mentioned environmental goods and services.

Further, perverse economic incentives have helped deplete biodiversity. Perverse incentives include subsidies, price support, and tax breaks. For example, European and North American nations give development aid to subsidise cattle production in developing countries. In consequence, livestock rapidly replaces wildlife as habitat is converted. McNeely (1988) outlines the problem of perverse incentives and the effects they have on biodiversity. He also discusses incentives at the community, national, and international level that promote the conservation of biodiversity.

Recent literature now emphasises that economic development, conservation, and the well-being of the natural environment are interrelated (WCED, 1987; McNeely *et al.*, 1990; Dixon *et al.*, 1992; World Bank, 1992; Pearce, 1994; Todaro, 1994; Weaver *et al.*, 1997).

> *Since the value of conserving biological resources can be considerable, conservation should be seen as a form of economic development. And since biological resources have economic value, investment in conservation should be judged in economic terms* (McNeely *et al.*, 1990, 27).

In tandem with this is the need to place economic values on all goods and services provided by the environment (McNeely, 1988).

In short, much of the literature emphases the underlying causes and institutional reasons that explain why individuals decide to over-harvest and/or alter habitats that then threatens biodiversity. Hence, the need to analysis these threats, especially institutional failure and generate solutions.

2.2.3 Underlying factors explaining biodiversity loss: economic

McNeely (1990) and Barrett (1993) identify the destruction of ecosystems as the biggest threat to biodiversity. Habitat destruction is largely the result of land-use conversion (Swanson, 1994a). Here humans and wildlife find themselves in competition for base resources, e.g., land. The decision to convert is usually determined by the rent capacity of the land after conversion to typically specialised agriculture and not its natural state value (McNeely *et al.*, 1990). Understanding why these decisions are made will help in conservation and economic development strategies.

Swanson (1994a) discusses this in detail. The key point being if there is a higher rate of return from converting the land (from its natural state) rather than the returns generated from the use (direct and indirect) of wildlife the land will be converted. Often all the values of the species are not known hence the natural state returns are perceived as less than the returns from conversion. TEV is not maximised and the decision to convert is more likely. One reason why conversion takes place is the problem of asymmetric information which leads to market and policy failures (Hayami, 1997). The result is accelerated loss of natural habitat and subsequent decline in wildlife populations.

In this study it is the underlying economic reasons influencing the decisions of individuals causing the decline in wildlife that are important. These include property rights, market opportunities, and cultural, legal, and institutional factors. According to Swanson (1994a, 1995a) these factors will influence decisions whether or not to invest in biological resources. A lack of understanding the underlying causes (why there is under/dis investment) is one of the greatest obstacles to implementing effective conservation policies.

> *It [the economic model of extinction] demonstrates that the different types of endangerment (over-exploitation, conversions, diversity losses) are merely different routes to the same ultimate result, driven by the same fundamental forces. This indicates that the policies for addressing all forms of endangerment, extinction, and diversity losses must be created in a fashion that will address this more fundamental problem. These reforms will often require policies much different from those that have been created in order to address the proximate (as opposed to the fundamental) causes of endangerment* (Swanson, 1994b, 801).

In summary, both the problems and solutions to biodiversity loss are built on economic foundations (McNeely *et al.*, 1990). Economic theory can be used to empirically analyse the variability and effectiveness of conservation strategies. Further, the use of economic instruments to slowdown biodiversity loss is now emphasised in the literature (McNeely, 1988; Pearce, 1991; Swanson, 1994a). Economics (properly interpreted) provides an array of novel tools in the defence of conservation (Pearce, 1991). Understanding these causes may also help in implementing economic development strategies. However, economics not properly interpreted, or implemented, can just as effectively contribute to the extinction of species.

2.2.4 Paths to extinction

Brown *et al.* (1993) and Swanson (1994a) discuss the fundamental causes of biodiversity decline and species extinction. They outline three paths towards species extinction. They are: (1) under-investment in conservation; (2) dis-investment in conservation; and (3) land-use competition for habitat. Brown *et al.* (1993, 37) add, 'all three routes to extinction have the same ultimate result (changed land use) and the same underlying cause (perceived investment worth).' Further, Swanson (1995b) identifies, "development path approach", "institutional failure", "inappropriate policies", and "development induced decline" as reasons to explain the decline in biodiversity.

Swanson has revised and added to the theory of extinction. He argues that the nature of extinction has changed since Clark's extinction model. The Clark model on the economic of species extinction evolved from the Gordon fisheries model (Swanson, 1994b). The Clark model demonstrates the conditions under which a natural resource is over-exploited to an extent incompatible with the capacity of the resource to regenerate itself (Swanson, 1994b). These conditions are open-access, a relatively high price to cost ratio of harvesting the resources, and a relatively low growth rate of the resource (Swanson, 1994b).

Swanson instead chooses to emphasise the effects of competing land uses and national sovereignty over resources into the model as factors to explain species extinction (Swanson, 1994a). Further, the concern has shifted from individual well known species (as used in the Gordon-Clark models) to a much broader concern to include the loss of unknown species, or what is categorised as "biodiversity losses" (Swanson, 1994a). Swanson also argues that a species survival will depend on its investment worthiness.

> *If a society views a species as sufficiently investment-worthy, then stocks of that species will be maintained...If the species is not viewed as relatively investment-worthy, then it will suffer disinvestment* (Swanson, 1994a, 11).

Further,

> *none of the 80-100 heavily utilised species listed in the Food and Agriculture Organisation (FAO) tables on world food production are in any danger of immediate extinction* (Swanson, 1992a, 4).

In summary, the status of the Earth's biological resources matters. Its maintenance is important. Should the extinction of species in different habitats continue this might degrade and damage the life-support systems upon which human survival is based. Concern over habitat loss and the decline in some wildlife numbers has led to actions being taken to preserve remaining biodiversity. The question and challenge then is to decide which species to exploit so to improve human welfare and which species to maintain?

2.3 TRADITIONAL CONSERVATION SOLUTIONS/ STRATEGIES

Humans derive utility from the instrumental value of biological resources (Table 2.2). Norgaard (1994, 456) defines utility as, 'personnel satisfaction of some economic desire, or the want-satisfying power of goods.' In addition, intrinsic value has importance for humans.

Table 2.2: Four categories of the instrumental value of biodiversity.

Category	Examples
Goods	Food, fuel, fibre, medicine
Services	Pollination, recycling, nitrogen fixation, homeostatic regulation
Information	Genetic engineering, applied biology, pure science
Psycho-spiritual	Aesthetic beauty, religious awe, scientific knowledge

Source: Callicott, 1994, 25.

These values have resulted in both the exploitation of natural resources and also the implementation of strategies for the protection and conservation of biodiversity. Traditional solutions include restrictions on the trade in endangered species and the creation of protected areas. However, these approaches have not been as successful as intended. Indeed many researchers now criticise the traditional approaches. For instance, Swanson and Barbier (eds.) (1992) and others in the book argue for the sustainable utilisations of the wilds. Martin (1997b) and SACWM (1997) present strong arguments against CITES. Freese (1998) identifies the conditions under which consumptive and commercial use of wild species can promote the conservation of biological diversity.

2.3.1 Conservation: what is being protected?

Conservation efforts have traditionally focused on creating protected areas to preserve wildlife. Protected areas range across all types of habitats (see Simon, 1995). The IUCN has developed a system of classifying protected areas based on certain conservation objectives (Table 4.3).

Although there are a large number of wild species (many of which are threatened) most of the protection and funding has been directed at large charismatic fauna (Soulé, 1991; Metrick and Weitzman, 1996). Examples include: the African elephant *Loxodonta africana*, black rhinoceros *Diceros bicornis*, and tigers *Panthera tigris* (Pearce, 1991; Soulé, 1991; Simon, 1995; Blackmore and Reddish, 1996). Many species that are protected are not however in danger of extinction or threatened. In addition, biodiversity "hot spots", have been identified as priorities for conservation (Myers, 1994; Holmes, 1995) (Table 1.3).

A paper by Metrick and Weitzman (1996), identified the variables that influence which species are listed as endangered and receive funding for protection by the United States Fish and Wildlife Service (USFWS). They concluded that certain characteristics rather than the species actual endangered status were determining factors.

> *The evidence indicates that we pay more attention to species in the degree to which they are perceived to resemble us in size and characteristics* (Metrick and Weitzman, 1996, 15).

In a later paper, Metrick and Weitzman (1998) developed a model to determine basic priorities for maintaining or increasing biodiversity under a limited budget constraint. In this model species are ranked for conservation priority based on four criterions. These are as follows:

(i) direct utility from the species D_i

(ii) indirect utility from the species genetic diversity i.e., its distinctiveness U_i

(iii) enhanced survivability of the species following a conservation plan $\triangle P_i$

(iv) the cost for improving survivability C_i

The priority ranking is based on the following equation:

$$R_i = [D_i + U_i]\,(\triangle P_i / C_i)$$

However, they also caution that the "real world" is more than a match for any model and that actual conservation behaviour[3] may not reflect a reasoned cost-benefit calculation (Metrick and Weitzman, 1998). The model offers an enhanced framework and guidelines for scientifically choosing which endangered species to allocate funds for protection and conservation. This has relevance to conservation as resources and budgets are limited and species should be prioritised depending on their endangered status not their resemblance to humans.

Nevertheless, by bringing attention to the plight of these charismatic or "flagship" species, there is a multiplier effect in protecting other "less charismatic" species. This is called the umbrella effect (Moyle, pers comm.). Using the tiger as an example, according to John Robinson,

> *If you protect a significant population of tigers [or other charismatic species] you're going to have to protect a significant chunk of dry forest in India, and that's also important* (cited in Holmes, 1995, 36).

Historically biologists argued that preserving genetic diversity was the key to a species survival (Schmidt, 1996). This view has been challenged. Some biologists are now arguing that to save a species the emphasis must be ecologically based, such as preserving ecosystems (Perrings *et al.*, 1995b; Slocombe, 1995; Lewin, 1996; Schmidt, 1996). Tisdell (1995) also argues that the approach of saving individual species is too narrowly focused. Rather, the focus should be on the system as a whole. This broad focus should ensure that genetic, species, and ecosystem diversity would all be conserved. Simon (1995) also argues for broader land-use strategies involving complete ecosystems, as opposed to an individual species focus. Holmes (1995) details several advantages to conserving habitat rather than individual species. Further,

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[3] Refer to Metrick and Weitzman (1996).

conservationists are no longer concerned with the extinction of specific species alone, but with the loss of biological diversity generally (Barrett, 1993; Perrings *et al.*, 1995a).

In summary, there is debate in the literature on what to conserve and the options and methods to conserve those resources. However there appears to be a shift towards conservation encompassing a broader ecosystem strategy. Leader-William and Albon (1988), caution however that due to budget constraints, increasing the size of habitat under protection is difficult and may be impractical. In addition, Wells *et al.* (1992) argue that traditional conservation places hardships on local communities. These issues and others are expanded and reviewed in the following sections.

2.3.2 Protected areas: size and spending

There is concern that protected areas are now inadequate (in size, management, and budget) to protect and preserve wildlife (Burgess, 1994). Resources have to be allocated to protecting these areas and this incurs costs at the national and local level. The World Bank estimated that spending on conservation-related activities amount between 0.01 and 0.05 percent of GDP in developing countries and about 0.04 percent in developed countries (World Bank, 1992). Developing countries in addition often lack the means (financially and politically) to successfully manage these areas. The result is poor management and weak enforcement.

For instance, Leader-Williams and Albon (1988) found a direct relationship between spending on protection and the change in rhino numbers. However, no protected areas were receiving sufficient funding to avert decline. Dublin *et al.* (1995) report that three-quarters of surveyed areas were allocating less than US\$5 km^2/yr, only one area was spending more than US\$100 km^2/yr in 1993.

2.3.3 Trade restrictions/bans: CITES

An overview of CITES is given by Favre (1993); Hemley (1995); and Wijnstekers (1995). Hemley and Favre outline future improvements for CITES for it to be effective. However, the argument against trade bans is increasing in the literature.

> *Some of the existing schemes for the international regulation of endangered species is ill-conceived...international policies concerning species decline usually focus on the proximate causes of their destruction such as poaching or unmanaged exploitation, without asking why such phenomena occur* (Swanson, 1994a, 11).

Brown *et al.* (1993); Burgess (1994); Swanson (1994a) and OECD (1997) all argue against trade bans.

> *Wildlife trade policies are often implemented with little analysis of their environmental implications nor their economic costs* (Burgess, 1994, 148).

For example, a CITES Appendix I listing effectively bans international trade in certain species. The ban usually applies to all populations. This may halt the decline caused by over-exploitation. However, it also reduces the species economic value. In consequence, the incentive to invest in that resource is destroyed and disinvestment is possible (Brown *et al.*, 1993). For instance, states that have invested in conservation and have healthy wild populations are not allowed to trade, even though trade could be sustainable and a high foreign exchange earner. In essence, these states are penalised, and nor are they compensated. This could result in a reduced conservation budget (Dublin *et al.*, 1995). Moreover, the threatened species is then shifted along the other paths towards extinction as outlined by Swanson (1994a).

Barbier *et al.* (1990) present evidence that attempts to regulate the international trade in African elephant ivory have been unsuccessful. They argue for a limited trade in elephant ivory. Simmons and Kreuter (1990, 29) suggest that 'the solution to saving the African elephant lies not in banning ivory trade but in applying the successful elephant conservation policies of south African nations.' Barnes (1996) has analysed the changes in the economic use value of elephants in Botswana caused by the international trade ban. Barnes concluded that the ban will probably not led to the long-term survival of Botswana's elephant populations.

Dublin *et al.* (1995) researched the impact and the working of the CITES ban on trade in ivory. The report concluded that the answer as to whether the ban has had a major effect remained inconclusive. They found that there is still an illegal trade in ivory in most of the African range states and poaching is still a problem with the African elephant still endangered (Dublin *et al.*, 1995) (see Appendix 1). The survey pointed out other factors that influence the illegal killing of elephants. These factors are low and decreasing wildlife conservation budgets, poor information gathering, poor government accountability, weakness in national legislation and judiciary, continuing markets, rhino poaching, and illegal killing for subsistence use and in the protection of life and property.

2.3.4 Traditional conservation strategies: implications for local communities

As stated in Chapter 1 the establishment of protected areas and trade bans has implications for local communities. Survival for a large percentage of these people depends on exploiting the local natural environment. Many of these communities have been adversely affected by conservation measures. Often they have been displaced and/or stopped from using biological resources that they had traditional utilised even when the use was sustainable. In addition, they receive little financial compensation for the losses and costs suffered if they are forced from their tradition lands that are then turned into protected areas (Leader-Williams and Albon, 1988; Burgess, 1994). Wells *et al.* (1992) also argue that creating protected areas has imposed hardships on local communities. Consequently, species in protected areas are now at risk from illegal over-exploitation by local communities. Recent literature now emphases actively involving local communities in conservation management and that they receive a share of the benefits from conservation.

2.4 RECENT INNOVATIONS AND IDEAS FOR CONSERVATION

Traditional strategies although having some effect only treat the symptoms of the problem. As stated previously, a major criticism in the literature is that the traditional approaches reduce the economic value of biological resources. This destroys incentives for investment. For long-term effective conservation the underlying causes must be tackled. This must include the economic reasons for the decline in biodiversity.

In the literature there are suggestions to improve the existing measures. Solutions offered include the sustainable use of wildlife (Barbier *et al.*, 1990; Webb, 1991; Swanson and Barbier (eds.), 1992; Freese, 1998). The participation of local people in protected area management is also emphasised (Barbier, 1992b; Wells *et al.*, 1992; Furze *et al.*, 1996).

2.4.1 Sustainable use

The literature on sustainable use is relatively recent. However it is gaining formal recognition as a legitimate and potentially powerful conservation tool (Webb, 1991; Swanson and Barbier (eds.) 1992; Freese, 1998). The IUCN- The World Conservation Union supports sustainable use. Recent CITES meetings also reflect a shift towards sustainable use as reported in the journal *Oryx* (Pendry, 1997).

Much of the literature has focused on single species such as large game animals rather than biodiversity as a whole. Large game animals are charismatic, they have both use and non-use values, and may be a "keystone" species. In consequence, a large amount of research and studies have been carried out on these species. For these reasons then large mammals are typically used in the literature as examples of sustainable use.

Barbier *et al.* (1990) wrote one of the first books on this topic. It dealt with the sustainable management of elephants and ivory trade from an economic perspective. Proposals to the CITES CoP's (Conference of the Parties) from some southern African states on the sustainable use of their elephant populations have added to the literature. This is reinforced by sustainable use proposals for various species from other countries. This includes the sustainable use of *inter alia* whales, rhinoceros, and hawksbill turtles from Norway, Japan, South Africa, and Cuba (ACSUG, 1997; Pendry, 1997). These proposals require sound scientific research and data to back the proposals. This has helped increase the knowledge of the species they would like to utilise. That is, the burden-of-proof is to prove that sustainable use is in the best interest for that species.

Further, an OECD report on the effectiveness of trade measures used by CITES concluded that other innovative ideas and incentives are needed to conserve wildlife (OECD, 1997). Burgess (1994) in a report for the OECD also supports the idea of allowing a limited amount of legal trade as long as the species population can be sustainably managed. Swanson (1994a) outlines the argument for a constructive wildlife trade regime based on use and investment rather than a total trade ban. Such a strategy will require investment in management and resources as well as changes in existing practises. Investment will require some form of property right regime. Furthermore, the involvement of local communities is also of importance.

Indigenous Peoples' use of wild species is gaining acceptance and recognition. This includes both subsistence and commercial uses. Freeman (1998) discusses the successful management of polar bears by the Inuit. Webb and Manolis (1993) and

Webb *et al.* (1996) investigates the advantages and disadvantages of the commercial use of crocodiles by Aboriginal people in the Northern Territory, Australia, and the links to species recovery. In summary, a sustainable wildlife utilisation strategy is now seen as important for successful conservation. Further, partnerships between local communities and protected area management is also gaining recognition.

2.4.2 Protected area partnerships

The literature on conservation and local communities now approaches this issue emphasising integration. Wells *et al.* (1992) links the need of local communities with protected area management. The social dimension of local level development, conservation and protected areas is emphasised and explored in Furze *et al.* (1996). McNeely (ed.) (1995) discusses partnerships between protected area managers and other sectors of society.

McNeely (1995a, 5-7) suggests 10 principles for successful partnerships. They are as follows: (1) provide benefits to local people; (2) meet local needs; (3) plan holistically; (4) plan protected areas as a system; (5) define objectives for management; (6) plan site management individually, with linkages to the system; (7) manage adaptively; (8) foster scientific research; (9) form networks of supporting institutions; and (10) build public support.

Integrated Conservation-Development Projects (ICDP's) are comprehensively evaluated in Wells *et al.* (1992). These are projects that attempt to ensure the conservation of biodiversity by reconciling the management of protected areas with the social and economic needs of local people (Wells *et al.*, 1992). Although ICDP's appear to promise some achievement of conservation and economic goals, the report identified weaknesses in the design of ICDP's analysed. These include *inter alia*, weakness in institutional arrangement, limited finances, human population pressure, and hostility between the parties.

In Furze *et al.* (1996) various case studies of local level involvement in protected areas are analysed. The main theme is that natural habitats are in fact cultural habitats largely created by human influence. In sum, environmental and economic goals must be linked with protected area management. Hence the need to study and analyse social and cultural aspects of conservation and involve local people.

Bromley (1997) adds that the re-incorporation of people and their domestic animals into wildlife reserves may actually facilitate improved conservation policy and outcomes. Bromley (1997, 20) defines Community-Based Conservation (CBC) as initiatives:

> *concerned with inducing certain behaviours among individuals and groups closest to particular ecosystems in order to enhance the long-run sustainable management of those ecosystems....In essence, we seek new resource management regimes in which the interests of those living in such regimes coincide with the interests of those living elsewhere who seek wildlife preservation.*

Biosphere reserves are another innovative strategy linking conservation and sustainable development as well as research and monitoring (Furze *et al.*, 1996; Bequette, 1998a, 1998b). The èssence of biosphere reserves are: the different roles mentioned above, plus the zoning of habitat into either core, buffer and transition zones and the involvement of local communities. Biosphere reserves also meet the objectives of various multilateral conservation agreements and in addition provide for the development of local communities.

2.4.3 Criticism of Community-Based Conservation projects

In theory such projects appear to succeed in achieving both conservation and economic development goals. However in many instances research points towards these new strategies not achieving their stated objectives. Barrett and Arcese (1995), critique the appropriateness and sustainability of ICDP's.

> *ICDP's are not yet analytically or empirically sound approaches. They proceed from untested biological and economic assumptions, many of which are likely false* (Barrett and Arcese, 1995, 1080).

Wainwright and Wehrmeyer (1998) in a study critique community-based natural resource management (CBNRM) projects in the Luangwa Valley, Zambia.

> *The research suggests that the Luangwa Integrated Resource Development Project (LIRDP) has generally failed to achieve its conservation and development objectives and that the program has achieved few community benefits.*

Kremen *et al.* (1994) conclude that fundamental differences between conservation and economic development will make integration difficult. Further, for success and feedback the projects need to incorporate ecological and impact monitoring. However, very few ICDP's carry out such monitoring. It is not known with certainty whether the projects are conserving biodiversity.

In summary, the current interest in community involvement in conservation has led to an increase in research and associated literature. Studies from east and southern Africa appear to dominant the literature. Throughout this study reference will be made to these initiatives and projects. Although there is criticism of community-based conservation and economic development projects, the criticisms often lack rigorous analysis of the advantages of property rights. Economic theory points out the benefits and gains of well-defined property rights. The above mentioned issues and criticism are further detailed and discussed in Chapter 5.

2.4.4 Literature on property rights

Property rights are covered extensively in the literature (Runge, 1986; Southgate, 1990; Bromley, 1991, 1997; Naughton-Treves and Sanderson, 1995; Hanna *et al.*, (eds.), 1996). In the literature assigning property rights is seen as a solution to halt the loss of biodiversity. The issue is usually explored from the exploitation of biodiversity as an open-access resource followed by an explanation into the advantages of assigning property rights. There is however debate in the literature over which property right regime is best for protecting biodiversity. According to Alchian and Demsetz (1973) private property is assumed to be the best regime to protect and maintain biodiversity. Runge (1986) and Bromley (1991, 1997) however, argue that state and common property can be equally efficient at conserving biodiversity. McKean (1996) presents a theoretical justification for using common property regimes as a mechanism to internalise environmental externalities and align property boundaries with ecological boundaries. Berkes *et al.* (1989) provides examples of successful conservation under common property regimes.

The informational value of biodiversity is increasingly looked upon as very valuable. Indigenous peoples are also demanding traditional ownership rights to this information. Intellectual property rights (IPR's) have been proposed to protect this information and for the owner to gain from the use of the information, as well as creating incentives for conservation. Swanson (ed.) (1995b) analyse the value of medicinal plants and outlines a property rights based approach to conservation.

2.4.5 A multi-disciplinary approach to conservation: Ecological economics

Traditionally economists focused on the economy, while biologists studied living organisms. Separately they have failed to conserve biodiversity. A new integrated approach is needed. Ecological economics has grown from the realisation of economist and ecologists of the need to incorporate each other's concepts into decisions and policy.

> *Ecological economics is not a single new paradigm based in shared assumptions and theory. It represents a commitment among economists, ecologists, and other academics and practitioners to learn from each other, to explore together new patterns of thinking, and to facilitate the derivation and implementation of new economic and environmental policies* (Norgaard, 1994, 463).

This has resulted in the growth of publications on this topic (Barbier (ed.), 1993; Perrings *et al.*, (eds.), 1995; Swanson (ed.), 1995; Guruswamy and McNeely (eds.), 1998). Tisdell (1990) summaries the origins of ecological economics. Barbier (ed.) (1993) presents studies to show how each discipline can learn from each other. Essentially the book revolves around "operationalising" sustainable development (Barbier, 1993). Perrings *et al.* (eds.) (1995) also supports this integration. Their focus is on addressing issues such as the ecological consequences of biodiversity loss and integrating economic and ecological issues.

Jordan (1995) adds that social factors are important as well as ecological and economic factors in achieving the goal of conservation. This theme is expanded upon by Furze *et al.* (1997). Berkes (1996) suggests that property rights will expand the scope of ecological economics by incorporating the social/institutional/cultural dimension. Moyle (1998a) outlines a bioeconomic methodology for species conservation and sustainable use.

Many of these publications agree that human economic welfare is dependent on the healthy functioning of the biosphere. To better understand and estimate the contribution of nature to human economic welfare an integrated approach is necessary. Such an approach may also correct misleading market signals and amend market and policy failures. However, as concluded by Perrings *et al.* (1995b) further research into the development of new paradigms that take a multi-disciplinary approach is needed.

2.5 ECONOMIC DEVELOPMENT

Economic development is a key concept in this study. According to the World Bank, development is the most important challenge facing the human race (cited in Todaro, 1994, 3). Although theories of development can be traced back to Adam Smith, the systematic study of economic development emerged after the Second World War[4] (Todaro, 1994). Theories of economic growth and development have evolved and gained recognition as a subdiscipline of economics since then. Over time several distinct theories and models have been generated. There is now an abundance of research, studies, and publications dealing with this subject. For instance, the World Bank annually publishes the *World Development Report*. These reports include socioeconomic statistical data and indicators that can be used for international comparisons of countries. In addition, each annual has a specific theme as its main subject. For example, the 1992 report explored the link between economic development and the environment (World Bank, 1992). This section will review the main theories and models that have been used to explain economic development and the processes by which countries have developed.

2.5.1 Economic growth and development defined

The concepts economic growth and economic development, although used interchangeably are distinctly different (Ezeala-Harrison, 1996). Economic growth refers to the increase in an economy's real gross domestic product (GDP) and per capita income, they are typically quantitative measures (Hayami, 1997). However, development is defined as, 'improving the well-being of people' (World Bank, 1992, 34). Development must however be preceded and prompted by economic growth (Ezeala-Harrison, 1996).

> *[Development] encompasses the process through which societies, or nations, or regions raise they per capita output and income by improvements and increases in productivity, and how these translate into per capita economic well-being in the society* (Ezeala-Harrison, 1996, 3).

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[4] See Ezeala-Harrison (1996, 61-83) and Hayami (1997, 104-111) for a review of the Classical Theories of economic growth and development.

To measure development accurately requires the inclusion of non-economic and social indicators of well-being. Furze *et al.* (1996, 5) describe development as,

> *the process of intervening in existing forms of society (which includes social, political and economic structures) in order to achieve desired social, political and economic goals.*

Hayami (1997, 2), states the scope of development economics:

> *The major task of development economics is to explore the possibility of emancipation from poverty for developing economies. It should be strongly focused on low-income developing countries where poverty is especially acute. How can low-income economies in the world today be set on the track of sustained economic development for the immediate goal of reducing poverty and the long-run goal of catching up to the wealth of developed economies? The ultimate goal of development economics is to obtain an answer to this question.*

Hence, development attempts to improve the economic well-being of developing countries.

Todaro (1994) states that there should be three objectives at the core of any development theory. First, to increase the availability and widen the distribution of basic life sustaining goods. Second, to raise the standard of living as measured by income. Third, to expand the range of economic and social choices.

> *Development must therefore be conceived of as a multidimensional process involving major changes in social structures, popular attitudes, and national institutions, as well as the acceleration of economic growth, the reduction of inequality, and the eradication of poverty* (Todaro, 1994, 16).

In summary, the literature on economic growth and development is about trying to understand the reasons for the inequalities and differences (such as income and wealth) between countries. This includes accounting for the relative affluence of a small number of countries, for instance the members of the OECD. Also of interest is why some countries are converging toward this group while many other countries remain stagnant. To shed light on these issues various economic growth and development theories and models have been generated. These theories and models are usually derived from the experiences of developed countries and are explained in the following section.

2.5.2 Theories and the process of economic growth and development

In the post World War II period many countries gained independence from colonial rulers. Since then there has been a procession of theories and models put forward to explain and assist these countries develop. Each has added new insights and understandings of the developing world. There are two main consideration in analysing economic development. First, the theories to explain economic development and second, the process of economic development. Development however can mean different things to different people. Moreover, development is ideologically constructed, with the theories mainly based on capitalist (market economies) or socialist (centrally planned) models. Different assumptions and conditions (some explicit, others implicit) therefore accompany the different theories and models.

Generally both the western and socialist development models perceived economic development as a process of change from traditional agrarian societies to a modern industrialised society (Grabowski, 1989). A key characteristic of both these schools of thought is to maximise the wealth of the nation. Further, these models in general viewed the environment and natural resources as inputs for the production of goods and services.

As many countries gained independence from the middle of the 20th century there was a requirement to understand and help them develop. There was however, a lack of readily available data, methods, and models to follow (Todaro, 1994). Nevertheless there was the historical pattern of development of the Western advanced countries and the recent experience of the Marshall Plan in Europe to analyse and apply to the developing countries. The recently liberated Western European countries were initially in a state of poverty similar to developing countries but rapidly recovered (Hayami, 1997). Therefore the process of development they followed should be able to be applied to developing countries.

In practise developing countries tried creating conditions similar to the historical path of development followed by developed nations. This path involved a process of change including *inter alia* industrialisation, urbanisation, nation state building, and centralised government. It was a top-down approach to development. It was thought that if developing nations could imitate this process they could escape from the trap of poverty and economic stagnation. In addition, these countries could also leapfrog the most damaging phases of development and achieve faster economic gains than experienced by developed nations. Leapfrogging included adopting technologies and institutions that were efficient. In consequence, productivity and income would increase thus achieving economic growth. According to Hayami (1997, 3):

> *An effective theory of development economics should be based on understanding the similarities and differences of these histories compared with the current situation in low-income countries.*

Todaro (1994) presents the leading theories of economic development. These are: (1) growth models; (2) theories of structural change; (3) the international dependence revolution; (4) the neoclassical, free market counter-revolution; and (5) the new theory of economic growth. Ezeala-Harrison (1996) and Hayami (1997) provide coverage of the main theories of economic development and the processes and problems through which countries developed. Weaver *et al.* (1997) outlines the different development strategies such as laissez-faire, import-substitution industrialisation, growth with equity, and the alternative of state socialism.

During the 1950's and 1960's economic growth theories dominated development thinking (Todaro, 1994). The objective was to maximise economic growth rates and per capita income as rapidly as possible (Todaro, 1994). Kuznets defined a country's economic growth as:

> *A long-term rise in capacity to supply increasingly diverse economic goods to its population, this growing capacity based on advancing technology and the institutional and ideological adjustments that it demands* (cited in Todaro, 1994, 106).

Kuznets in analysing modern economic growth concluded that growth is predominantly dependent upon sustained improvements in technology (Hayami, 1997). Kuznets identified six characteristics of the modern growth process in developed countries (Todaro, 1994, 106). These are as follows:

1. High rates of growth of per capita output and population.
2. High rates of increase in total factor productivity, especially labour productivity.
3. High rates of structural transformation of the economy.
4. High rates of social and ideological transformation.
5. The propensity of economically developed countries to reach out to the rest of the world for markets and raw materials.
6. The limited spread of this economic growth to only a third of the world's population (this relates to the distribution of income and wealth between the rich and the poor countries).

Each of these characteristics are interrelated and mutually reinforcing (Todaro, 1994). The benefits of growth would flow through the economy in the form of jobs, increased income, more opportunities and improved social conditions.

However, there were many constraints to achieving high economic growth rates in developing countries. For instance, low savings, and high population growth rates trapped many countries in a low-income equilibrium (Hayami, 1997). In short, they were unable to escape from a subsistence economy. To escape this trap and achieve sustainable growth, capital investment was a priority.

> *Capital has always been recognised as the single most critical factor determining a nations ability to develop. In fact, it is regarded as the prime mover of the development process.... In a LDC [less developed country] capital accumulation would enable the economy to break, and indeed reverse, the vicious circle of poverty that tends to constrain the ability to initiate growth* (Ezeala-Harrison, 1996, 129).

Two important growth models are the Harrod-Domar model and the Solow model. These models emphasised savings, capital accumulation, and investment as key elements in the process of achieving economic growth.

The Harrod-Domar growth model assumed that the economy's potential GDP is a function of net investment spending and that savings are the ultimate generator of the capacity to invest (Ezeala-Harrison, 1996). That is, the rate of savings determines the growth rate in the economy. Important to this model is capital accumulation. Financial capital needed to acquire physical capital is dependent upon savings decisions (Ezeala-Harrison, 1996). In short, investment in capital goods depends on savings. Hence if savings could be increased this would lead to higher levels of investment and economic growth as concluded by the model.

The Solow growth model also emphasised the role of savings, capital accumulation, and investment. However, Solow included a production function and showed that productivity and technology were the key factors that determined how fast nations developed (Hayami, 1997). In essence, the growth of national output depends on the resources the economy processed. This is expressed as:

$$Y = Af(K, L)$$

Where Y is national output, the parameter A represents an index of technological change, K denotes physical resources and L represents labour and entrepreneurial abilities (Ezeala-Harrison, 1996). The results of the Solow growth model showed that the underlying force in economic growth is technological progress (Dornbusch and Fischer, 1994). Further, even in countries with low savings, borrowing or importing technology from advanced economies might achieve high growth rates (Hayami, 1997). Although the Solow model emphasised market forces, Keynesian macroeconomics was dominant at this time (Ezeala-Harrison, 1996). Consequently the state played an important and increasingly large role in the management of developing economies.

There are several additional reasons for state control and management. First, capitalism and market economies were rejected by developing countries as they were viewed as a mechanism for continued exploitation by ex-colonial rulers (Hayami, 1997). In addition, the Soviet economy had purported to generate good growth rates up to the 1960's (Hayami, 1997). This contributed to the attractiveness of the centrally planned socialist model. Second, developing countries are often distinguished as having thin capital and financial markets and institutions, especially in the rural areas. Hence to mobilise the savings required, governments would have to actively intervene and implement savings regulations to achieve the needed level (Hayami, 1997). Third, central planning and government intervention was necessary to promote greater equity in income distribution and to supervise national development strategies (Ezeala-Harrison, 1996).

The developing countries did achieve high economic growth rates during this period that was consistent with the growth models. However, many nations also remained in a state of underdevelopment. For instance, the technology imported was in general capital intensive. This was inappropriate technology given the relative prices of capital and labour in developing countries. Most developing countries are endowed with abundant supplies of labour but possess very little financial or physical capital (Todaro, 1994). Consequently, importing capital intensive technologies in labour surplus developing economies led to an increase in unemployment and increased poverty. In turn, the unemployed may return to the subsistence sector and exploit biological resources in order to survive, or to derive an income by illegally harvesting valuable species for the internationai market or bushmeat for the domestic market. Finally, these models assumed the existence of similar structural, institutional, and attitudinal conditions found in developed countries (Todaro, 1994). In developing countries many of these conditions are either lacking, incomplete or culturally inappropriate.

In the 1970's development took on a broader perspective. Economic growth was still important, however, the elimination of poverty, income inequality, and distributional matters gained recognition. Meeting the basic needs (food, shelter, water) of the poor and providing them with opportunities to reach their full potential becomes a priority (Streeten and Burki, 1978). Streeten and Burki (1978) outlines the basic needs development strategy.

> ***The aim of a basic needs strategy is, then, to increase and redistribute production so as to eradicate deprivation that arises from a lack of basic goods and services*** **(Streeten and Burki, 1978, 413).**

Further, in the 1970's alternatives to the capitalist growth models and centrally planned models were offered. These include the dependency-school and structural-change models. The dependency-school model argued that it was in the interest of developed nations to keep developing nations in a state of underdevelopment. According to Prebisch, if developing countries continued to export primary commodities and import manufactured products from developed countries, their terms of trade would remain unequal (Hayami, 1997). Therefore economic growth would be slow and developing countries would remain trapped in a dependency relationship with developed countries.

The dependency-school model promoted the import-substitution-industrialisation strategy (ISI) to break this relationship.

> *Leaders in newly independent nations in the Third World were very attracted by this strategy, partly because of their repulsion against the colonial system which imposed the role of the material supply base as well as the manufactured product on them* (Hayami, 1997, 210).

ISI promoted industrialisation in developing countries by substituting previously imported products from abroad with domestically manufactured products (Hayami, 1997). Domestic industry would be protected from foreign competition by trade barriers. Proponents claimed that ISI would create jobs, provide a ready domestic market demand for locally produced raw materials, and the saving of foreign exchange ((Ezeala-Harrison, 1996). 'Unfortunately the experiences of these countries proved very disappointing, and their ISI strategies failed massively' (Ezeala-Harrison, 1996, 191). Todaro (1994) and Ezeala-Harrison (1996) presents various reasons why the ISI strategy failed in developing countries. These include foreign exchange constraints, low per capita income constraints, foreign dependence, economic inefficiency, and technical inefficiency.

Structural change theory focused on the mechanism by which underdeveloped economies transformed their domestic economic structure (Todaro, 1994). Structuralist theories are based on the belief that orthodox growth models may not be suitable for developing countries because the assumptions of those models are largely invalid for developing countries (Ezeala-Harrison, 1996). Important models from this school of thought are the Lewis-Two-Sector model and Chenery Structural Change model (Todaro, 1994). These models emphasised transforming the structure of the economy. Such a transformation might be from a subsistence agricultural economy to an industrial economy through a process of identifying patterns of development and applying the "correct mix" to developing countries (Todaro, 1994).

The neoclassical counter-revolution approach prevailed throughout the 1980's. The central argument of this theory is that underdevelopment results from poor resource allocation due to incorrect pricing policies and too much state intervention by overly active governments (Todaro, 1994). This school of thought favoured supply-side economics with less government involvement in the economy and restoring the free market (Hayami, 1997). Promoting competitive free markets and *laissez-faire* economics the invisible hand of markets and prices would guide resource allocation and stimulate economic growth (Todaro, 1994). In addition, the World Bank and the International Monetary Fund (IMF) were requesting policy reform from governments as a condition for loans (Hayami, 1997). This would help stabilise the macroeconomy and liberalise the microeconomy of developing countries.

However, the market while necessary is not sufficient to achieve sustainable economic growth and development. Appropriate institutions such as civil laws, commercial and contract laws, police and a judicial system to protect property rights and enforce contracts, must support a competitive market (Hayami, 1997). Again, these are usually lacking, incomplete, or thin in developing countries. Further, high levels of corruption impede the effective and efficient functioning of markets as well as enforcement. The above issues all act as constraints to the effectiveness of the neoclassical model as well as any other development model.

Trade strategies play an important role in the process of economic development. In the literature on trade theory comparative advantage (CA) is emphasised. A country has a CA over another country if in producing that good it can do so at a relatively lower opportunity cost (Todaro, 1994). Countries are better off concentrating and specialising in the production of these goods and services and then trade these goods for goods from other countries (Ezeala-Harrison, 1996). In relation to developing countries, CA is usually discussed as low labour costs, and the conditions for growing fruit and vegetables for developed countries markets. Developing countries also have a CA in biodiversity but this is not generally mentioned in the literature.

In general, the trade debate is divided into two main camps, the free traders and the trade protectionists (Hayami, 1997). More often the capitalist models preferred to follow an outward market model while the socialist models followed an inward path. Outward oriented trade policies encourage international trade with little distortion

(Todaro, 1994). On the other hand an inward trade policy is one where production incentives are biased in favour of domestic production and against foreign trade, commonly referred to as import-substitution (Rajapatirana, 1987).

As stated ISI predominated development in the 1950-1960's and since the 1970's export promotion has gained dominance (Todaro, 1994). The successes of the newly industrialising countries (NIC's) of Asia are the most well known examples of export promotion. Todaro (1994) covers the trade debate in detail. He concluded:

> *In short, the consensus for the 1990s leads toward an eclectic view that attempts to fit the relevant arguments of both the free-trade and protectionist models to the specific economic, institutional, and political realities of diverse Third World nations at different stages of development. What works for one may not work for another* (Todaro, 1994, 508).

Following a free trade policy will bring increased economic wealth given certain conditions and institutions that need to be established. In regard to biodiversity this study argues for the unrestricted trade of wild species. However, heavily illegally traded species whose populations have been depleted by poaching may need protection and strict restrictions on any trade to allow populations to recover before free trade is allowed.

2.5.3 Economic development and the environment

This study assumes that all societies have a right to develop. Developing countries have attained positive advancements in economic indicators since the 1950's. Weaver *et al.* (1997) reviews the development record for the period 1950-90. Weaver *et al.* (1997) assessment is that the developing countries averaged a real gross domestic product per capita growth rate of 3.4 percent between 1950-1975. Moreover, social indicators have improved. For example, infant mortality has declined, life expectancy rates have improved, school enrolments have increased and some diseases have been virtually eliminated (Weaver *et al.*, 1997).

> *Between 1950 and 1990, combined life expectancy for men and women in developing countries increased from forty years to sixty-three years, the under-five mortality rate dropped from 280 per thousand to 100 per thousand, and the literacy rate increased from 46 percent in 1970 to 69 percent in 1992* (Weaver *et al.*, 1997, 4).

These results however are not universal. In reality the development record for all nations has been mixed. For a large percentage of the world's population, poverty is still a fact of life. For example, in 1990 a joint study the Food and Agricultural Organisation (FAO) and the World Health Organisation (WHO) reported chronic undernourished people amounted to 800 million, and as high as 40 percent in Africa (Hayami, 1997). Further, there is still a large income inequality between countries. For example, average per capita income in 1990 ranged from $US 20,000 in high-income OECD countries to about $US 100 for low-income countries such as Mozambique and Tanzania (World Bank, 1992).

Moreover, the cost to biodiversity from development has been high (Box 1.2). As economies grow the pressure to over-use biological resources and convert natural ecosystems intensities. In most instances to develop countries converted natural capital into other forms of capital or for consumption. Consequently in developing countries there is often a conflict between preservation of the natural environment and economic growth and development (Barrett, 1993). The result is a trade off between environmental quality and economic growth with,

> *environmental quality often needlessly sacrificed in the name of economic growth, or secured at the expense of economic growth* (Pearce, 1994, 9).

Developing countries have tended to over-exploit natural resources to achieve economic growth (McNeely, 1988).

Brown (1998) from the Worldwatch Institute has researched the environmental effects of continuing economic growth as economies outgrow the Earth's ecosystems.

> *While economic indicators such as investment, production, and trade are consistently positive, the key environmental indicators are increasingly negative* (Brown, 1998, 4).

The costs of following this type of development includes over-exploitation of natural resources, loss of species, the degradation and stress on environmental quality and distributional inequalities. Hence, the relevance to this study of the implications that economic development has on biodiversity and local communities.

Past development had a bias towards land conversion (Swanson, 1994a). The conversion process can be attributed to biological resources not being accurately valued (Pearce, 1991; Barbier, 1993). Hence, natural habitats were converted to specialised homogenous agricultural production (Pearce, 1991; Swanson, 1994a). McNeely (1990) presents tables on estimates of deforestation and wildlife habitat loss in tropical countries. McNeely concludes that conservation action and co-operation is required at local, national, and international levels. In regards to development, there should be a transfer of the best conservation technology to the countries that most need it (McNeely, 1990). The Solow growth model similarly encouraged the import of technology.

Development agencies are now becoming concerned about the depletion of species and the degradation of ecosystems. There is a growing awareness that development depends on the maintenance of biodiversity (McNeely, 1988). Moreover, biodiversity loss now threatens the sustainability of economic development (Tisdell, 1995). Developing countries must therefore be convinced that development and conservation are both possible (Swanson, 1994a). However, according to Dickinson III (1995, 37), 'persuading them to maintain biodiversity is a formidable challenge.'

The success of improved medical and health standards and technology in developing countries has seen a rapid increase in human population growth. However, without an increase in wage employment opportunities and income, environmental degradation results, especially in rural areas. Population growth has caused demand for land to increased and has decreased the endowment of land per agricultural worker, resulting in an pauperisation of the rural population (Hayami, 1997). In consequence, the agricultural frontier has been pushed into ecologically fragile and marginal lands (quite possibly with high biodiversity value, such as Myers hot-spots: refer Table 1.3) but to which property rights are not assigned (Hayami, 1997). Compounding this is the increase in protected areas in developing countries (Table 4.1 and 4.4). However, the lack of administrative capacity in developing countries has been unable to prevent encroachment and illegal harvesting in these areas.

Bromley (1991) presents an argument that governments of developing countries perpetuate the belief that human population growth is to blame for natural resource degradation. Bromley agrees that population growth is a contributory cause of much degradation but it is not the primary cause. Governments have a vested interest to blame natural resource degradation on human population growth. The state then avoids being held responsible and it helps attract development assistance from external donors.

2.5.4 Critique of development

In aggregate development has improved the standard of living in developing countries. However, a high variance exists between countries, and regions (Todaro, 1994; Ezeala-Harrison, 1996). Moreover, the benefits from economic growth have been unevenly distributed and income inequality has been identified as a major development problem (Todaro, 1994). Rural poor communities are most often adversely effected. They in turn exploit and degrade the local natural environment and biological resources (legally and illegally) to survive.

Important to this study are the institutional reasons for these actions. Hayami (1997) argues that the importation of technology from developed countries is profitable for developing countries. However, the importations of foreign institutions for the use of this technology without due regard to differences in cultural values and social conventions may not serve its intended purpose but only create social disorder. What is needed is development policy that utilises local institutions including local knowledge.

As previously stated developing countries have a CA in biodiversity. Free trade should therefore allow developing countries to specialise in the production of biological resources and gain from exporting those goods. In reality developing countries face many barriers external and internal to free trade, such as quotas, tariffs, and microeconomic and macroeconomic constraints. External barriers may occur as trade bans. For instance, a CITES Appendix I listing acts as a barrier to trade in endangered species. Developing countries are prevented from legally exporting these species. There are strong arguments against allowing trade in endangered species. However, Swanson (1994a) argues that these species (unless accurately valued) will probably become extinct via other paths.

In addition there are internal barriers to trade. Macroeconomic policies effect trade and adversely effect biodiversity as the following example demonstrates. Many developing countries have a fixed and often over-valued exchange rate plus controls on foreign exchange (Moyle, 1998a). This makes imports cheaper while exports are more expensive, consequently a current account deficit may emerge (Moyle, 1998a). In addition, this limits the ability of many individuals to maximise income and productivity. The repercussions are that domestic output is reduced and this effects employment levels. For many unemployed the only option is to return back to the subsistence sector and a reliance on biological resources. Further, developing countries may try and increase exports by offering attractive concessions on the extraction of natural resources such as timber and minerals, often accelerating the loss of forests or increasing pollution (Moyle, 1998a). This is a quantity response to a price problem (Moyle, 1998a). Moyle (1998a) argues that a floating exchange rate is needed to solve this problem plus a complementary free trade policy.

In addition, many other factors are also detrimental. These include a lack of transportation, communications, financial, and credit markets. Further, minimum wage regulations can indirectly adversely effect biodiversity. A minimum wage increases wages rates. In consequence employers substitute capital for labour. The result is a decrease in employment and an increase in unemployment. In developing countries this could result in an increase in subsistence living and hunting for bush-meat.

External and internal barriers are linked. For instance, a trade ban denies countries the opportunity to maximise the species TEV. The ban may however, generate opportunities for poachers to gain by illegally harvesting and smuggling the species to overseas markets. At the national level governments have to fund budgets for the protection of biodiversity. However, the banned species is viewed as a cost to developing countries rather than an asset. In consequence, less investment and therefore protection and maintenance are accorded to those biological resources. At the local level, rural villagers and/or the subsistence sector are legally prevented from sustainably utilising these species. They may turn to illegal hunting which will contribute to the loss of biodiversity and species extinction.

However, by reducing barriers to trade and removing or reforming the macroeconomic and microeconomic barriers, may all enhance conservation. For instance, export earnings for developing countries would increase. This could be directed into conservation. To avoid an uneven distribution of export revenue, well-defined property rights are required.

Since the 1970's there has been a revision and reinterpretation of what economic development really means and its relationship to the natural environment. "Sustainable development" and "sustainable use" were proposed as approaches that incorporated the environment and development. Sustainability links ecological and economic aspects into the decision-making process and involves local communities. To achieve this development and conservation must be seen as compatible (McNeely, 1988; Swanson, 1994a). The challenge then is to integrate conservation and economic development. That is, correct the adverse effects such as market and government failure while at the same time ensuring that there are opportunities for nations and local communities to develop.

2.5.5 Sustainable development

Sustainable development is a recent concept used in both development and environmental literature. It provides a framework for the integration of environmental policies and development strategies (WCED, 1987). According to Trzyna, (1995, 7) 'the concept of sustainable development has now acquired such a pedigree that no contemporary discussion on environment and development is considered complete without it.' Lélé (1991) however, identifies weaknesses with the definition of sustainable development. Lélé concludes that it requires further clarification and rigour for it to be a successful form of development. Sustainable development tries to determine the optimal rate of resource and environmental degradation that would neither result in making future generations worse off nor leave the needs of the present unsatisfied (Ezeala-Harrison, 1996).

Sustainability integrates environmental and development concerns into conservation. The IUCN, WWF, and UNEP first used the concept of sustainability in 1980 for the *World Conservation Strategy*. The WCED (1987) first defined the term sustainable development. According to the WCED sustainable development should generate development that meets the needs of the present generation, especially the poor and that it enhances and conserves the natural resource base for future generations. The WCED discussed the challenges of past development and suggested strategies to solve these problems. The report also recognised the role of economics in analysing the costs involved and the design of incentives.

The WCED (1987) also identified causes of species extinctions, trends, and current problems of conservation. New approaches to conservation are also discussed with regard to how species and ecosystems can be managed. These include altering land-use patterns, reforms of land tenure systems and international action. In regards to wildlife, increased protected areas, and anti-poaching measures are discussed.

These issues were an important part of the 1992 Rio Earth Summit. The message from the Earth Summit was that economic and social development is not sustainable unless the environment is afforded a much higher profile in national economic planning and management (Pearce, 1994). One outcome of the Rio summit was *Agenda 21*. *Agenda 21* has the goal to halt and reverse environmental damage and to promote environmentally sound and sustainable development that emphasises the value of indigenous people and involves them in the development process (Sitarz, 1994).

Weaver *et al.* (1997) identify four subgoals to achieve sustainable development, they are; a healthy growing economy undergoing structural transformation, equity in distribution, effective governance, and a political economy that is consistent with the preservation of the environment. In Munro (1995) ecological, social, and economic sustainability are discussed. Finally, Meffe and Carroll (1994d) present a range of sustainable development case studies.

Interestingly many of these reports support the sustainable use of biodiversity as an effective conservation measure, rather than a preservation approach. The emphasis is now on effective management and methods that sustainably utilise biological resources. Importantly biodiversity must be valued as an economic resource to be assured protection. Property rights are the key to what level of investment in protection is expended on biodiversity.

Sustainable development goes further than past definitions of development. It has a human and time dimension. Sustainable development values life both in the present and the future. Sustainable development has however been criticised for being too utilitarian and anthropocentric (Meffe and Carroll, 1994d). That is, the orthodox definition does not go far enough to recognising the extraordinary complexities of nature. Although sustainable development is future orientated it is equally, if not more important to ensure that the present generation's needs are met. In conserving wildlife it is important that local communities are not disadvantaged and that conservation creates opportunities for these people to develop.

2.5.6 Local communities and development

Much is written on local communities and development. Often, they have the closest relationship with biodiversity. The IUCN (1997) present case studies and actions that link indigenous peoples, sustainability, and biodiversity. Runge (1986) describes common characteristics of village life in developing countries and how they relate to economic development. They are relative poverty, dependence on natural resources, and uncertainty with respect to income streams. However, local communities have largely been overlooked in conservation efforts (Pearce, 1991). From an economic point of view local communities are likely to be the most affected by biodiversity decline and conservation (Tisdell, 1995). Further, the majority of the economic growth and development theories recommend large-scale government or private interventions. Very little is paid to the characteristics of the local environment, or the unique opportunities offered by local people and their institutions to successfully implement local initiatives.

In the 1990's there has been a rethinking and a growing interest in participatory or a "bottom-up" approach to conservation and development, that involves all stakeholders (Trzyna 1995; Furze *et al.*, 1996; Weaver, *et al.*, 1997). For conservation projects to be successful the literature argues for the support and involvement of local people (Wells *et al.*, 1992; World Bank 1992; Furze *et al.*, 1996). Without this active collaboration conservation may not succeed (Pearce, 1991; Wells *et al.*, 1992; Barrett and Arcese, 1995; Tisdell, 1995; Simon, 1995). This will entail local communities appropriating a fair share of the benefits of conservation and sustainable use (Pearce, 1991; Brown *et al.*, 1993; Jordan 1995; Tisdell, 1995). To ensure benefits of conservation are distributed to local communities, property rights will need to be well defined.

2.6 CONCLUSION

Economic explanations that account for present loses of biodiversity have become more sophisticated. In the literature economic solutions are now offered for what was once thought to be biological problems. Recent literature on conservation and economic development now emphasises the interdependence between economics and ecology. Moreover, much is made of the advantages of the sustainable use of wildlife. Research, analysis and solutions have developed around the conservation of charismatic species. Politically the larger charismatic animals are important. However, very little analysis and information is available on lower order vertebrates and invertebrates and their role in maintaining the ecosystem (Simon, 1995). They are just as important and sometimes more important than the larger species (Gillings, 1998).

The literature on economic growth and development acknowledges the connection and dependence of sustainable economic growth on environmental goods and services. In general the effects of pollution, human population growth, and degradation of forests and desertification are the major topics discussed in the literature. The quality of the environment is the main concern. Solutions are directed towards using economic tools (pollution permits and taxes) to improve environmental quality.

Rarely was biodiversity as a specific resource and valuable asset in economic growth and development referred to in detail. Recently, there has been a shift towards acknowledging the value of biodiversity in economic development. Importantly to understand economic development there is a need to analysis social and cultural institutions especially in regards to property rights for biodiversity. Not understanding or incorporating these institutions (such as common property regimes), or replacing them with alternative but appropriate institutions can have adverse effects on the environment and biodiversity. This area needs further research.

Past literature has tended to neglect local communities in conservation polices. The definition of local communities also requires clarification. This is especially so with regard to property right regimes. This includes whether ownership is just of the land and resources or involves just the use of the resources. Further, will local community ownership ensure species are maintained? Local communities may decide to convert natural habitat into agricultural use. In addition, they may lack full information on values and uses of local biodiversity, especially if markets are thin.

A large amount of research still needs to be carried out on economic, biological, and cultural issues. There is still much ignorance and uncertainty with regards to integrating these disciplines. Conservation efforts need to be empirically tested regarding effectiveness. Further uncertainties of the effects of past economic development and current policies are unknown and only predictions can be made on future impacts and results. Nevertheless, the theories and models are continuously evolving. Moreover, the issue of sustainable use of wildlife is very emotive. However, by researching and analysing these issues and identifying problems and constraints, solutions can be found that can improve and achieve economic development and conservation objectives. It is believed that property rights can overcome many of the above mentioned problems and constraints.

3

Property Rights

***It would not be overstating things to assert that environmental problems are property rights problems* (Bromley, 1997, 1).**

***Today, much of the dispute over wildlife conservation involves property and property rights* (Naughton-Treves and Sanderson, 1995, 1265).**

3.1 INTRODUCTION

It is now recognised that much of the Earth's biodiversity is threatened. Extinction is a distinct possibility for many species of flora and fauna. Ecological services and functions have been in many cases severely degraded. The issues surrounding the reasons and causes of these problems are complex. However, human decision-making and behaviour especially with regard to ownership, rights, and use are thought to be responsible for much of the current problems and the resulting status of biodiversity.

Further, to achieve economic efficiency certain institutions are required. These factors must be analysed fully and precisely to both understand the problems and supply answers. Property right institutions, the structure, consequences, and benefits are important variables in developing strategies to protect biological diversity.

> *Policy reform, to achieve a better regime of resource and property rights is one of the most pressing issues in achieving biodiversity conservation* (Furze *et al.*, 1996, 208).

Although a lack of property rights has been blamed for the degradation of biodiversity, this chapter will argue that institutional change and institutional weakness are causes of biodiversity loss that have been neglected. North (1994, 360) defines institutions as,

> *humanly devised constraints that structure human interaction. They are made up of formal constraints (e.g., rules, laws, constitutions), informal constraints (e.g., norms of behavior, conventions, self-imposed codes of conduct), and their enforcement characteristics. Together they define the incentive structure of societies and specifically economies.*

Institutions exist in developed modern economies and traditional indigenous groups and communities. An authority system that enforces the existing and relevant institutions is essential for the proper and efficient functioning of any property right regime. Finally, common property right regimes have also been misunderstood and can provide effective protection of biodiversity. Hence, strengthening property rights may therefore enhance the protection of biodiversity which would allow biodiversity to be used more efficiently and sustainably.

3.1.1 Chapter outline

Section 2 reviews the different property right regimes, the conditions needed for efficiency, how they are constructed and addresses what is meant by property and property rights. Section 3 will specifically look at property rights in relation to biodiversity. Discussed first are some indigenous institutions that regulate the exploitation of biodiversity. Next a critique of the "tragedy of the commons" is presented. Following this is a discussion on the four different types of property regimes. Limitations of state and private property are also commented upon. Finally, evidence is presented to show that common property regimes and local level management are viable alternatives. This chapter concludes that protection and sustainable use of biodiversity is more about strengthening and building institutions than about specific property right regimes. These institutions must be stable and accepted before there will be adequate investment in biodiversity conservation.

3.2 PROPERTY RIGHTS

3.2.1 Why assign property rights to biodiversity?

Biological diversity in all its three forms has present and future value to humans. Human decisions now determine which biological resources will be protected and maintained. Biological resources are also scarce, so there will be conflicts of interest and demands over use and protection. This has local, national and international dimensions.

To resolve these conflicts, decisions and choices have to be made. These decisions and choices relate to the allocation of resources to protect and maintain biodiversity and to who can utilise those natural resources. This will require efficient and effective institutions and systems especially in relation to ownership. Implementing and enforcing property right regimes is seen as a solution to the biodiversity problem. Property rights create incentives for the owner(s) to seek the highest-valued uses of their property (Randall, 1987; Pejovich, 1997). However, the problem with biodiversity is much of it is unowned or owned with poorly defined property rights (Moran and Pearce, 1997). In effect then resource owners have little incentive to invest in conservation.

3.2.2 Types of property rights

Bromley (1991, 1997); Pearce (1994); Sedjo and Simpson (1995); Furze *et al.* (1996); Hanna *et al.* (eds.) (1996); Moran and Pearce (1997) all discuss the different types of property right regimes: private property, state property, common property and non-property or open-access (Table 3.1). All four ownership regimes are legal institutions that assign rights to individuals, groups, and firms to use those resources (Randall, 1987).

According to Pejovich (1997) the right of ownership contains three key elements; exclusivity, transferability of ownership, and constitutional guarantees of ownership. Randall (1987) lists the set of conditions required for nonattenuated property rights to achieve economic efficiency (also see Table 3.2). They are as follows:

1. completely specified, so that it can serve as a perfect system of information about the rights that accompany ownership, the restrictions on those rights, and the penalties for violations;
2. exclusive, so that all rewards and penalties resulting from an action accrue directly to the individual empowered to take action (that is, the owner);
3. transferable, so rights may gravitate to the highest-valued use. The right to sell and to buy must be free from any restrictions so that there is mutual agreement over the terms of transfer. Further, for efficiency the rights to transfer base resources (land) and biological resources must be independent;
4. enforceable and completely enforced. This enables owners to accumulate wealth via their investments. An unenforced right is no right at all.

Table 3.1: Four types of property regimes.

State property	Individuals have *duty* to observe use/access rules determined by controlling/managing agency. Agencies have *right* to determine use/access rules.
Private property	Individuals have *right* to undertake socially acceptable uses, and have *duty* to refrain from socially unacceptable uses. Others (called "non-owners") have *duty* to refrain from preventing socially acceptable uses, and have a *right* to expect only socially acceptable uses will occur.
Common property	The management group (the "owners") has *right* to exclude nonmembers, and nonmembers have *duty* to abide by exclusion. Individual members of the management group (the "co-owners") have both *rights* and *duties* with respect to use rates and maintenance of the thing owned.
Nonproperty	No defined group of users or "owners" and so the benefit stream is available to anyone. Individuals have both *privilege* and *no right* with respect to use rates and maintenance of the asset. The asset is an "open-access resources."

Source: Bromley, 1989, 872.

Table 3.2: Property rights regimes and conditions for efficiency.

	Open-access	Common property	Private property	State property
Universality	No	Defined for the group	Yes	No
Exclusivity	No	Defined for the group	Fails in the presence of externalities and public goods	No, but non-nationals excluded
Transferability	No	Applies for the group	Yes	No
Enforceability	No	Yes: legal and social sanctions	Yes: legal and social sanctions	Yes: legal sanctions
Overall Efficiency	Very low: no incentive to conserve	Many regimes are efficient, but inherent risk of breakdown	Efficient but market failure occurs in presence of externalities and public goods	Often inefficient due to government failure

Source: Pearce, 1994, 20.

Hanna and Jentoft (1996, 42) define efficiency in resource management, 'as the cost-effectiveness with which the property rights and rules are implemented; that is, the ratio between the effectiveness of the outcome and the effort required to achieve it.' If all the above conditions exist, divisibility, transferability, full information, and well functioning markets then private property is the most efficient regime to allocate resources. Bromley (1989) argues that these conditions are rarely present for biodiversity.

Many factors such as: biological, economic, informational, cultural, political, geographical, historical, and legal will help determine the appropriate regime (Hanna *et al.*, 1996). In addition, each property regime has associated administrative and transaction costs, which will also determine what regime operates (Bromley, 1991). In essence, there is no prior reason to suppose that one regime will be adequate in all cases to protect biodiversity. Consequently each of the different property right regimes will generate some conservation.

For instance, in many parts of Europe private land ownership has enabled deer, boar, and bears to survive (Moran, 1992). However, in Switzerland while it is efficient and cost effective for valley-bottom land to be privately owned, summer pastures are managed under a common property regime (Bromley, 1991). In South Africa private game ranching and private wildlife recreation are important to overall conservation

(Benson, 1991). For example, in 1990 private enterprise utilised 8.6 million hectares for wildlife conservation compared to 5.4 million hectares that was managed by the state (Benson, 1991). What has made private game ranching successful in South Africa is that ranchers have control over access and use of their land (Benson, 1991).

In North America approximately 33 percent of the land base has been acquired and managed by the state (Benson, 1991). According to Geist (1994) public ownership has helped to return wildlife from the brink of extinction and made it abundant and economically important. In New Zealand the Department of Conservation (DoC) manages about 32 percent of the country (Hartley, 1997).

In Africa much land is vested in group or common ownership. The group leaders or owners then allocate use rights on portions of the land to various individuals or families (Bromley, 1991). For example, the CAMPFIRE[1] (Community Area Management Programme for Indigenous Resources) model in Zimbabwe is seen as an example of a successful common property right regime. CAMPFIRE is a decentralised natural resource management program designed to give full control of wildlife management to local people (Furze *et al.*, 1996). In Nepal communal management of forests has halted the deforestation rate and degradation experienced under state ownership (Berkes *et al.*, 1989) (Box 3.3).

The above examples illustrate that conservation can occur under different property regimes. It is however recognised that some form of ownership rather than open-access is preferred if preservation and conservation of biodiversity is an objective (Berkes, 1996). Assigning property rights will increase the incentives for owner's to manage and protect their resources. This is most likely if the use of that resource provides an opportunity to increase income. However, rights have to be legally and/or culturally recognised and enforceable for there to be an incentive for the owner(s) to look after that resource.

Further, property rights while necessary, are not sufficient to guarantee conservation of that resource. Markets are also necessary both for the property right and the resource. Markets (given certain assumptions) are dynamic mechanisms that are able to respond to changes in prices, and in consequence can add to the efficient allocation of resources to the highest valued use (in this study to conservation). If markets for biodiversity are non-existent owners of the habitat may convert the base resources into

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[1] Chapter 5 includes an expanded case study on CAMPFIRE.

alternative market uses. Equally important is that market returns must be at least equal to the values captured from alternative uses to maintain the biological resource. For instance, the returns from wildlife viewing or trophy hunting for example, must be equal to or higher than the returns from cattle ranching for example, before natural habitat is preserved for wildlife. If returns are lower, habitat may be converted and some biodiversity lost. Moreover, if the state intervenes in the market (e.g., impose a trade ban) this may also have adverse consequences for biodiversity. For instances, owners of biological resources may decide to harvest all of the resources before the ban without conservation in mind and invest the returns into other investments. Well-functioning markets may therefore help increase incentives for investment in conservation.

3.2.3 Property and property rights

A common interpretation (or misinterpretation) of ownership is that it gives the owner the right to use that resource as they see best. However, Alchian and Demestz (1973; Pejovich, 1997) note that it is not the resource (as a physical object) that is owned but the socially recognised rights of action that are owned. Bromley (1991) distinguishes between *property* and *property rights*. According to Bromley (1991, 1997), property is to have control of a benefit (income) stream from the resource and the right to this income is only secure as long as others respect the conditions that protect that stream.

> *A right is a triadic relationship that encompasses my interest, the outcome or object of my interest (whether a physical object or a stream of benefits), plus all others with conflicting interests, yet with a duty to respect my right. Rights are not relationships between me and the object, but are rather relationships between me and others with respect to that object. Rights only have empirical content when there is a social mechanism that gives duties to those interested in the particular outcome guaranteed to the right-holder* (Bromley, 1997, 3).

Pearce (1994, 11) adds,

> *A property right is an entitlement on the part of an owner to a resource or good and where the entitlement is socially enforced. Such rights tend to be attenuated by various legal and customary restrictions which define limitations on the use or consumption of the good or resource.*

In summary, property is a stream of benefits and rights to property offer varying degrees of security over that benefit stream.

Property and property rights only exist if there are social mechanisms that give rights and duties and that bind individuals to those rights and duties (Bromley, 1991, 1997) (Table 3.1 and 3.3). An authority system is needed to transmit these rights and duties. In many instances the authority system is assumed to be what society decides and wants. However, Arrow's Impossibility Theorem has proven that there is no possible mechanism that would allow us to aggregate over individual choices to arrive at consistent and coherent collective choices (Bromley, 1991). That is, there is no "perfect" way to make social decisions (Varian, 1993). Arrow proposed an alternative:

> *In a capitalist democracy there are essentially two methods by which social choices can be made: voting, typically used to make 'political' decisions, and the market mechanism, typically used to make 'economic' decisions... voting would lead to inconsistent choices. The message [is]... markets are the only way that consistent choices can be made* (Bromley, 1991, 213).

This conforms to Adam Smith's articulation of the "invisible hand."

Nevertheless an authority system is necessary. It has a role that includes specifying rights, plus enforcing and protecting property rights when they are threatened. Examples of authorities range from local village chiefs to local and national governments. When no authority exists, is weak or inconsistent any property right regime may degenerate into open-access (Bromley, 1997).

Property rights are dynamic. If economic, social, and political conditions change and/or tastes and preferences change property institutions may have to be reevalulated to make sure they are not counterproductive (Bromley, 1989). For example, open-access may initially apply to land and biological resources. Once those resources become scarce, valuable and marketable, individuals, groups, or the state will try and establish a property right regime to exclude other potential users. Fishing communities are good examples of this process in action (Berkes *et al.*, 1989; Hartley, 1997). This conforms to the Hayekian argument that in many situations, social mechanisms evolve spontaneously from the bottom up (Streit, 1997). Randall (1987) adds that the institutional structure of property rights must be consistent and accepted by members of the group. If they are not there may be little compliance and many conflicts over rights and duties. For example, conflicts over property rights and access in many national parks in Africa have resulted in local communities not complying with park management rules (Newmark *et al.*, 1993).

Table 3.3: Types of property right regimes: owners, rights, and duties.

Regime type	Owner	Owner rights	Owner duties
Private property	Individual	Socially acceptable uses, control of access	Avoidance of socially unacceptable uses
Common property	Collective	Exclusion of nonowners	Maintenance, constrain rates of use
State property	Citizens	Determine rules	Maintain social objectives
Open-access (nonproperty)	None	Capture	None

Source: Hanna *et al.*, 1996, 5.

Importantly property rights for land do not necessarily equate with rights of use. Nor do resource use rights mean ownership of land. It is possible to have specific resources rights without property rights to the land. For example, birding rights were granted to former Maori owners of the Trios Islands in the Marlborough Sounds, New Zealand as one of the conditions under which land was transferred to the Crown (New Zealand Conservation Authority, n.d.).

Further, ownership is often divisible. That is, more than one group or more than one type of property right regime may exist over resources. For example, in South Africa the state may own a core conservation area while surrounding private land owners are contracted to manage their land in such a way which complements the government's conservation policy (Bakker-Cole, 1996). In Australia, indigenous communities often supply the land and jointly manage and land and wildlife with the government (see Furze *et al.,* 1996 for case studies). In Africa, land is managed as common property while held in *usufruct* by community members (van den Brink and Bromley, 1992). Finally, it is possible for private property owners to supply public goods and for public owners to supply private goods.

3.2.4 Summary of property rights

The property rights surrounding the ownership, use and protection of biodiversity are embedded in institutions. Successful property right regimes are characterised by strong, enforceable and accepted laws and regulations (institutions). In the absence of consistent and coherent institutional arrangements the socially determined authority is broken down and existing property rights are delegitimised resulting in resource degradation (Bromley, 1991). In short, property rights are very complex institutions. Owners have certain rights while nonowners have to follow certain restrictions. Property rights are however important for the effective conservation of biodiversity.

3.3 PROPERTY RIGHTS AND BIODIVERSITY

3.3.1 Property rights, social customs, biodiversity and indigenous peoples

The developing world is home to much of the world's remaining biological diversity and many indigenous peoples. Indigenous groups live close to, and depend on the local natural environment for subsistence. Many have very strong cultural, spiritual, religious, and economic attachments to biodiversity. Many indigenous communities have successfully managed land and biological resources sustainably for thousands of years (IUCN, 1997). For instance, the Makah a native North American tribe from the northwestern USA has sustainably hunted the grey whales for more than 2000 years (De Alessi, 1996). This successful harvesting is due to the Makah institution of property that creates incentives for conservation (De Alessi, 1996).

Indigenous groups and local communities further enforce resource rights and sustainable use practices through customary laws, beliefs, and relationships with wildlife (Box 3.1). For instance, in Maori society, the authority to enforce and respect resource management practises and concepts such as tapu and rahui are derived from myths and legends (The Natural Resource Unit, 1991; Orbell, 1995). In South East Asia, the harvesting of edible birds' nests comprised of saliva from the *Collocalia fuciphaga* Gmelin and *Collocalia germani* Oustalet species of swiftlets has been practised for at least two centuries (Er *et al.*, 1995). Many superstitions, religious beliefs and taboos have evolved alongside this harvesting and helps to conserve biological stocks (Er *et al.*, 1995). Finally, in the Artic region Inuit sustainably hunt polar bears (Freeman, 1998). An example of a hunting taboo is the prohibition on hunting females accompanied by yearlings or two-year olds amongst the Canadian Inuit (Freeman, 1998).

Box 3.1: Social customs in Indonesia and West Africa.

In traditional communities, social customs and taboos limit what individuals can do with common land. For example, harvest rights from communal tropical forest areas in Indonesia are given to individuals who plant valuable trees. Beliefs in forest gods and supernatural forces and concerns for future generations limit excess harvesting. These social custom, not the form of ownership *per se*, effectively check individual incentives to overharvest. Thus it is distinctly possible to conserve resources under common property systems.
Source: Mendelsohn and Balick, 1995.

In many parts of West Africa, for example, forest areas and specific trees are protected and valued for particular cultural occasions and as historic symbols. Each community has its own traditions associated with sacred areas. These groves are often found to be in areas where they protect water-sheds, or which have specialised ecological functions. They often contain rare or special species. The set of rules governing these areas may allow for limited exploitation; by particular people from the community (priests, herbalists, healers), or only at certain times of the year (holy days, feast days).
Source: Brown, 1995.

However, these practices and beliefs were seen as "backward" or "witchcraft" by many colonial administrations (Brown, 1995) and also by nationalist post-war governments. Their cultural, economic, and environmental values, plus institutions and rules were not recognised as legitimate. Rather they were discouraged, outlawed and replaced by other rules, regulations, and administration.

3.3.2 The tragedy of the 'tragedy of the commons'

Today much of the Earth's biological diversity is listed as threatened or endangered. Overgrazing and deforestation of natural habitat are major causes for the decline. Traditional property rights and common property institutions in many instances have been blamed. However the problem is that many common property systems have been misidentified as open-access (Berkes *et al.*, 1989; Bromley, 1991; Furze *et al.*, 1996; Hanna *et al.*, 1996).

> *One of the greatest tragedies of recent times has been the degree to which Hardin's 'tragedy of the commons' has been accepted* (Young cited in Furze *et al.*, 1996, 180).

Hardin constructed a model showing that when property rights are poorly defined or impossible to establish this will lead to unconstrained use and degradation (Hanna *et al.*, 1996). Hardin used herdsmen increasing the size of their herd grazing on common land as an example demonstrating the problem of overgrazing (Hanna and Jentoft, 1996). This he called the "tragedy of the commons," a parody of individual human behaviour responding to the incentives of open-access resources (Hanna and Jentoft, 1996). However, Hardin confused open-access with common property. 'The Hardin metaphor is not only socially and culturally simplistic, it is historically false' (Bromley, 1991, 22).

Berkes (1996) argues that the "tragedy" occurred only after the destruction of communal property that created open-access conditions. Bromley (1991) adds that the real tragedy is not that of the commons but the process whereby indigenous property right structures have been undermined and delegitimised. Existing local level/indigenous authority systems are assumed to be deficient.

> *The fallacy of the tragedy of the commons allegory is that by failing to understand property, and thus to see the world as dichotomous between open access (which is bad) and private property (which is claimed to be good), the commentators could leap from the presumption of destruction to the presumption of wise management with one quick sleight of hand* (Bromley, 1991, 147).

Bromley (1991) further argues that governments in developing countries failed to provide conditions for economic growth and development. This has added to the tragedy of the commons. Most governments nationalised natural resources and set aside areas as national parks and reserves. These centralised actions were largely ineffective. Furthermore, they had an effect of stifling the market sector. The most adversely affected are many of the people who once sustainably utilised communal property resources that are now owned by the state. Many are still living a subsistence lifestyle and the state actions have shrunk their access to natural resources. Consequently as a result of the breakdown of past traditional rights and duties regulating resource use combined with an inefficient state authority the tragedy of the commons occurs.

In addition a second tragedy occurs. This is the failure of governments to replace traditional property rights with other property institutions. 'This institutional vacuum and the attendant poverty of people at the village level, then give rise to the tangible tragedy of resource destruction of so much concern' (Bromley, 1991, 106). The tragedy of the commons was the rationale for implementing developed world state and private management over biological resources from indigenous control.

3.3.3 Private ownership

In developed countries, private ownership is a familiar type of property right (Bromley, 1991; 1997). Supporting this are the legal and political institutions that first, are relatively well-functioning and second, are able to enforce the rights and duties of private property. These include the right to control, the right to transfer, and the right to use (Randall, 1987; Bromley, 1997). In addition, the conventional wisdom is that private property is a necessary condition for the efficient allocation of resources. That is, allocating private property rights can minimise costs and generate incentives for owners to manage their resources to greatest benefit (Alchian and Demsetz, 1973; Bromley, 1991; Hartley, 1997). Consequently economic wealth is maximised. Hence, private property receives support from a number of economists, especially from the neoclassical school (Bromley, 1991; Naughton-Treves and Sanderson, 1995; Freese, 1998).

In recent years policy makers have argued for the privatisation of rights for natural resources (Naughton-Treves and Sanderson, 1995). In regards to conservation, property rights have usually focused on the ownership of individual species and/or specific landscape, especially if the resource is marketable. For instance, fisheries are a well-known example of an open-access type regime (The Gordon-Clark extinction models used fisheries to demonstrate over-harvesting). In New Zealand, an individual tradeable quota (ITQ's) management system is now used to assign property rights to fishers. ITQ's enable market forces to direct the allocation of fishing resources to the most efficient use (Berkes, 1996; Hartley, 1997). Moreover, conservation rather than over use is now the objective (The Natural Resource Unit, 1991).

Although private property can achieve conservation objectives and is efficient, there are problems with this regime. Much of biodiversity is a public good. Pure public goods are those whose consumption does not reduce the quantity available to others to consume; they are therefore ubiquitous and cannot be depleted (McKean, 1996). Further, much of the value of biodiversity (in particular its information value and ecological value) is diffuse and intangible (Box 3.2).

> *In general, property right institutions operate well if the flow of goods and services is densely concentrated in at least one 'dimension' at one point in that flow. Then it is possible to delineate and segregate the investor's flow from others'....Many assets are not of this nature. Some have the characteristic that their benefits are instantaneously diffusive, so that investments in the asset generates benefits throughout a wide area. Others are of the character that their benefits diffuse rapidly and it is costly to segregate between beneficiaries and non-beneficiaries of the flow. This is the general nature of assets that are not easily subjected to property rights institutions* (Swanson, 1995c, 164).

Box 3.2: Arrow's Fundamental Paradox of Information.

Information is a global public good. Information as a product has an innate capacity for diffusion, as remarked on in Arrow's Fundamental Paradox of Information. The paradox states that information is not marketable until it is revealed (because its value is unknowable prior to revelation) whereas the consumer's willingness to pay can be concealed after revelation of the information (because the transfer has already occurred). In addition, information is often revealed on the mere inspection of a tangible product within which it is embedded. Therefore, the mere act of marketing of a product created from useful information often releases that information to the world, rendering it far less valuable.

Source: Swanson, 1995c, 165.

Moreover, many species are legally protected from hunting and trade. Hence, legal markets are non-existent. This will reduce incentives for private owners to invest in conservation. Even in situations where biological resources are valuable and marketable, owners may not capture all the values of the resource and this may reduce the incentive to invest in protection. However, as long as the costs of maintenance and protection are less than the economic return, owners will probably invest in conservation.

Moran (1992) lists three additional reasons why biodiversity may not be protected privately. These are that: species may have no present use value, or may have a negative value, or it has not yet been discovered. Naughton-Treves and Sanderson (1995) present an argument why privatisation of wildlife may fail. Many wild species are classified as fugitive resources; they migrate over large areas of land and sea and across local community and international boundaries. The species is therefore difficult and expensive to monitor and protect. Owners may then harvest the resources as quickly as possible before the resources moves outside the local area or off their property (Naughton-Treves and Sanderson, 1995). This is also a problem for common property situations.

Bromley (1989, 1991, 1997), also outlines arguments against private property. These include the following points: First, private property may not be economically efficient. For example:

> *in parts of Latin America over 80 percent of the land is owned by as few as 5 percent of the families... Additionally, we are often told that private property leads to the "highest and best use of land." With large segments of Latin America's best agricultural land devoted to cattle ranching while food crops fight for survival on steep and rocky mountain sides, skeptics should be excused if they challenge that particular conclusion* (Bromley, 1991, 25).

This however may be a result of perverse incentives rather than the fault of private property. As well, private owners may lack information, especially on all the values of biodiversity and so they may convert base resources into alternative uses. Further, high transaction costs may preclude bargaining therefore property rights may not go to the highest valued use. Second, Bromley (1991) points out that what private owners want to do with the resource they own may not be compatible with the current social institutions. That is, as owners it is their prerogative with regard to how they should use their land and resources. However, in order to survive and get the best value for their resource, private owners have to produce what other people (society) demand (Pejovich, 1997). Finally, it is impractical to assign private property rights to all biodiversity. In many instances only commercially valuable game species are privately managed. For these reasons it is assumed private owners will protect too little biodiversity.

3.3.4 State property

State property is where ownership is held by the state through various government agencies (Bromley, 1997). In many countries (developed and developing) the state manages a large percentage of the sea and land area; protected areas, national parks, and forests are example of state-owned natural resources. The state legislates the rules and regulations over access and the utilisation of the natural resources they own; it is a top-down approach to conservation. In many instance the state may "take over" management from existing property regimes (Box 3.3).

Box 3.3: Nationalisation results in degradation.

Nationalisation of resources, once a popular approach in many postcolonial countries, has resulted in social dislocation and resource degradation. In a move to curb deforestation, the government of Nepal nationalised forests in 1957, converting what were often communal forests into state property. But the result was the creation of *de facto* open-access. Villagers whose control of nearby forests had been removed, now viewed the state forests as an "ownerless" resource open to anyone's exploitation. Deforestation accelerated; during 1964-1985, Nepal lost approximately 570,000 hectares of forestland. In the face of worsening conditions, the government reversed its policy and began in 1976 to re-create communal property rights.

Sources: Berkes *et al.*, 1989, 92; Bromley, 1991, 23, 152-156; Pye-Smith *et al.*, 1994, 17-36; Berkes, 1996, 97; Freese, 1998, 84.

Although the state owns and manages these areas and resources individuals, groups, and firms (who abide by the laws) have rights to use and/or lease those resources (Bromley, 1997). For example, in India,

> *[t]he "tree growing associations" in West Bengal, consisting of groups of landless or marginal farmers who are given a block of marginal public land for tree planting, are examples of usufructuary rights. The members are not granted titles in land, but the group is given usufruct rights on the land and ownership rights of its produce* (Cernea, 1985 cited in Bromley, 1991, 23).

Nevertheless, many people have been banned from harvesting traditionally utilised resources and state ownership has been shown to be inefficient in many instances.

For effective management to occur certain conditions need to be in place (Furze *et al.*, 1996). This includes, a legitimate government governing in the best interest of the people, with minimum corruption and effective administration capacity. Infrastructure and procedures to efficiently manage and enforce conservation policy need to be operational and well functioning. If these conditions are weak or insufficient, the government may fail to protect biodiversity and resource degradation may occur (Bromley, 1997). Importantly there must be sufficient revenue to fund conservation objectives.

In developed countries, state control has been reasonably successful in the management of national parks and protection of biodiversity. This is because the above mentioned conditions and institutions are relatively effective and efficient. Importantly, although the state owns the national parks, people are allowed to make use of the biological resources within these areas. Many private individuals and firms such as camping and hunting outfitters derive a positive economic returns from utilising these resources without actually owning them. In North America (United States and Canada) this amounts to more than $US70 billion annually for the private sector (Geist, 1994). This creates incentive for the private sector to monitor and enforce the rules and regulations for the protection of North American biodiversity.

Developing countries are however a very different matter. Although these countries have their own conservation laws, protected areas, and state ownership over natural resources, most lack the ability to control and regulate access to these areas and to biodiversity (Barbier, 1992a). For instance, according to Metcalfe (1995, 274), 'privatisation of the African rangelands is often not feasible and state control has proven inadequate.' Furthermore, with the breakdown of traditional or customary laws and the implementation of culturally inappropriate and incompatible legal system, abuse and exploitation of biodiversity occurs. For example, in Kenya where the government owns and manages the national parks, poachers and communities encroaching on national parks are depleting wildlife (Moran, 1992). The state has been ineffective in protecting biodiversity; a *de facto* open-access situation exists (also see Box 3.3).

Moreover, many local communities now have no active and participatory involvement in the management of these resources. In consequence, they no longer gain legal and social benefits from resources they traditionally once used. Hence, they have reduced incentives to conserve local biodiversity. This issue is explored in detail in Chapters 4 and 5.

Panayotou and Ashton 1992 cited in Ostrom and Schlager (1996) identified six reasons why national governments have been powerless to enforce their claims to ownership and thus achieve an enforceable set of rights. These include:

1. the vastness of the area transferred to state ownership;
2. the speed and manner in which the transfer of ownership has been made;
3. the failure to recognise and accommodate the customary rights of individuals and communities, which has created resentment among local populations;
4. the limited budget and administrative, technical, and enforceable capacities of the newly established estates;
5. growing pressures from expanding rural populations; and
6. the failure of rural development to provide alternative employment and income opportunities.

In short, the large areas of natural habitat managed by the state, combined with underfunding and bureaucratic constraints and procedures have limited the state's ability to effectively manage protected areas and the biological resources within them.

3.3.5 Other problems with private and state property

A problem with private and state property rights is that often these regimes do not take into account the interrelatedness and interdependence of the entire ecosystem. Naughton-Treves and Sanderson (1995, 1267), argue that, 'the proprietary arrangements governing human appropriation or preservation of wildlife are not necessarily consistent with those required to protect ecosystems.' That is, property rights are most often associated with land or individual species not with biodiversity overall. Further, private ownership rights to resources are usually parcelled into units smaller than ecological boundaries (McKean, 1996). This may create negative spillover effects or externalities (Bromley, 1991; McKean, 1996).

If ecosystems are not protected over time the systems resilience could be undermined with future negative consequences for wildlife and humans. Resilience can be defined as the ability of ecosystems to resist disturbances and the speed they return back to a steady-state equilibrium (Holling *et al.*, 1995). An alternative approach would be to assign property rights based on ecological rather than anthropocentric principles. However the feasibility of this approach is not investigated in this study.

Pejovich (1997) provides a critique of state ownership. Public decision-makers may have weak incentives to invest time and resources identifying the highest valued uses for state owned resources. Rather, they have strong incentives to invest time and resources seeking benefits for themselves. In short, according to Bromley (1989, 873), 'the record of state management of natural resources has been very disappointing.'

3.3.6 Summary: private and state property

Although large areas of land and sea are under state and private ownership there will probably not be enough natural habitats conserved to protect the diverse range of biological resources that exist. Moreover, with state ownership largely unsuccessful in developing countries other strategies need to be developed and implemented. In addition, institutions for the efficient functioning of private property rights are often limited, incomplete, or thin in developing countries. Furthermore, there is a growing awareness that for successful protection of biodiversity a much broader view must be taken. Common property or communal ownership rights have been suggested as an alternative conservation strategy (Runge, 1986; McNeely, 1988; Berkes *et al.*, 1989; Bromley, 1991; Arora, 1994; Thomas-Slayter, 1994; Berkes, 1996; McKean, 1996).

> *The establishment of strong community village level institutions can be the single most effective incentive for behaviour which contributes to conservation of biological resources* (McNeely, 1988, 74).

3.3.7 Common property regimes (res communes)

Fundamentally common property is similar to private property (McKean, 1996; Bromley, 1997). That is, there is a well-defined group of owners who share resources and who can exclude non-owners from using those resources (Bromley, 1991). However, unlike private property owners, members ordinarily have no rights to sell the property right (Pejovich, 1997). Examples of common property regimes include tribal/indigenous groups, villages, neighbourhoods, and clubs (Bromley, 1991).

Increasingly indigenous resource management systems such as communal property are being recognised as legitimate. Other options also include co-management approaches between the state and local communities in managing wilderness areas and wildlife (Arora, 1994; Metcalfe, 1995; Berkes, 1996), as well as cooperative initiatives between local communities and private enterprise (Lewis and Alpert, 1997).

A growing trend in conservation is towards a more participatory and co-management type of approach (Metcalfe, 1995; Berkes, 1996; McKean, 1996). That is, a bottom-up decentralised approach to conservation involving all stakeholders. Land and customary resource rights are being returned to indigenous people. For example, in Australia, indigenous Aborigines are sharing management of protected land with the state.[2] Many states in India are introducing joint forest management systems (JFMS) between the state and local people.

> *The Joint Forest Management regulations have, in that sense, attempted to 'induce' people's participation in forest protection by making protection an economically rewarding activity from the viewpoint of the people. This, it was realised, could also be the most cost-effective method of forest management from the viewpoint of the state. Its implications for the overall state of development could be very far-reaching too* (Arora, 1994, 692).

Moreover, the WCED, the United Nations Conference on Environment and Development (UNCED), the CBD, *Agenda 21* and the IUCN all acknowledge indigenous resource management systems (Box 3.4). A criticism of some these multilateral Conventions is that while they acknowledge community and local level participation they still maintain a centralised top-down approach to development.

Returning property rights to indigenous people under common ownership has not resulted in open-access exploitation of local biological resources; rather conservation is often enhanced. For example, in contrast to Kenya, in Zimbabwe wildlife is flourishing under the CAMPFIRE model with CAMPFIRE areas now covering a large proportion of Zimbabwe's natural habitat.

Common property regimes offer certain other advantages and efficiencies in resource management. First, incentives to protect rather than exploit the resource increase, consequently enforcement costs are reduced. That is, if exploiters can be turned into resource protectors, they can band together collectively to protect their resources. Second, common property rights can privatise rights to things without dividing the things into pieces. So common property can reap the advantages of private

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[2] See Furze *et al.*, 1996, 139-145, joint management at Uluru-Kata Tjuta National Park.

property while retaining the overall wholeness of the resource. That is, private property type externalities are internalised. Hence, ecological boundaries remain intact and ecological services and functions are maintained (McKean, 1996). Finally, communal property rights regimes may be more efficient where productivity is low and costs of establishing and enforcing private property rights are high (Freese, 1998).

Box 3.4: International Conventions and agreements recognising and acknowledging indigenous peoples rights.

WCED: These communities are the repositories of vast accumulations of traditional knowledge and experience that links humanity with its ancient origins. Their disappearance is a loss for the larger society, which could learn a great deal from their traditional skills in sustainably managing very complex ecological systems. Source: WCED, 1987.

Agenda 21: Chapter 3 of *Agenda 21* puts much emphasis on local solutions to both environmental and development problems. Indigenous people have developed over many generations a holistic traditional scientific knowledge of their lands, natural resources and environment...[States must] recognise and foster the traditional methods and the knowledge of indigenous people and their communities...and ensure the opportunity for the participation of those groups in the economic and commercial benefits derived from the use of such traditional methods and knowledge. Source: *Agenda 21*, cited in Robinson, 1993, 267.

CBD: [States must] respect, preserve and maintain knowledge, innovations and practices of indigenous and local communities embodying traditional lifestyles relevant for the conservation and sustainable use of biological diversity and promote their wider application with the approval and involvement of the holders of such knowledge, innovations and practices and encourage the equitable sharing of the benefits arising from the utilisation of such knowledge, innovations and practices. Source: CBD, cited in Mead, 1995, 3.

There are problems with common property. Population growth, technology advancements (such as modern firearms) and economic changes (such as wage employment) can all contribute to the breakdown of communal property regimes (Naughton-Treves and Sanderson, 1995; Berkes, 1996). Further, as the benefits are shared between all members of the community or group, individual behaviour may not be rewarded. For example:

> *An important finding is that by giving quasi-public goods to communities to abstain from hunting, programs fail to reward individual behavior. Consequently, programs create a free-rider problem in which individuals continue to hunt while receiving the benefits of community-level projects* (Gibson and Marks, 1995, 942).

There could also be high transaction costs as decisions may require consensus from all co-owners. However, Bromley (1991) argues that in many traditional societies such meetings and decision-making processes have great social significance. They may be viewed as benefits to the community and co-owners rather than treated as a cost.

In summary, other alternative and/or complimentary approach's to protect biodiversity are needed. The most successful appear to be those that are best suited to the specific ecological and social environments. One option gaining recognition is restoring common property rights over natural resources back to local communities and indigenous peoples. This is of particular relevance to developing countries.

Indigenous people have historically protected biodiversity through common property regimes. The history of economic development (especially from the enclosure movement onwards) has replaced this style of resource management with state and private property rights. These approaches have failed to adequately protect biodiversity. Moreover, as previously stated in developing countries the institutions and infrastructure to effectively enforce and protect state and private property are lacking and weak. In all, creating conditions for an open access regime.

3.3.8 Non-property rights or open-access (res nullius)

In open-access situations property rights are not enforceable. Open-access is where resources can not be owned (Bromley, 1997). An open-access situation is one of mutual privilege and no rights (Bromley, 1989). That is, anyone can have access too and use these resources without concern for other user rights. In short, open-access is not a property as it cannot yield an appropriable benefit stream until it is in the physical possession of an individual or group (Bromley, 1991). Hence, only the rule of first capture; the "first in" secures the property stream from the resources (Bromley, 1997). As a consequence no effort will be made to maintain or improve that resource as others can gain with no effort (Moran, 1992). Moreover, the resource will tend to be over-used and over-exploited, the familiar "tragedy of the commons."

Open-access may also result when there is a breakdown in existing management systems. For instance, the enforcement of the rules and regulations governing access and use of resources in national parks is virtuality non-existent in many developing countries (Box 3.5, cf. Box 3.1). In addition, the breakdown in authority affects feedback loops that previously help stabilise the over-harvesting and degradation of the resource (Berkes, 1996). In open-access there are no institutions to respond to such signals from the resource. In these instances sustainability cannot be maintained in the long-run, depletion occurs and extinction is a possibility.

Box 3.5: Tragedy of the commons in Indonesia.

In Indonesia, the government declared in 1967 that it had sole legal jurisdiction over the nation's forests-74 percent of the land area. Customary rights, which had evolved as a complex and sustainable management system over many generations, were not legally recognised. As elsewhere, by removing power from local communities, a real life "tragedy of the commons" was created. The government, which has the authority, is unable to police the nation's vast forests, and the communities who are in the forest have no power to stop exploitation by outsiders. Little of the economic benefits from forest exploitation in Indonesia or elsewhere return to the communities who lost access to forest resources. In fact, their standard of living has declined.

Source: Abramovitz, 1998, 29.

3.3.9 Summary: Property rights and biodiversity

For effective and efficient protection of biodiversity will require all three enforceable property right regimes. In open-access situations property institutions must be changed to state, private, or common property before there will be investment (Bromley, 1991). Further, protection and maintenance of biodiversity will also require co-ordination, co-operation, and institutional change at the local, national, and international level. For instance, at the local level local customary laws and common property regimes must be strengthen and enforced. At the national level the government must enter conservation partnerships with local people as well as carry out macroeconomic reform, such as removing barriers to trade in wild species. At the international level free and legal trade in biological resources rather than trade bans can enhance conservation. Moreover, ownership rights must be legally recognised and enforced.

3.4 CONCLUSION

Property regimes are important and necessary to protect and efficiently use biological resources. Equally important are strong institutions to enforce the conditions of rights and duties. Current thinking and practise is that state ownership is the most effective regime to conserve biodiversity. This is a consequence of the misinterpretation of Hardin's "tragedy of the commons." Contemporary conservation strategies practised by the state include protected areas and trade bans (the subject of the next chapter). Although these strategies have had some success, biodiversity is still being lost and degraded. In addition, they have removed or weakened many existing property right institutions. In turn the degradation and loss of biodiversity continues.

As stated, the key to long-term sustainable management is stable ownership (Mendelsohn and Balick, 1995). Hence, appropriate property right regimes and institutions are necessary. Importantly local communities must actively participate in conservation management. Many communities have developed and maintained successful sustainable resource management for thousands of years. The use of this knowledge is crucial to protecting biodiversity. Re-establishing common property regimes through strengthening traditional resource rights are seen as a logical step towards protecting biodiversity, especially in developing countries. This chapter has shown that in particular settings common property can be just as effective and efficient as state and private property in protecting and conserving biodiversity.

However, what is important is not the property regime, but the establishment of a regime that can achieve better gains and benefits for biodiversity and people than other types of management. This may involve a full or partial transfer of rights between the local, state, and international levels. It will require a genuine partnership between all stakeholders, incorporating top-down and bottom-up approaches.

Essentially no matter what property right regime is in force, biodiversity will only be conserved if it is the owner(s) interest to do so. Members of communities determine the institutions and deem what behaviour is acceptable. Ultimately if they want to protect and wisely use biodiversity they must accept those institutions that achieve this objective. Incentives, such as a secure income stream will be a determining factor in regard to what and how much biodiversity is protected.

4

Conservation Strategies: Protected Areas & Trade Regulations.

***We are reaching a point at which traditional means of conservation, in the familiar guise of protected areas and endangered species recovery programs, are no longer adequate* (Freese, 1998, 1).**

4.1 INTRODUCTION

Genetic resources, species, and ecosystems provide a variety of natural goods and services. Many of these are valuable to human society. The exploitation of biodiversity however has generally remained unregulated until it is perceived to be threatened. Biodiversity may then be protected, especially if the threat has direct consequences to the welfare of humans. One of the most significant threats to biodiversity today is the expansion of human economic activity into natural areas. That is, biodiversity loss is primarily a result of land conversion, specialisation, and degradation of natural habitats (Swanson, 1994a).

This loss of biodiversity has led to public and private expressions of concern at local, national, and international levels. In response, a number of strategies and laws to protect natural land/seascapes and to preserve remaining populations of threatened species have been introduced. To date, protected areas and trade restrictions are the most common conservation strategies.

4.1.1 Chapter outline

This chapter will analyse the conservation strategies regularly employed by conservation agencies. The protected area strategy is reviewed first in Section 2. In this section discussion focuses on the reasons for establishing protected areas, what is meant by protected area, and associated issues and problems in protected area management. In Section 3, trade restrictions as a strategy will be examined. Specifically CITES a wildlife trade treaty is analysed. Comments on its effectiveness, limitations, and criticisms are given. These analyses are elaborated upon in the following discussion.

4.2 TRADITIONAL CONSERVATION STRATEGIES: PROTECTED AREAS

Jordan (1995, 3) defines conservation as, 'a philosophy of managing the environment in a way that does not despoil, exhaust or extinguish.' Generally, two main schools of thought have dominated conservation arguments.[1] First, is the stewardship ethic that wishes to manage natural resources *wisely* to mitigate future scarcity. This philosophy is based on instrumental and utilitarian principles (Randall, 1991). The motive is to maintain biological resources for their current direct and indirect uses and services or amenity value as well as insurance for future usefulness (Randall, 1991).

The other group is environmentalists who wish to *preserve* nature for its own sake, regardless of any utility to humans. They believe humans have a moral and ethic duty to protect and preserve biodiversity for its intrinsic value (Randall, 1991; Raven,

[1] For an overview on the history of conservation see, Jordan, 1995; Wright and Mattson, 1996.

1995). Both points of view have an emphasis on the establishment and maintenance of protected areas.

4.2.1 Protected areas

Scarcity is one of the reasons for the initiation of laws and regulations governing the use of remaining natural resources. Establishing national parks and reserves are the most well-known approaches used for the conservation of species and/or ecosystems. The protection and maintenance of biodiversity is of concern in this study. Halvorson (1996, 21) defines a national park as,

> *an extensive natural area that is protected from exploitation, protected from occupation, is the responsibility of the national government, and is publicly owned.*

The objective of protected areas is to retain the diversity of biological elements and ecological processes (Meffe and Carroll, 1994c). In theory these areas should be as large as possible (Leader-Williams and Albon, 1988).

Many national parks are established in "unpopulated" areas and/or marginal lands or land not "claimed" or where ownership is not recognised by the state (Naughton-Treves and Sanderson, 1995). Further, the creation of protected areas usually ignores ecological and biological factors (Meffe and Carroll, 1994c).

> *[M]ost parks and wilderness areas were set aside because they were pretty, and because they had little of value to mining, timber, and cattle interests; biological factors, Soulé points out, were generally not considered* (Mann and Plummer, 1993, 1869).

Moreover, parks and reserves were often established for anthropocentric reasons (Erwin, 1991). They were created to prevent the extinction of useful wildlife resources. These include "charismatic" large land mammals or plants with pharmaceutical properties (Erwin, 1991). Further, many were for the exclusive use of privileged groups in society. These groups would attribute high values to species that were valued for recreational hunting (Naughton-Treves and Sanderson, 1995). In addition, gamekeepers were employed to manage these reserves and to keep other users (possibly previous legal users) out ensuring that there was no illegal poaching. For instance, in Europe from the Middle Ages on, exclusive reserves for hunting game animals were created for royalty and gentry (Meffe and Carroll, 1994a; Wright and Mattson, 1996). The reserves were a response to the loss of forested areas and game animals due to land conversion.

The underlying causes of conversion were human population growth pressure, the enclosure of land, and the commercialisation of agriculture.

The modern concept of national parks originated in North America during the 19th Century (Wright and Mattson, 1996). In general the state assumed ownership, control, and management over those areas, usually through a wildlife or national parks department. Rules and regulations govern the public's use and non-use of resources within these areas. Fences marked the boundaries of the parks and reserves in many instances.

An important element of national parks was that people were not allowed to live permanently in the area (McNeely *et al.*, 1994). Protected areas were seen as separate from surrounding areas and from the economy (Moyle, 1998a). Many local people were forcibly relocated with little real compensation for the loss of traditional rights, access, livelihood, and resources (Barbier, 1992b). In effect the state was assuming that it was the appropriate authority and better at managing these lands than any other authority, especially local people (Barbier, 1992b). This model of conservation has spread throughout the developed and developing worlds. In Africa for example:

> *The history of wildlife conservation efforts in Africa has been dominated by a universal approach of divorcing local communities from any control or rights of exploitation of their wildlife, coupled with law enforcement efforts by the central and local authorities (notably the national parks and wildlife departments) to protect the wildlife* (Barbier *et al.*, 1990, 142).

As economic growth and development accelerated, governments recognised the importance of legally protected areas. Accordingly, the majority of parks and reserves (approximately 75 percent) were established from the 1960's (McNeely *et al.*, 1994; Harris *et al.*, 1996). Around two-thirds of this growth is in developing countries (McNeely *et al*, 1990) (Table 4.1). However, many parks exist on paper and are unprotected on the ground (Tuxill and Baxter, 1998).

Further, it was realised how valuable these areas are in providing essential ecological processes. Ecological services and functions include *inter alia* watershed protection, photosynthesis, climate regulation, and soil productivity (McNeely *et al.*, 1990). Consequently,

> *[t]he ability of the earth to maintain ecological integrity has replaced the supply of resources as the central problem of conservation* (Jordan, 1995, 25).

Nevertheless, the quantity of biological resources and ecological quality are interconnected. Many species, in particular invertebrates and microorganisms are essential to the well-functioning of these services. Hence, the quantity and diversity of biological species is still important.

Table 4.1: The development of the protected areas system of the world by CNPPA region.

	% area established up to 1962	**% area established 1962-1971**	**% area established 1972-1981**	**% area established 1982-1991**	**Date of establishment unknown**	**Total area designated**
North America	19.2	5.1	70.1	5.6	99,367	2,560,500
Europe	15.4	20.9	36.1	27.6	12,918	460,671
North Africa and Middle East	1.4	9.1	28.2	61.4	8,630	440,725
East Asia	7.4	3.3	25.5	63.8	19,574	427,414
North Eurasia	17.0	12.3	19.9	50.8	0	237,956
Sub-Saharan Africa	42.5	24.1	17.1	16.3	13,620	1,247,998
South and Southeast Asia	7.5	4.4	53.5	34.6	9,539	487,435
Pacific	52.4	6.1	26.8	14.8	47	4,857
Australia	4.1	17.3	74.9	3.8	143,057	814,113
Antarctic/ New Zealand	73.7	11.3	10.9	4.1	2,033	34,334
Central America	1.3	1.4	21.4	75.9	314	45,869
Caribbean	16.5	6.3	12.4	64.8	4,911	22,857
South America	10.1	14.6	37.8	37.4	3,313	1,145,891
Total	**14.3**	**9.7**	**38.0**	**38.0**	**317,322**	**7,930,629**

Notes: Data in square kilometres. Minimum size for inclusion is 10 sq km, except for sites in the Pacific and the Caribbean where it is 1 sq km.
Source: McNeely *et al.*, 1994, 11.

In addition to national parks there are other types of protected areas. The IUCN defines a protected area as:

An area of land/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means (Furze *et al.*, 1997, 26).

The IUCN through its Commission on National Parks and Protected Areas (CNPPA) has developed a system of classification for the different types of protected areas. This is presented below (Table 4.3). Further, game management areas (GMA's) are extensively used throughout Africa. GMA's are semiprotected areas that allow some human settlement and wildlife utilisation such as hunting. GMA's can extend the available habitat for many wild species. For instance, in Zambia while 9 percent of land is gazetted as national parks some 22 percent is classified as GMA's (Leader-Williams and Albon, 1988; Lewis and Alpert, 1997).

Although each of these protected areas has the same general purpose and definition they can have different management objectives (Table 4.2).

Table 4.2: Management objectives of protected areas.

	Category						
	IA	**IB**	**II**	**III**	**IV**	**V**	**VI**
Management Objective							
Scientific research	1	3	2	2	2	2	3
Wilderness protection	2	1	2	3	3	-	2
Preservation of species and genetic diversity	1	2	1	1	1	2	1
Maintenance of environmental services	2	1	1	-	1	2	1
Protection of specific natural/cultural features	-	-	2	1	3	1	3
Tourism and recreation	-	2	1	1	3	1	3
Education	-	-	2	2	2	2	3
Sustainable use of resources from natural ecosystems	-	3	3	-	2	2	1
Maintenance of cultural/traditional attributes	-	-	-	-	-	1	2

Key: see Table 4.3 for category classifications.
1 Primary objective
2 Secondary objective
3 Potentially applicable objective
\- Not applicable

Source: IUCN 1994 from Furze *et al.*, 1997, 27.

Table 4.3: Types of protected areas by IUCN category.

Category	*Type*	*Objective*
I	Strict Nature Reserve/ Wilderness Area	Areas of land and/or sea possessing some outstanding or representative ecosystems, geological or physiological features and/or species, available primarily for scientific research and/or environmental monitoring; or large areas of unmodified or slightly modified land, and/or sea, retaining their natural character and influence, without permanent or significant habitation, which are protected and managed so as to preserve their natural condition.
II	National Park	Protected areas managed mainly for ecosystem conservation and recreation. Natural areas of land and/or sea, designated to (a) protect the ecological integrity of one or more ecosystems for this and future generations, (b) exclude exploitation or occupation inimical to the purpose of designation of the area and (c) provide a foundation for spiritual, scientific, educational, recreational and visitor opportunities, all of which must be environmentally and culturally compatible.
III	Natural Monument	Protected areas managed mainly for conservation of specific features. Areas containing one, or more, specific natural or natural/cultural feature which is of outstanding or unique value because of its inherent rarity, representative or aesthetic qualities or cultural significance.
IV	Habitat/Species Management Area	Protected areas managed mainly for conservation through management intervention. Areas of land and/or sea subject to active intervention for management purposes so as to ensure the maintenance of habitats and/or to meet the requirements of specific species.
V	Protected Landscape/Seascape	Protected areas managed mainly for landscape/seascape conservation and recreation. Areas of land, with coast and sea appropriate, where the interaction of people and nature over time has produced an area of distinct character with significant aesthetic, cultural and/or ecological value, and often with high biological diversity. Safeguarding the integrity of this traditional interaction is vital to the protection, maintenance and evolution of such an area.
VI	Managed Resource Protected Area	Protected areas managed mainly for the sustainable use of natural ecosystems. Areas containing predominantly unmodified natural systems, managed to ensure long term protection and maintenance of biological diversity, while providing at the same time a sustainable flow of natural products and services to meet community needs.

Source: McNeely *et al.*, 1994, 10.

Currently there are over 8000 sites covering 7,928,928 km^2 or approximately 5 percent of the Earth's land area designated as protected that adhere to the IUCN categories I-V (McNeely *et al.*, 1994) (Table 4.4). In addition, 115 countries have established an aggregate 1,328 sites (3,061,300 km^2) that are marine or coastal protected areas (UNEP, 1993). The United Nations (UN) through the United Nations Educational

Scientific and Cultural Organisation (UNESCO) also recognise areas of significance. These include *Biosphere Reserves* (Box 4.1) and *World Heritage Sites* (McNeely *et al.*, 1994; Furze *et al.*, 1996).

Box 4.1: What are Biosphere Reserves.[2]

Biosphere reserves are geographic areas considered typical of the balanced relationship between people and nature. As of April 1997, 337 Reserves in 85 countries met the criteria for this designation laid down within UNESCO's "Man and the Biosphere" (MAB) Program. They combine three functions:

- safeguarding samples of the Earth's landscapes, plant and animal species and ecosystems;
- fostering economic development that is ecologically and culturally sustainable; and
- providing support for research, monitoring, training and education relating to local, regional and global conservation issues.

Biosphere Reserves are organised around a core area surrounded by a buffer zone and a transition area. Biosphere Reserves have a common interest in seeking concrete solutions to reconcile the conservation of biodiversity with the sustainable use of natural resources for the benefit of local people.

Source: Bequette, 1998a, 1998b.

[2] **A list of contacts for Biosphere Reserves and the guiding documents for Biosphere Reserves are available on the Internet URL: http://www.unesco.org:80//mab/themabnet.html**

Table 4.4: Numbers and combined areas of reserves at least 1000 hectares in size in IUCN protected area categories I-III (i.e., strict nature reserves, national parks, and natural monuments) by region, and percentage of region protected.

Region	Number of reserves	Area (ha)	Region protected (%)
North and Central America	610	170,344,290	7.25
South America	289	58,190,622	3.27
Africa	260	88,722,877	2.93
Asia (including Russia)	585	59,305,756	1.32
Europe	289	8,056,879	0.77
Australia and South Pacific	443	67,872,385	8.79
Antarctica	12	220,649	0.02
Total	**2,488**	**452,713,458**	**3.04**

Source: Noss, 1996, 95. (Adapted from World Resources Institute, World Conservation Union, and UNEP. *Global Biodiversity Strategy* Washington DC: World Resources Institute, 1992.

These statistics however understate the area of protection while ignoring quality aspects. For instance, many small protected areas and privately owned protected areas are not included (all examples from McNeely *et al.*, 1994 unless referenced otherwise). For example, in Sweden, there are 1,200 small nature reserves (<1,000 ha) that together total 430,000 ha. New Zealand has almost 600 protected areas in reserves or convenants over private land, covering more than 20,000 ha. In Europe, several non-government organisations (NGO's) own nature reserves. In the United States the largest private owner of nature reserves in the world is The Nature Conservancy, consisting of 1300 reserves encompassing 526,000 ha and 1725 rare species and communities (Murray, 1995). Hence, the official aggregate size of protected areas will be underestimated.

There is growing criticism of the benefit, role and management of traditionally managed protected areas. Importantly, the assortment of protected areas does not take account of the range of species or land/seascape diversity. Consequently, conservation of global biodiversity is not accurately represented (Naughton-Treves and Sanderson, 1995; Moyle, 1998a). McNeely *et al.* (1990, 60) point out that,

> *[r]ecent advances in conservation biology have shown that by themselves the strictly protected categories (I, II, III) will not be able to conserve all, or even most species, genetic resources, and ecological processes. Far greater expanses are required for conservation than modern societies are willing to remove from direct production. The best answer to this dilemma is to design and*

manage different types of protected areas –including very large expanses in the categories that permit, and even encourage, compatible human uses of resources.

Nevertheless, legally protected areas are growing (Table 4.1). Many countries have set conservation targets. The WCED (1987) have called upon all nations to set aside 12 percent of their land area as protected areas. There is criticism's of setting conservation targets. Such targets are often set for political expediency rather than for ecological reasons. According to Soulé and Sanjayan (1998) targets based on ecological knowledge would be much higher but would be politically unacceptable in many nations. In the few detailed studies available the typical estimate of the land area required to represent and protect most elements of diversity is about 50 percent (Table 4.5).

For example, a controversial conservation plan, the Wildlands Project, has been proposed to protect biodiversity in North America (Mann and Plummer, 1993; Jordan, 1995; Noss, 1996). The project calls for a network of wilderness areas, human buffer zones, and wildlife corridors occupying as much as half the continent (Jordan, 1995). In the State of Oregon 23.4 percent of land would be returned to wilderness while restrictions on human use would apply to another 26.2 percent (Mann and Plummer, 1993). The concept and framework of this plan if accepted could have extensive consequences for developing countries regarding how much habitat they should reserve as protected areas.

Table 4.5: Estimates of minimum areas for the protection of biodiversity.
(the objective and criteria of these studies varied).

Area required (%)	Region	Goal
45	Australian river valleys	To contain all plant species.
49	Oregon coast range	To capture regions of high diversity, represent all ecosystems, maintain target species, and provide for connectivity.
75	Norway	To protect all plant species in deciduous forests.
33.3	Florida	To preserve habitats essential for rare and declining species.

Source: Soulé & Sanjayan, 1998, 2061. (Also see Soulé, 1996).

However, it is doubtful if such large expanses of land can be reserved strictly for habitat and species preservation. First, economic and political constraints will limit the size and design of protected areas (Meffe and Carroll, 1994c). Second, for many developing countries the option of preserving natural habitat and managing these areas in the orthodox way is not feasible given the social and economic pressures for increased economic development and poverty alleviation (Barbier, 1992a). These issues will act as constraints on the size and effective management of protected areas, unless the areas and the biological resources within them can be utilised for economic benefits.

4.2.2 Other considerations for protected areas

There are several technical, biological and economic aspects to consider in deciding where and how to establish a protected area. These are discussed in depth below.

The first issue is whether a species or ecosystem is the objective of conservation and protection. The former emphasises the protection of individual (usually endangered) animals. They are species-specific actions. The conservation goal is to protect the quantity of that species in their natural habitat. The protection of landscapes is emphasised in the latter; this can include ecological processes and species diversity (Soulé, 1991). There is debate over the appropriate approach.

There are valid reasons for focusing conservation of biodiversity on individual species. Eisner *et al.* (1995) identifies four reasons: (i) species provide a more objective means of determining the location, size, and spacing of protected areas; (ii) population declines in key species may indicate stress in an ecosystem before it is system wide; (iii) individual species have information value for humans and; (iv) individual species can play a pivotal role in the provision of ecological services.

However, many scientists believe that the species-by-species approach is inadequate (Meffe and Carroll, 1994b; Jordan, 1995). Simply there are too many endangered and threatened species and not enough scientists, funds, and time to adequately study and develop appropriate conservation strategies for each individual species. Further, in many instances recovery programs are not taken until the species is classified as endangered. Often this will be too late to ensure the survival of that species in its natural habitat. Critics also argue that with the species approach, limited conservation funds will be directed to a few charismatic species.

Rather, conservation at the habitat, ecosystem, landscape, or biome level may be a more practical approach to saving the millions of species that constitute global biodiversity (McNeely, 1990; Jordan, 1995). According to Patrick Bourgeron, a senior ecologist with The Nature Conservancy, there is a move away from conservation of species to conservation of ecosystems (Schmidt, 1996). It is now recognised that conserving a portfolio of reserves representing the full spectrum of habitat types is more important than trying to save the maximal number of species (Schmidt, 1996). According to David Olson from the WWF,

> *[t]he [ecosystem] approach is a more rigorous and effective way to do conservation....It buys you almost everything. If you capture all ecosystems in all ecoregions and maintain them, you could save the majority of species* (Schmidt, 1996, 917).

Nevertheless, a species-specific focus can be valuable in achieving conservation objectives especially if it is a keystone species. A keystone species is defined as:

> *One that makes an unusually strong contribution to community structure or processes. A keystone species may be a major predator,... a unique food source,... or a species that maintains critical ecosystem processes,... their critical feature is that the removal of keystone species can make many other members of the community vulnerable to extinction* (Myers, 1994, 129).

Further,

> *[c]ertain "keystone" and critical species, especially of vertebrates, may be used as diagnostic indicators of the adequacy of the protected area system, it being assumed that if habitats capable of assuring the survival of viable populations are protected, the lesser-known species will also be safeguarded* (McNeely *et al.*, 1990, 61).

However, despite the attractiveness of focusing conservation onto keystone species, it is difficult to make a case that one species is more important than another (Jordan, 1995).

Finally, species are distributed nonrandomly (diversity is concentrated in certain areas) (Meffe and Carroll, 1994b). For example,

> *25% of bird species are found in 5% of the world's land area; protection of 14 of 90 studied localities in Thailand would conserve all hawkmoth species in that country; 16% of the land area of South Africa, if properly selected, would protect 95% of vascular plant species of the region* (Meffe and Carroll, 1994b, 75).

In short, preserving certain areas will bring protection to a large range of species. However, preservation of species and habitats are not mutually exclusive objectives, conservation should be jointly pursued.

The second issue is the physical size of the protected area. Protected areas need to be large enough to maintain viable populations of native species, to accommodate natural disturbances, and to allow species to evolve (Meffe and Carroll, 1994b). Size of protected areas vary from for example, the 65,000 sq. km. Air-Tenere Nature Reserve in Niger and the five sq. km. Monarch Butterfly Overwintering Reserves in Mexico (Wells *et al.*, 1992).

Importantly, wild species need areas of natural habitat to survive. There are two main issues to consider. The first is the minimum area required (MAR) to preserve a species. The second, is the minimum area to maintain a minimum viable population (MVP). MVP is possibly the more useful measure to determine the size of an area to ensure the protection of a species (Jordan, 1995). The goal of MVP is not to maintain a minimum number, but to identify a population size below which extinction is likely, and then to maintain populations well above that size (Meffe and Carroll, 1994c). However, accurate population data over several generations is generally required to accurately model the population (Moyle, 1998b). Only a few population viability analysis (PVA) studies have been carried out on selected species (Mann and Plummer, 1993). Consequently, research, information and data on MVP numbers for most species are lacking.

The main factors to consider with regard to the size of protected area are the physical size of the species and its range. Megafauna species need a habitat in the order of hundreds or thousands of square kilometres. For example, the United States Fish and Wildlife Service (USFWS) recovery plan for the grizzly bear *Ursus arctos* suggests that each animal requires 76 square kilometres of roadless land (Mann and Plummer, 1993). Further, according to Soulé such mammals need populations of several thousand to survive inbreeding, disease, and demographic stochasticity (Mann and Plummer, 1993). In essence, this means setting aside much more protected land than at present and well above the official political conservation targets.

Migrating birds may travel thousands of kilometres and only require many small feeding and resting areas (stepping-stones) strategically located along the migration route. However, transboundary problems and ownership issues arise with migrating birds and to ensure adequate sites are available requires cooperation between the countries that these birds cross. For the protection of plants, reserves of less than 10 ha can be effective in conserving viable populations of some plants, provided their boundaries can be secured (McNeely *et al.,* 1990). However, because of the large number and diversity of bird and plant species and their different biological requirements again not enough habitats will be preserved.

Moreover, protected areas and reserves are usually never "designed", conservationists have to work with what they can (Meffe and Carroll, 1994c). Consequently, these areas tend towards a minimum rather than the maximum size, regardless of ecological and biological modelling. Further, natural die-offs have implications for conservation, the size of protected areas and population viability (Young, 1994).

There are additional problems and constraints with the size of protected areas. The area may not take into account land use changes. For instance, the initial selection and size of a protected area is based on a species needs, this will require that the size and quality of the area remain constant or increase with population growth. As previously stated, in many developing countries land designated as protected is only protected on paper. In consequence, protected areas are often illegally over-exploited and under development pressure. The actual size may be less than the recorded size. For example, official estimates place the standing forest cover of India at 72 million hectares (22 percent of the total land area), but the actual figure is said to be closer to 23 million hectares, or 7 percent (Bromley, 1989).

An empirical method to determine the relationship between number of species and area is the species-area model (see Barrett, 1993; Myers, 1994; Orians, 1994; Eisner *et al.*, 1995; Krautkremer, 1995; Smith *et al.*, 1995; Swanson, 1995b for detailed explanations). This model suggests that the number of species found in an area tends to increase with the size of the area. This model can also be used to estimate species extinction rates. The following determines species diversity (Barrett, 1993).

1. As the sample area is increased, new habitats are encountered, harbouring new species and;
2. immigration and extinction rates determine the number of species found within an area. The number of species in then related to the ease of interchange between areas. While the population size of a species within an area depends on the physical area of the reserve.

The relationship is usually expressed in the form

$$S = cA^{a}$$

Where S is the number of species; A is area; and c and a are parameters. The parameter a can be interpreted as the elasticity of species diversity with respect to area (Barrett, 1993). Estimates for a tend to fall in the range 0.2-0.4 (Eisner *et al.*, 1995). For example, if a = 0.3 then the preservation of 10 percent of habitat will conserve 50 percent of species (Barrett, 1993). Further, a tenfold increase in habitat approximately doubles the number of species within it (Eisner *et al.*, 1995). While the conversion of 90 percent of a natural area will result in the loss of half of its naturally resident species (Swanson, 1995b). Moreover, the final 10 percent conversion will result in a loss of species as great as the initial 90 percent (Swanson, 1995a).

An important significance of the species-area relationship is that land protected on behalf of animals with large home ranges or low population densities will provide *de facto* protection for numerous other species with smaller home ranges or higher densities (Eisner *et al.*, 1995). However, there is criticism of the species-area relationship for ignoring ecological data and low forecasting accuracy (Boecklen and Gotelli, 1984; Swingland, 1993). Nevertheless, it is not doubted that the loss of natural habitat leads directly to the loss of some species.

Additional problems related to the physical size of protected areas are that they may be fragmented and isolated from other natural areas (Meffe and Carroll, 1994c). Consequently, individual areas may not be able to support certain species especially large mobile mammals that need large expanses of habitat. Should some species populations increase above the areas carrying capacity this could have adverse ecological impacts on the habitat and other species. Further, should the increased population spill-over onto the surrounding land, human-species conflicts may increase (Newmark *et al.*, 1994; Dublin *et al.*, 1995).

The consequence of this is that some form of population control may have to be carried out. For instance, in the South African Kruger National Park about 250 elephants per year for the last 30 years have been culled out (TATIS, 1998). However, many environmentalists argue against this on ethical and moral grounds. In Botswana, culling has been put off for fear of international censure (Sharp, 1997). Consequently, in Botswana elephant numbers are very high and causing a range of problem for humans and for plant diversity (Sharp, 1997).

Further, the continued fragmentation and isolation of protected areas increases the edge ratio. This is the ratio between the boundary of a protected area and surrounding areas. If the edge ratio increases this may have negative effects for many species. Especially *forest-interior species* that require interior conditions free from the effects of edges, and *area-sensitive species* that require large expanses of suitable habitat (Harris *et al.*, 1996; Woodroffe and Ginsberg, 1998). Larger edge ratios may also increase human-wildlife contact and conflicts resulting in higher human-induced mortality (Woodroffe and Ginsberg, 1998). Edges also increase the entry area for illegal poaching (Mann and Plummer, 1993). For instance, results from a study have shown that large wide-ranging carnivores[3] are more likely to become extinct by human causes than stochastic processes compared to those species with smaller home ranges (Woodroffe and Ginsberg, 1998). Further, an increase in the edge ratio may help in the spread of disease, pests, predators and invasive plants, for example (Mann and Plummer, 1993; Bonner, 1994).

A recent proposal is wildlife corridors.[4] Wildlife corridors create natural links and connections between protected areas, and act in a way to maximise species numbers and reduce the risk of extinction. The use of corridors allows species and/or populations to move between reserves. This can happen daily, seasonally and in response to catastrophic events or long-term environmental change (Harris *et al.*, 1996). For instance, if global warming alters the biology of habitats species would be able to immigrate along corridors to areas that are more suitable. Importantly corridors allow the outward radiation of species and a continuation of speciation (Erwin, 1991). In short, corridors should allow interchange between populations and increase the species survivability.

However, the corridor approach may have perverse effects. For instance, corridors may increase the edge ratio of a reserve. This will have negative effects on interior type species within the protected area. Moreover, the cost of corridors may be high. If conservation budgets are limited, corridors may not be the best use of these funds. However:

> *The crucial questions are not whether corridors are all good or all bad, not whether they are the sole answer to the present biodiversity crisis, or even whether there are costs and liabilities. Rather, the key question is whether an integrated system of protected and managed natural habitats will be less discriminatory against our rare and endangered native species and also less supportive of alien, exotic, and pest species. Will such an interconnected system of habitats be superior for more natural assemblages of native species and natural levels of ecological processes, such as competition, predation, and parasitism, than a disjunct system of isolated preserves? The consensus of leading conservationists is a resounding yes* (Harris *et al.*, 1996, 184).

Third, budget constraints will also affect the management and enforcement of protected areas. Leader-Williams and Albon (1988), argue that conservation budgets

[3] 10 species were studied.

[4] Also known as, greenways and landscape linkages.

depend on the economic value of each species. The higher the value of the species the greater the investment in conservation. Traditionally managed protected areas reduce the economic value of species. That is, they do not usually allow the harvesting of species for consumptive use. Leader-Williams and Albon (1988) in a study found a direct relationship between rhino numbers and conservation spending. They calculated that at a minimum US$200 per sq. km. was needed (in the 1980's) to prevent organised poaching of elephants and rhino from protected areas in Africa. In most instances developing countries lack the financial means to implement effective conservation plans and strategies. In short, limited conservation budgets will result in inadequate conservation and do not deter illegal harvesting.

There is also the possibility of a principal-agent problem occurring in species recovery work with a limited budget (Moyle, 1998b). Principal-agent problems are a result of the inability of a principal (manager) to monitor the actions of agents (workers). Agents may then decide to pursue their own interests rather than that of the principal (Moyle, 1998b). Moyle (1998b) has constructed a model showing how this problem could emerge in conservation work and that it may have adverse effects and consequences for species recovery. For instance, field workers may decide to concentrate recovery work on species they find charismatic (Moyle, 1998b).

Fourth, social issues are equally important in the success of protected areas (Brandon and Wells, 1992; Meffe and Carroll, 1994c; Furze *et al.,* 1996). In the past social and cultural issues were not deemed important or necessary in the establishment of protected areas, and the conservation authorities were generally unsympathetic to local communities (Barbier, 1992b;Wells *et al.,* 1992). For instance, in the Luangwa Valley Zambia human inhabitants were relocated when four national parks were established in 1938 (Leader-Williams and Albon, 1992).

Critics now claim that parks and reserves are elitist and anti-people, conflicting with the needs and aspirations of local human communities, especially in developing countries (Brandon and Wells, 1992; Noss, 1996). This increases their impoverishment and resentment against conservation and often leads to conflict with park management over wildlife. Many protected areas are now at serious risk, partly because of the social, cultural, and economic hardships they have imposed on members of local communities (Wells *et al.,* 1992). In essence, government conservation policies divorce local people from the responsibility for conservation. They are discouraged from improving the local natural ecosystems and have an incentive to over-exploit the local area and biological resources.

Fifth, conservation does not mean maintaining a given number of a species. Conservation should be concerned with the continuation of the species evolutionary process (Erwin, 1991; Barrett, 1993; Halvorson, 1996). In theory, a population of a species could be maintained *ex-situ* in a minimum area such as a zoo or a seed bank. Some species have recovered *ex situ,* through captive-breeding programs and game farming for example (McNeely *et al.*, 1990). However, there are many problems with *ex situ* conservation. For instance, increased disease and genetic problems are common in captive rearing (Martin, 1997b). *Ex situ* is a difficult and expensive task and only a tiny number of highly valued species will ever be preserved this way (Swanson, 1992a). More importantly *ex situ* does not provide the natural background against which a species can thrive, develop, and evolve against natural predators and competitors (Moran, 1992; Swanson, 1992a). Hence the importance of *in situ* management.

Although the number of protected areas are increasing, few species that are classified as endangered or threatened and are officially protected, have recovered sufficiently to be declassified. In fact, numerous species especially invertebrates remain endangered and this number is growing (OECD, 1991). The notably exceptions are the recovery of the elephant and rhino populations in some southern African states. For example, the white rhino (square-lipped) *Ceratotherium simum cottoni* population declined from 1500 in 1960 to 20 animals in 1989 across five African countries; however in South Africa the population increased tenfold during this same time to number 6000 (Simmons and Kreuter, 1990).

Finally, many endangered species in recovery programs may be regarded as ecologically redundant (Moyle, 1998b). The removal of these species from their natural habitat or very low populations has apparently not undermined the resilience of the ecosystem. That is, they appear not to serve any ecological role of importance. However, these species still receive resources for conservation and species recovery. For instance, the Kakapo *Strigops habroptilus* Psittacidae is now extinct over its natural range and only surviving in small numbers on off-shore islands (Moyle, 1998). In short, it is not possible to maintain all endangered species. Conservation funds should be directed to those species that are ecological important or commercially valuable species whose protection may also cover other species.

4.2.3 Summary: Protected areas

According to Soulé and Simberloff (cited in Noss, 1996, 93):

> *Nature reserves should be as large as possible, and there should be many of them. The question then becomes how large and how many. There is no general answer. For many species, it is likely that there must be vast areas, while for others, smaller sites may suffice so long as they are stringently protected and, in most instances, managed. If there is a target species, then the key criterion is habitat suitability. Suitability requires intensive study, especially in taxa that contain species with narrow habitat requirements.*

Evidence suggests then that considerable areas of land and water need to be protected. In reality, there is not enough of the Earth's surface officially classified as protected, adequately funded and managed to conserve all biological diversity. That is, to meet the goals of conservation. It is impractical to lock away such areas and the biological resources within them and neither is it efficient or effective to ban economic activity within these areas.

New strategies in the management of biodiversity are needed. Importantly protected areas can no longer be seen as separate from the economy. An effective conservation strategy should integrate protected area management, conservation goals and economic development objectives.

> *There is an interdependence both because of the way we manage the economy impacts on the environment, and because environmental quality impacts on the performance of the economy* (Pearce *et al.*, 1989, 4).

Although governments, NGO's and individuals have invested in conservation and protected areas this has been inadequate in many cases. Generally, as wildlife becomes more threatened further controls and restrictions are placed on the use of protected areas. However, these additional protection measures have not improved overall species recovery. In fact, in some circumstances these measures have helped increase the illegal poaching of wildlife from protected areas. This has led to further attempts to control and regulate these activities and the trade in endangered species. In consequence, protection of biodiversity is now a global concern and in response international conservation accords have expanded to manage and regulate biodiversity.

4.3 TRADE MEASURES AS A CONSERVATION STRATEGY

4.3.1 The wildlife trade

Since ancient times wildlife has been traded domestically and internationally (Milner-Gulland and Leader-Williams, 1992b). For instance, ivory has been demanded in Asia for several centuries (Barbier, 1991). However, the modern era is marked by the commercial harvest of species on a large scale linking distant markets (Freese, 1998). Figures for some of the species traded are presented in Table 4.6. However, it is only recently that international trade bans on wildlife have been passed into law. Although there is still a large legal trade in wildlife, the trade restrictions have promoted an illegal trade in banned species.

Table 4.6: Annual world trade in selected wildlife.

Product	At least
Primates (live)	40,000
Birds (live)	3 million
Ornamental fish (mostly freshwater)	350 million
Furs	40 million
Reptile skins	20 million
Reptiles (live)	100 million
Coral (raw)	1,000 tons
Cacti	10 million
Orchids	2 million

Source: Hemley, 1995, 636.

The most valuable and highly traded species originate from developing countries (Wijnstekers, 1995). The major importers and consumers of wildlife products reside in developed countries (Swanson, 1992b). For instance, it is estimated that the USA imported a minimum of 703,000 Neotropical parrots between 1981-85 (Burgess, 1994), while Japan is the main consumer of ivory (Barbier, 1990). In 1994 Colombia exported 446,000 live reptiles and the USA imported 672,000 live reptiles (OECD, 1997).

> *According to the World Wide Fund for Nature, the annual trade in wildlife could be worth as much as $20 billion [US], perhaps a quarter of it illegal. In the US alone, trade in reptile skin products is worth around $250 million a year. And according to TRAFFIC, the organisation that monitors trade in around the world, 1.8 million items of medicine containing wildlife ingredients entered the country between 1984 and 1992. At least 30 per cent of them claimed to contain material from protected species, such as rhinoceros horn and tiger bone* (Pain, 1994, 24).[5]

For those involved in the wildlife trade it can be very profitable. The attraction of harvesting wild species is that the harvests costs are relatively low compared to the retail price. Accordingly, for individuals in developing countries there are large financial gains from trading in wildlife relative to alternative income opportunities. 'Prime crocodile skins can fetch the equivalent of many months conventional income for a peasant farmer' (Simon, 1995, 193). However, it is the intermediaries, importers, corrupt officials and retailers that usually gain the most (Hemley, 1995) (Table 4.7). For example, Caiman skins may cost $US5-$50 to harvest (legally and illegally) but by the time it reaches a retail store in New York as a handbag it may retail for several thousand dollars (Brazaitis *et al.*, 1998). Likewise, rhino horn sells for up to $20,00/kg in some Asian retail markets (Hemley, 1995) and hyacinth macaws *Anodorhychus hyacinthinus* sell for US$8,000 a pair in the USA (Simon, 1995).

Table 4.7: Psittacines from Irian Jaya, Indonesia (early 1990s)

	US$/Bird	% of wholesale Price
Paid to trapper	2.57	1
Paid to trader	7.09	3
Paid to exporter	49.43	19
Wholesale price	256.00	

Source: Freese, 1998, 74.

[5] This figure does not include many wild timber and fish products. The annual value of fuelwood and wood-based products worldwide is estimated at US$418 billion, or nearly 2 percent of the world's gross domestic product (Freese, 1998). Fisheries yield the second most important wild species commodity in the world, with global revenues exceeding US$70 billion per year (Freese, 1998).

In short, such prices create strong financial incentives to harvest and trade in wildlife. The heavy trade in some species has raised concerns of possible extinction. Trade bans and controls are proposed as strategies to counter this threat and save endangered species.

4.3.2 CITES

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) is the largest and well-known treaty relating to trade in wild species (Freese, 1998). The treaty was signed in 1973 and came into force in 1975 (Klyza, 1994). There are 143 countries party to the Convention[6] (TATIS, 1998), with more countries joining each year. Member countries include most of the major wildlife exporting and importing nations (Wijnstekers, 1995). CITES establishes the international legal framework for the prevention of trade in endangered species and for effective regulation of trade in others (Wijnstekers, 1995). By prohibiting trade or allowing controlled trade these actions are intended to stop the extinction of wild species. Hence, CITES can be regarded as a conservation treaty as well (Hemley, 1995).

4.3.3 How CITES works

CITES addresses the issue, of how to prevent the extinction of endangered species from over-exploitation due to international demand. The goal of CITES is 'to regulate the complex wildlife trade by controlling species-specific trade levels on the basis of biological criteria' (Meffe and Carroll, 1994b, 68). The treaty works through a system of regulations and restrictions. When trade is considered a major threat to the survival of a species a total trade ban can be applied. When trade could have a significant effect on the species survival trade controls can be applied.

[6] Countries not party to the Convention do not have to comply with any of the regulations, so trade may continue from or through those countries.

The biological status of the species is important for determining the relevant trade restrictions (Burgess, 1994). Data on the status of species is gathered from a variety of national, international, and non-government agencies. A database of endangered species is maintained by the WCMC based in Cambridge, England. WCMC monitors all trade in wildlife (listed and not listed under CITES) and conducts surveys on species. According to McNeely *et al.* (1990) monitoring of the wildlife trade and its impact is essential for the effective operation of CITES. TRAFFIC (Trade Records Analysis of Flora and Fauna in Commerce) also collects information on the illegal trade, infractions and enforcement of CITES trade controls (OECD, 1997).

The Convention operates via a system of appendices. Once information on the status of the species is known, the species can be listed on one of three appendices. Different conditions and restrictions for trade apply depending on which appendix the species is listed under (Wijnstekers, 1995). Species can be uplisted and downlisted as more information on their status becomes available or from proposals submitted by CITES member countries.

A change in a listing however requires a 2/3-majority vote from member states at CoP's meetings. This method has been criticised, as it is difficult to get such a majority. This is especially true for proposals for the sustainable use and trade of some "charismatic" species. For instance, at the 1997 CoP's only the elephant proposal from Namibia, Botswana, and Zimbabwe gained the necessary majority (ACSUG, 1997). At the same meeting the South African proposal to trade in rhinoceros was turned down and proposals from Norway and Japan to transfer grey, minke, and Bryde whales from Appendix I to Appendix II were defeated (Pendry, 1997; ACSUG, 1997). It appears that politics rather than scientific merit determines whether some sustainable commercial trade proposals will be adopted (Steffen, 1992; ACSUG, 1997).

Appendix I includes species most threatened with extinction. Approximately 600 animals and 300 plants species are listed (OECD, 1997). Examples include primates, great whales, rhinoceroses, certain birds, sea turtles, certain orchids and cacti. Commercial trade is generally prohibited in these species and products (Wijnstekers, 1995). However, in exceptional circumstances parties to the Convention can apply for reservations allowing some limited form (such as scientific research) of trade in that species. Exemptions may also include commercially captive-bred animals and plants not wild specimens. In these cases the wild populations remain listed in Appendix I while the commercially bred populations are listed in Appendix II. Examples include Nile crocodiles *Crocodylus niloticus* that are ranched in Zimbabwe (Swanson, 1992b). For species exempt from a total trade ban and listed in Appendix I export permits from the country of origin and import permits from the importing country must be issued before any trade can take place. The overriding criteria for countries is to prove that this trade will not be detrimental to the survival of the species as international trade is thought to have a significant effect on the species survival.

Appendix II listed species are 'all species which although not necessarily now threatened with extinction may become so unless trade in specimens of such species is subject to strict regulation' (Klyza, 1994, 190). Further, Article IV of the Convention, states that exports of species should not effect the species role in its ecosystem (OECD, 1997). "Look-alike" species are also listed in this appendix (Wijnstekers, 1995). Appendix II includes about 4,000 animal and more than 25,000 plant species (mainly orchids) (OECD, 1997). Examples include bears, hippopotamuses, most parrots, lizards, giant clams, and carnivorous plants. Appendix II listed species are not yet threatened with extinction so can be internationally traded. Consequently this should increases the economic use value of the species. Therefore creating incentives for investment in conservation.

To monitor and control trade in Appendix II listed species, export permits or re-export permits are required from the country of source (Swanson, 1992b). Export permits should only be issued when such trade will not harm the survival of the species in the wild. CITES does not require import permits for Appendix II species (Swanson, 1992b). Many party states have however taken the added precaution and do require import permits. Appendix II rather than Appendix I is probably the most important piece of CITES legislation.

> *The inclusion of a species in Appendix I should be seen as tacit acknowledgment those conservation efforts for that species have failed.... Transfer from Appendix I to II, rather than being regarded as bad for species conservation, may represent a success story* (Pendry, 1997, 226).

In reality the history of CITES has witnessed one species after another progress from Appendix II to Appendix I (Swanson, 1992b). For example, the African Elephant was uplisted to Appendix I in 1989 after a 12 year listing on Appendix II (Swanson, 1992b). This ban has not stopped the trade in elephant ivory (Dublin *et al.*, 1995).

Appendix III allows individual signatory parties to list species they consider nationally important and in need of regulation and control. Some 200 animals and six plants are currently covered with an Appendix III listing (OECD, 1997). Examples include certain gazelles from Tunisia and American mahogany *Swietenia macrophylla* from Costa Rica. Export permits are required from the country that lists the species, while other countries that have stocks of the listed species require a certificate of origin before they can legally export the species.

To combat the illegal wildlife trade CITES and party countries focus on reducing demand for listed species and products (through campaigns in consumer states and legislation) and the criminalisation of the supply (through fines and trade sanctions) (Swanson, 1994a; OECD, 1997). This may be accomplished through exporting countries banning all exports of the species, while importing countries ban all imports of the species. In theory, if quantity demanded decreases this should lower the market price. At the same time criminalising the harvesting process increases supply costs narrowing the price-cost ratio. In consequence, the decreased demand and increased supply costs should in theory reduce the harvesting of the species. Further, increased enforcement, monitoring, and penalties should also deter illegal activities. In many instances this has however not happened.

Wildlife exploitation levels in many cases depend on foreign markets and the price paid by consumers for these products (Wijnstekers, 1995). If demand destruction is ineffectual and some consumer demand remains, in response prices may increase. This demand creates a monetary incentive for individuals to illegally harvest and supply the market. Hence, individuals will continue to harvest and trade in endangered species so long as they perceive that the relative returns to this activity remain high (Burgess, 1994). In addition if monitoring, enforcement, and fines are weak or poor, poaching and smuggling will continue. In short, a black-market may be nurtured by the higher prices generated by the ban.

4.3.4 Why CITES fails

CITES is a non-self-regulating convention (OECD, 1997). That is, it is the responsibility of individual countries once they have ratified the convention to pass domestic laws as required under the treaty. This entails setting up domestic agencies and authorities to issue permits, monitor, enforce regulations, penalise offenders, gather data on species and report back to the CITES Secretariat. Articles VIII and IX of the

Convention state the minimum condition's countries have to implement (OECD, 1997). However, the level of compliance, reporting, and enforcement varies from weak to strong from country to country.

According to a 1993 report by the IUCN around 85 percent of CITES parties have incomplete or otherwise inadequate legislation for implementing the Convention (OECD, 1997). Nevertheless, Article XIV states that countries can adopt stricter domestic measures. Several countries have passed stricter wildlife conservation and trade legislation laws. For instance, many South American countries have a total ban on wildlife trade (Hemley, 1995) while Australia has banned the export of its native birds since 1960 (Williams, 1997). However, these actions have not stopped the illegal trade and may even promote this trade. For example, in Australia the illegal harvesting and smuggling of parrots still continues (Moyle, 1998c). Likewise, parrot smuggling continues from South America (Simmons and Kreuter, 1990).

Although CITES has helped restrict the trade in some endangered species it has not altogether stopped the illegal trade in many species. Loopholes will and have emerged creating opportunities and incentives for illegal trade to prosper, reducing the global effectiveness of CITES (Burgess, 1994; Hemley, 1995). A report on the effectiveness of CITES based on a selected sample of twelve endangered listed species was carried out by Environmental Resources Management (ERM).[7] They concluded that, 'the CITES trade controls have been variable: positive in some cases, indifferent or less effective in others' (cited in OECD, 1997, 59). In fact, the status of only two of the twelve sampled species may have improved as a result of CITES (SACWM, 1996; Martin, 1997b). The Dublin *et al.* (1995) report arrived at a similar conclusion regarding the ivory trade ban.

Further, to scientifically prove that CITES has improved the status of any listed species would be very difficult (Martin, 1997b). Moreover, poaching and smuggling operations are often well organised and fluid to evade the trade bans. Corruption and bribery of officials, permit forgery, and lack of political will all lessen the effectiveness of any bans, creating illegal trade routes (Swanson, 1994a; Dublin *et al.*, 1995; Simon, 1995). For example, the standard bribe for smuggling wildlife out of the Pantanal region in cental South America is reported to be US$15,000 per vehicle (Simon, 1995). In short, trade bans have not been as successful as first purported and there is no definite

[7] For a critique of this report see SACWM, 1996.

proof that trade bans have benefited any species or achieved their goals (SACWM, 1996). Indeed, there is mounting evidence that trade bans are detrimental to many species.

4.3.5 Limitations of CITES[8]

CITES covers only those species internationally traded and threatened. In many instances however, data on listed species is lacking or incomplete. Further, it does not list all known endangered species. A survey has been carried out that compared CITES Appendix I and II listed species with the IUCN Red Data Book (SACWM, 1996). The survey found that the CITES appendices were oversubscribed with charismatic large mammals and "pretty" species of birds and undersubscribed with genuinely endangered species (SACWM, 1996).

What is of major concern is the literally millions (estimated) of unknown species, mainly unrecorded insects and plants that are threatened and endangered. This is commonly referred to as the biodiversity problem (Swanson, 1994a). International trade is not directly the cause for their decline so CITES does not cover them. However, the lost of many of these species could have a great impact of ecological systems and functions. This may affect human well-being more than the loss of the higher order charismatic species covered by CITES.

Further, the quantities traded of listed species are not that high in volume (Burgess, 1994). According to Martin (1997a, 4) 'international trade in wildlife products is not a significant cause in the loss of biodiversity.' Quantities traded are usually smaller than that for domestic use. In fact, domestic use of some species is now of greater concern. For instance, commercial hunting especially of primates for bushmeat in tropical countries has help push those species to the brink of extinction (Fa *et al.*, 1995; Spinney, 1998). In the Serengeti in Tanzania, poachers supplying local meat markets have reduced buffalo numbers (Milner-Gulland and Leader-Williams, 1992b).

Nevertheless, for some charismatic species such as rhinoceroses and tigers over-harvesting rates have increased. International trade is rapidly threatening population

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[8] See Appendix 2 for a summary of Martin (1997b).

levels endangering the survival of that species with imminent extinction possible. This has occurred despite CITES protection.

> *In the case of tigers and rhinos, the result has been disastrous. For instance, the world's rhino populations... have plummeted to a fraction of their numbers 2 decades ago, down to <11,000 animals. In just the last 5 or 6 years, as much as 15-20% of the world's tiger population in Asia and the Russian Far East may have been lost to poaching for the traditional oriental medicinal trade, with <6,000 tigers estimated to remain in the wild* (Hemley, 1995, 637).

In Africa, black rhinos *Diceros bicornis* have been subject to sustained illegal harvesting. For example, in the Luangwa Valley in Zambia, black rhino numbered between 4,000 and 12,000 in the early 1970's but by the mid–1980's were reduced to a few hundred (Milner-Gulland and Leader-Williams, 1992b). Martin, (1997b, 9) lists several reasons why the black rhino trade ban has failed. These include:

- the magnitude and nature of the demand was underestimated;
- the costs of the protection system pursued by CITES were too high;
- the illegal trade could not be controlled through the limited mechanisms available;
- the market was captured entirely by the illegal traders and;
- there were no incentives, such as existed for the Nile crocodile, to conserve the species.

Compounding these factors is the large areas rhino's inhabit, underfunding of anti-poaching patrols to deter poachers and an increase in the price of rhino horn (Leader-Williams and Albon, 1988; Leader-Williams *et al.,* 1990; Milner-Gulland and Leader-Williams, 1992).

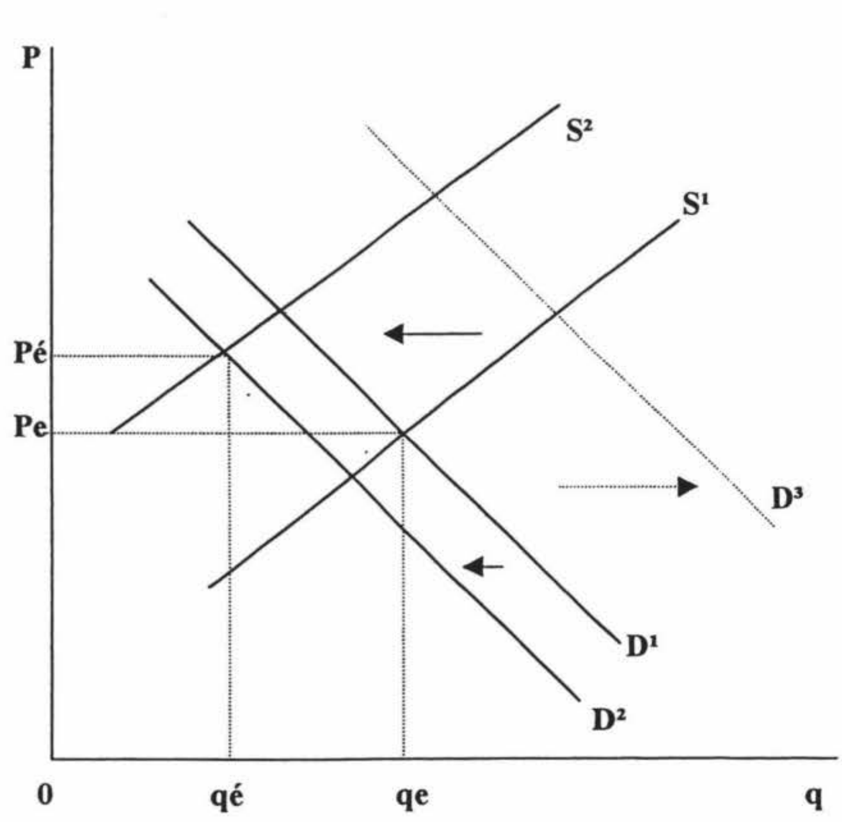
To be effective CITES requires the international co-operation of all parties in implementing measures to control the trade in endangered species. This is one of the major weaknesses of CITES. It is estimated that the enforcement of CITES is only 60-65 percent effective world-wide (Klyza, 1994). This is especially true in poor developing countries that also have the highest concentration of biodiversity and endangered species.

> *Many of these countries have simply lacked the expertise, interest, or political will to address specific wildlife trade control needs and to make wildlife trade enough of a priority that it can compete on the national agenda with other development and economic expansion needs* (Hemley, 1995, 637).

Swanson (1992a) also points out that there is very little cooperation between party states. Many range states act independently with respect to conservation, monitoring, and enforcement (Swanson, 1994a; Hemley, 1995). In addition, there is little cooperation between domestic law enforcement agencies (Halsted, 1991). In short, weak and poor enforcement makes poaching and smuggling of wildlife across international borders difficult to detect and difficult to prevent (Wijnstekers, 1995). The low probability of detection for a poacher or smuggler acts as an incentive for them to continue or to increase the illegal harvesting of wildlife. For example, the Wegner conspiracy operated for 8 years smuggling Australia parrots to the USA (Moyle, 1998c).

Barbier *et al.* (1990) illustrates how a trade ban is ineffective under weak enforcement and how illegal trade will increase. That is, banning legal trade while there is still an illegal component will expand the illegal component and may open up new markets. Fig 4.1 illustrates such a scenario.

Fig 4.1: Effects of an ineffectual trade ban.



1. Initially the species can be legally traded, quantity demanded (D^1) and quantity supplied (S^1) are at equilibrium at qe and price Pe.
2. Assuming a trade ban is introduced. This effectively stops all legal trade. However, the ban can not wholly eliminate the illegal supply and demand. Hence, the supply curve shifts to S^2. Although the ban should have destroyed all demand, some consumer demand remains. Consequently, the demand curve shifts inwards to only D^2. The new equilibrium is now qé and Pé, a lower quantity and higher price relative to the pre-ban equilibrium. This is the illegal component of the trade. Although the illegal quantity is less than the pre-ban quantity (qé < qe) this quantity does not include the high mortality rate estimated to occur with smuggled live species. Therefore it may be concluded that quantities taken from the wild would be much higher than qé and even qe.
3. The higher price also acts as an incentive for new poachers to enter the illegal market. Especially if enforcement is poor, penalties are weak, or corruption of officials is easy. In response, the original poachers may decide to harvest the remaining population as quickly as possible, particularly as an open access regime operates. In consequence the species is rapidly over-harvested.
4. Finally, should new markets emerge, demand may increase for example to (D^3) and this will exacerbate the situation.

In short, unless demand can be completely destroyed, opportunities exist to continue an illegal trade.

Increasing law enforcement may not however be the most appropriate strategy. Moyle (1997) has constructed a bioeconomic model of illegal wildlife harvesting demonstrating this point. That is, increasing law employment may signal imminent scarcity to consumers, who may respond by increasing their price offer to the smugglers for the wild species (Moyle, 1997). In consequence, a speculative bubble may emerge pushing up the price (Moyle, 1997). This increases the profit-maximising opportunities for poachers and smugglers, who may harvest more species. Moreover, increasing law enforcement will not halt the main cause of species extinction, that is the conversion of habitat into agricultural use for example (Moyle, 1997) Rather, creating legal markets and incorporating economic incentives may be a more effective conservation strategy (Moyle, 1997).

The justification for listing a species on a CITES appendices is that international trade is, or could be, a threat to that species. Over-exploitation while one of the causes of species loss is not the fundamental cause of the problem. Further, the trade in wildlife is complex and CITES has not prevented the illegal trade in endangered species. Moreover, although CITES and the related domestic wildlife laws can punish poachers and in turn deter these illegal activities, CITES does not offer incentives to conserve endangered species.

Simply, CITES can create conditions for an illegal trade to prosper. That is, the economic returns to individuals from over-harvesting and poaching are greater than the aggregate economic benefits of conserving the species. Exploiters (poachers) do not pay the full costs while those who pay the most gain the least benefit. McNeely (1990) suggests that ultimately the solution is to redress this imbalance through ensuring that exploiters pay the full costs of their exploitation, and that conservers earn more of the benefits from their action. Many believe that easing such disparities would do more than anything else to protect the world's endangered species (Pearce, 1997). Hence, the importance of property rights. As a conservation treaty and as an effective strategy to conserve biodiversity CITES has its limitations.

4.3.6 Criticisms of CITES

There are other paths to species extinction. McNeely (1990); Brown, *et al.* (1993); Burgess (1994); and Swanson (1994a) discuss these other possible reasons for species extinction. These include loss of habitat, introduced species, and pollution. CITES does not address these causes. CITES is only a solution for over-exploitation due to international demand. However, few if any species extinction have been the result of international trade (SACWM, 1996).

There is criticism of the decision-making process. For instance, at CITES CoP's in the large, diplomats rather than scientists attend (Martin, 1997b). Consequently, decisions and positions on species are more than often politically motatived (SACWM, 1996).

> *The positions taken by many nations are now primarily political and they are decided in advance of the meeting so that debate is futile. Many of the delegates are junior bureaucrats.....Essential background documents, such as the reviews from IUCN/TRAFFIC on listing proposals, are not read or ignored. Few Parties take any interest in proposals involving species which occur outside their boundaries* (Martin, 1997b, 8).

Further, voting to ban trade in a species usually incurs no cost for many of the parties. Hence, parties may vote against sustainable trade proposals without considering the consequences for the range country. Furthermore, Martin (1997b) argues that CITES allows importing countries rather than exporting countries to dictate the terms of trade for permitted species. As previously stated, countries may impose much stricter laws with respect to the exporting and importing of wildlife and countries can take this too far (Box 4.2).

Box 4.2: Importing countries dictating the terms of trade.

In 1992 Zimbabwe received a notification from the European Union to the effect that *"in the interest of conservation of species in your country, the following imports of live animals will not be permitted –"*

- one primate which did not occur in Zimbabwe;
- *Papio ursinus* – the common Chacma baboon of which thousands are killed as pests every year.
- *Cercopithecus pygerythrus* – the common vervet monkey, in the same category.
- *Agapornis nigrigenis* – the black-cheeked lovebird which disappeared from the wild in Zimbabwe sometime in the 1970's when elephants destroyed their habitat, but which are common in other countries in the sub-region. Zimbabwe managed to breed large numbers of the species in captivity and it was these which were now being prohibited for import.

Source: SACWM, 1996, 8.

Swanson, (1994a) critiques the trade ban approach. Of concern is that CITES trade bans effectively reduce the consumptive use value of a species. According to Swanson, a trade ban may protect a species in the short-term from over-exploitation but conversion of habitat in the medium-term will led to its extinction. Swanson (1994a, 7) calls this the "global conversion process", defined as the 'replacement of the diverse with the specialised.' Swanson argues that existing policies of demand destruction and supply criminalisation are misconstructed and misguided. Species need investment in land, protection, and management to exist. Current trade bans create conditions for underinvestment and a disincentive for future investment in the conservation of that species. That is, extinction is a result in the mismanagement of the biological resource and misappropriation of the base resources (natural habitat) required for a species survival.

In addition, the IUCN have argued against a blanket trade ban as an effective conservation policy believing that it is unworkable in the long-run (Pearce, 1997). 'Such violent swings in policy from overexploitation to trade bans (are) generally counterproductive to the development of more rational conservation programmes' (Pearce, 1997, 15). Brown *et al.* (1993) also argues against the case of trade bans as a solution to halt species extinction.

> *A system of bans, and the resulting reductions in the profitability of all diverse resources, is the antithesis of a constructive approach to the fundamental problem of extinction* (Brown *et al.*, 1993, 77).

Martin argues against the inflexible approach to conservation taken by CITES:

> *CITES operates in a vacuum, taking no account of human or economic considerations. This is very much out of step with the growing global realisation that conservation and sustainable use in the developing countries can only be achieved with participation of local peoples* (Martin, 1997a, 7).

Martin (1997b) furthers criticises the appendices for their inflexibility. For instance, trade may be beneficial for some Appendix I species but trade is ruled out. SACWM (1996) likewise criticises the effects of an Appendix I listing. They argue that once a species is listed this restricts the use of the habitat in the range countries. Owners of land lose a potentially valuable source of income from the consumptive use of that species, in turn this may reduce the value of the land.

Brown *et al.* (1993) propose that conservation policies should focus on inducing investment by owners and states into these resources. Barnes (1996) also agrees that demand suppression does not address the issue of conservation. Rather, the conservation solution is one of controlling access through appropriate property rights. Well-defined property rights protect owners' rights and decrease the risk to investments. Moreover, if owners are allowed to trade in wild species they should be able to receive an economic return from their investments in conservation. In short, the perceived investment worth of biological resources must be altered and increased, commercial trade and property rights help achieve this.

4.4 SUMMARY: TRADITIONAL CONSERVATION STRATEGIES

Natural areas have been managed for thousands of years and wildlife traded for a similar age. Nevertheless, it is only in the last couple of hundred years that real concern has been raised regarding the quantity and quality of natural areas. Protected areas and trade measures are traditional responses and they do have a crucial role in preserving biological diversity. Both however have limitations.

Many countries have large areas of natural habitat officially protected. These protected areas are however often poorly managed with inadequately enforced laws. Increasingly more demands for development and intensive use are being placed on protected areas and surrounding habitats. Conversion is a distinct possibility. In addition, much biodiversity is located outside protected areas on private or communal lands. These areas may be occupied and used by humans.

Contemporary orthodox management solutions while necessary will only partially protect biodiversity. Nevertheless, preservation of natural habitat is still the most efficient mechanism for the conservation of biodiversity (McNeely *et al.*, 1990; Brandon and Wells, 1992). The preservation of land in its natural form remains important for the continuation and functioning of the ecological services essential for human well-being. Equally important it allows for the continuation of species speciation. Therefore, effective conservation must involve all three dimension of biodiversity. Moreover, not maintaining the species that compromise the landscapes will result in degradation of the integrity of landscapes. The questions are, can current strategies be improved? Can current resources (financial and natural) be utilised more effectively and efficiently?

Additional and innovative approaches to conservation are required. Strategies that incorporate biodiversity into economic planning and management need to be developed and implemented if biodiversity is to remain healthy. Conservation must be viewed as more than merely biological it must also include social and economic factors. There is a growing acknowledgment of the importance of involving local communities in conservation management and the benefits of the commercialisation of wildlife such as sustainable trade.

5

Conservation Strategies: Partnerships & Wildlife Utilisation.

Commercial trade may be beneficial to the conservation of species and ecosystems and/or to the development of local people when carried out at levels that are not detrimental to the survival of the species in question. **1992 Conference of the Parties to CITES, Resolution Conf. 8.3.**

Wildlife utilisation affords the possibility of making development consistent with diversity. At the heart of a new conservation policy is enabling local communities to make use of the diversity that exists all around them **(Swanson and Barbier, 1992, 215).**

5.1 INTRODUCTION

Chapter 4 reviewed traditional conservation strategies protected areas, trade bans, and their limitations. These strategies have not stemmed the rate of species extinction. In fact, more species than ever are at serious risk of extinction (McNeely *et al.*, 1990). According to the WCED (1987) measures need to be implemented to halt this loss and then economic strategies developed to sustainably use these resources. Further, biodiversity needs protection not against people, but against inappropriate uses and overexploitation (McNeely *et al.*, 1990). Hence, to be effective any conservation strategy must include economic, social, and cultural dimensions. Without a consideration of such issues, habitat preservation is not possible.

> *Regardless of the laws or strengths of the national or international authorities, a conservation area will not be successfully protected without the cooperation of the local people, or at least without provision for the needs of the local people* (Jordan, 1995, 244).

5.1.1 Chapter outline

This chapter will explore recent innovative conservation proposals. They are partnerships in conservation involving local level participation and the commercialisation and sustainable use of wildlife. Section 2 examines conservation partnerships. In particular the importance of actively involving local communities in the management of these areas is discussed. Community-based conservation initiatives and their limitations are reviewed. Wildlife utilisation and commercialisation is considered in Section 3. First, ways to improve CITES are explored. Second, the sustainable use and trade argument is examined as an alternative. Comments on its problems and limitations are given. Following this is a case study on the African elephant. This case study illustrates a successful conservation strategy involving local communities. Evidence is presented supporting the advantages of a sustainable utilisation strategy and highlighting gains from trade. While these new conservation strategies have certain advantages and benefits over traditional methods, success will depend upon appropriate property rights regimes.

5.2 INNOVATIVE IDEAS IN PROTECTED AREA MANAGEMENT

5.2.1 Partnerships in conservation

Protected areas do not adequately represent or protect all biodiversity. More areas are needed, especially in light of the forecasted extinction rates. Further, to be effective conservation must incorporate human needs and wants. Hence, to achieve conservation of biodiversity as well as economic development new strategies need to be adopted. This will require the blending of conservation inside protected areas with economic development outside protected areas. Importantly the conservation and economic development objectives of these new strategies must be compatible (WCED, 1987). Otherwise the same problems identified with traditional strategies will not be resolved.

Therefore the challenge facing conservationists today is the development of innovative strategies for maintaining habitats outside protected areas (Webb, 1991; Luisgi, 1995). Conservation now requires a perspective that stretches beyond park boundaries and involves programs affecting rural communities (Wells *et al.*, 1992). These strategies will only succeed if communities are actively involved in decision-making and management, where the partnership is fair and legitimate, and when all stakeholders are united in a common purpose. Evidence suggests that incorporating local communities will help to increase the effectiveness of the strategy and can contribute to local development (Furze *et al.*, 1996). The rationale is simple:

> *Local people are the most familiar with the area and the wildlife within it; the failure to ensure their co-operation will make them indifferent and perhaps hostile to conservation efforts, which they will see as being "imposed from outside"* (Barbier, 1992b, 131).

The key to such strategies is the distribution of the benefits from conservation. If local communities do not secure a fair share, the strategy may fail. Underlying this issue are property rights.

5.2.2 Local communities/Indigenous Peoples

Within and surrounding protected areas live indigenous peoples and local communities[1] (Furze *et al.*, 1996). Further, these people are dependent upon and share local ecosystems and biological resources with wildlife. However, this does not mean that they over-exploit these resources.[2] To some extent many communities practise sustainable resource management (McNeely, 1988).

In addition, local people have accumulated a vast amount of legitimate knowledge and information on the uses and functions of plants, animals and landscapes (Swanson,

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[1] Note that there are many problems at the local level, with community and bottom-up development. See Furze *et al.*, 1996, 1-15.

[2] Note that many indigenous peoples are responsible for major extinctions of some species. For example, the spread of Polynesians across the islands of the Pacific 8,000 years ago led to the extinction of at least half of the endemic species they encountered (Freese, 1998).

1994a; Raven, 1995; UNEP, 1997). This knowledge is of value for conservation strategies (McNeely *et al.*, 1990; Meffe and Carroll, 1994a; Furze *et al.*, 1996; Moyle, 1998a).

> *[T]hey possess, in their ecological knowledge, an asset of incalculable value: a map to the biodiversity of the earth on which all life depends....Biological diversity – is inextricably linked to cultural diversity* (Furze et al., 1996, 133).

This information also has high economic value. For example:

> *traditional shifting cultivation is a system that is well adapted to the tropical forest environment, helps maintain the biological diversity of the forest, and often provides significant benefits to wildlife populations. The maintenance of such systems is of considerable importance to modern forms of development. The wild relatives of a variety of important crop plants occur in the forest, and these and the primitive cultivars grown by swidden cultivators are valuable sources of genetic material for modern plant breeders* (McNeely et al., 1990, 50).

Furthermore, hunting communities have acquired valuable knowledge on species (Moyle, *in press*). Hence, it is now recognised that the involvement and cooperation of such groups of people render an important capacity for effective conservation.

One of the failures of traditionally managed protected areas was the way local communities were historically treated. Conservation programs often treated local people as opponents rather than partners (McNeely *et al.*, 1990). Centrally controlled state management (top-down) replaced local level communal ownership. The result was a breakdown in traditional resource management practices. Further, conservation planners rarely used local or indigenous knowledge. Consequently, valuable knowledge on species was lost due to the replacement of traditional management and the disruptions from the introduction of foreign institutions. These effects often had negative ecological consequences for wildlife, with little or no control over the use of wildlife open-access results, in turn species are over-exploited. Finally, the new park authorities are often incapable of preventing illegal harvesting.

In addition, the limited size of protected areas makes it difficult to contain many species. Protected areas although having an official boundary and possibly fenced do not stop wildlife from leaving the protected area. Marauding wildlife can destroy, kill and eat livestock and crops grown by local communities on land surrounding protected areas (Barbier, 1992b). Wildlife rather than being treated as a valuable resource is now looked upon as a nuisance (Skonhoft, 1998). In short, there is little incentive for local communities to conserve wildlife and a great deal of resentment from local communities towards conservation, wildlife, and protected area management.

What is now becoming clear is to ensure local community participation in innovative conservation initiatives (UNEP, 1997). Effective protection of biodiversity, requires conservation strategies to be based on the integration of protected areas with the surrounding landscapes and people (McNeely, 1988; McNeely *et al.*, 1990; Jordan, 1995; Furze *et al.*, 1996). This implies conservation will be generated as a bottom-up approach. Moreover, it is important to approach biodiversity conservation through a process that also improves the economic and social well-being of local people (Dickinson III, 1995). That is, the strategy, policy or project must not be orientated solely towards the preservation of wildlife, but that wildlife conservation is a source of income for local people. For instance, research has shown that wildlife management programs receive greater support from local communities when there are tangible incentives for participation (Burgess, 1994; Jordan, 1995; Tisdell, 1996). For example, when Latin American parrot trappers were paid a regular salary to become parrot protectors, illegal levels of harvesting declined (Moyle, 1997). In essence, local trappers' income increased from protecting the parrots rather than poaching them and this generated conservation benefits.

McNeely (cited in Furze *et al.*, 1996, 138), lists 10 principles to achieve linkages between local communities and protected area management. These are as follows:

1. Build on the foundations of the local culture.
2. Give responsibility to local people.
3. Consider returning ownership of at least some protected areas to indigenous people.
4. Hire local people.
5. Link government development programs with protected areas.
6. Give priority to small-scale local development.
7. Involve local people in the preparing of management plans.
8. Have the courage to enforce restrictions.
9. Build conservation into the evolving new national cultures.
10. Support diversity as a value.

In sum, there are positive motives for including local communities in conservation partnerships. First, many governments lack the financial and other resources to exclude and remove local communities from using parks, hence by working with local communities these resources could be allocated to other activities. Second, the successful integration of local people into decision-making, management, and development lessens the conflicts that had previously surfaced between local communities and protected areas, again saving resources that were devoted to law enforcement. Third, by including local communities they become guardians for local biodiversity and behave in a way that conserves those resources. Finally, local communities have knowledge to draw upon to generate conservation and development.

5.2.3 Applied approaches linking protected areas and local people

Integrated Conservation and Development Projects[3] (ICDP's) have been promoted as initiatives attempting to link conservation and economic development (Wells *et al.*, 1992). The core objective is protected area conservation (Brandon and Wells, 1992). ICDP's also have several other goals. They are to stabilise land use outside and inside protected areas, and to increase local incomes in order to reduce the pressure for further exploitation of protected areas (Wells *et al.*, 1992). By promoting socioeconomic development inside and outside protected areas compatible with conservation the threat to biodiversity may be lessened.

Moreover, the projects attempt to change local communities' attitude and behaviour towards wildlife. To accomplish this ICDP's should include local people in the planning and implementation process and ensure that they receive a share of the benefits generated. The aim is to reconcile the management of protected areas with the social and economic needs of local people (Wells *et al.*, 1992). ICDP's are largely co-management projects between the government, the conservation authority, and local communities. Generally local communities have ownership of the land while the state manages the wildlife. Park protection is coupled with the promotion of environmentally benign development activities carried out by local communities surrounding protected areas (Brandon and Wells, 1992; Southgate, 1995).

The development component separates ICDP's from other conservation projects (Wells *et al.*, 1992). A key assumption of ICDP's is that conservation strategies will only work if local communities receive sufficient benefits that change their behaviour from exploiting protected areas to protecting them (Gibson and Marks, 1995). ICDP's seem to work best at sites where wildlife attracts large numbers of tourists (Lewis and Alpert, 1997). Overall, these projects endeavour to minimise the impact of local

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[3] **Also similar to community-based natural resource management, CBNRM (Wainwright and Wehrmeyer, 1998) and primary environmental care (PEC) (Pye-Smith *et al.*, 1994).**

development on wildlife and areas of natural habitat. In addition, the UNESCO is implementing Biosphere reserves around the world (Box 4.1).

Box 5.1: ADMADE in Zambia.

Administrative Design for the Management (ADMADE) of Game Management Areas operates in areas surrounding national parks in Zambia, and specifically the Luangwa Valley. Wildlife is protected inside national parks but is utilised on surrounding areas. The surrounding land is communally owned and classified as GMA's but the wildlife is owned by the state. Hence, ADMADE is a comanagement approach to conservation involving a partnership between the state, the wildlife authority, and local communities. It is an initiative for conserving wildlife based on the sustainable offtake of wild animals, with the proceeds of the harvest shared between the stakeholders. The ADMADE program divides the responsibilities and benefits from the use of wildlife more equitably between the state, the private sector, and local communities than previously.

Revenue is generated from safari concessions, fees, hunting licenses (trophy and residential), donor contributions and own profits from culling. Trophy hunting is the big revenue earner. For example, in the Luangwa Valley wildlife viewing tourist paid about $US140 per day whereas trophy hunters paid between US$800-1000 per day (mid 1990's). In general under ADMADE, 35 percent of revenues are distributed to community projects within the GMA; 40 percent to wildlife management and enforcement programs within the GMA, mainly the village scout program; 15 percent to national park management; and 10 percent to the Zambia National Tourist Board.

ADMADE has provided jobs to over 500 local residents as scouts. This has greatly expanded the capacity of the National Parks and Wildlife Service (NPWS) resulting in increased enforcement and arrests. In addition, the revenue from wildlife use has been used to fund schools, health clinics, provide capital for local enterprise, and to provide basic needs such as food when there is a shortage.

Importantly, local people's attitude towards wildlife has improved. For example, in 1994 several communities shifted families and gardens to reduce village encroachment on wildlife habitat. More communities are asking to have their land classified as GMA's or extended to earn revenue from wildlife through ADMADE. As local people how have an economic interest in preserving wildlife in their area, poaching has been greatly reduced. In fact, some wildlife populations may exceed those inside national parks. In particular, elephant poaching has been reduced and elephant populations are recovering.

Note that in the same area's the Luangwa Integrated Resource Development Project (LIRDP) also operates. The LIRDP is however, a multi-sectoral program for economic development. In addition to wildlife management LIRDP includes programs for agricultural, forestry, fisheries, water resources and infrastructure. See Barbier, 1992b and Wells *et al.*, (1992) for detailed descriptions of this program.

Sources: Barbier, 1992b; Wells *et al.*, 1992; Gibson and Marks, 1995; Lewis and Alpert, 1997.

ICDP's have however been criticised. In a review of ICDP's only 5 out of 36 projects were found to positively contribute to the conservation of wildlife (Kremen *et al.*, 1994). Brandon and Wells (1992); Wells *et al.* (1992) found difficulties and flaws in the design of, and achievements of ICDP's goals. These include *inter alia* conflicts between conservation and development objectives[4], a lack of understanding of the interaction between the local communities and the environment, the scale of the projects, and limited financial resources (Wells *et al.*, 1992). Importantly, the ICDP's studied appeared unlikely to generate enough economic or financial benefits to become self-sufficient, hence affecting local communities incentives to participate in these projects (Wells *et al.*, 1992).

Barrett and Arcese (1995) summarise many of the problems of ICDP's. Their main concern is over the biological and socioeconomic sustainability of the projects. They argue that there is much uncertainty concerning the sustainability of commercial harvesting of wild species and the conditions necessary to generate a profit. Further, the

[4] Linking conservation and development objectives is in fact extremely difficult (Brandon and Wells, 1992).

social and economics effects from commercial harvesting of wild species are largely unknown. Moreover, some developing countries governments are hesitant to give such power, control and organisation strength back to local communities (Thomas-Slayter, 1994).

Gibson and Marks (1995) identified some faults while studying community-based wildlife programs in Zambia. Gibson and Marks (1995) criticise the distribution of the benefits of ADMADE; and make note of the fact that very little revenue from wildlife utilisation is directly returned to rural residents. For example, very few projects are started and are able to be sustained without outside donor contribution. This raises doubts about the long-term sustainability of these projects. Further, projects are often clustered around the chief's residence (Gibson and Marks, 1995). Culled meat more than often ends up in urban markets and local residents have to resort to hunting (often illegal) for sustenance (Gibson and Marks, 1995). Finally, the projects were successful in that they increased protection for large mammals by changing local hunters' behaviour. However, the projects failed to account for the social significance of hunting and the role of the hunter in the community (Gibson and Marks, 1995). In consequence, these hunters switched to hunting other smaller species.

In a recent publication on community-based natural resource management also in Zambia, Wainwright and Wehrmeyer (1998) were critical of the success of these initiatives. They concluded that the LIRDP 'has generally failed to achieve its conservation and development objectives and that the program had achieved few community benefits' (Wainwright and Wehrmeyer, 1998, 933). Although there are some problems, ICDP's are relatively new so there will be a learning process that should discover solutions to these problems. In short, Wells *et al.* (1992) remark that there are very few other alternatives attempting to link conservation and economic development.

For ICDP's to work will require strong and effective partnerships in conservation. To be effective ICDP's have to be site-specific (Brandon and Wells, 1992). Design and implementation will depend on local biological and ecological characteristics as well as social, economic, and cultural characteristics. It is important that local communities derive economic benefits from local biological resources. Local communities have to have incentives to devote habitat for the conservation of biodiversity. That is, they will conserve wildlife if they can derive a positive economic return from local species. A recent proposal is the commercial utilisation of wildlife.

5.3 SUSTAINABLE USE AND TRADE IN WILDLIFE

5.3.1 Ways to improve the effectiveness of CITES

CITES is an important wildlife conservation treaty. It has brought protection (in the short-term) to many animals and plants that had previously been excessively traded (Hemley, 1995). CITES has identified wild fauna and flora species that are endangered (Wijnstekers, 1995). It has raised the awareness and support for conservation of endangered species, which is assumed to have reduced extinction risks for many species. Importantly, CITES has the mechanisms that allows the sustainable regulated trade of wild fauna and flora. However, to be an effective *long-term* conservation treaty CITES will have to evolve and adopt new methods towards the goal of wildlife conservation. This will entail increased funding and greater commitment to monitoring and enforcement for instance.

A lack of finance is a major impediment to the implementation of the Convention's regulations (Hemley, 1995). As stated previously, CITES requires countries to finance the cost of setting up the domestic authorities and infrastructure required under the legislation. This is a major obstacle for developing nations. There is also a great variation in the commitment given to enforcement. In a study researching the effects of the CITES trade ban on African elephant (Dublin *et al.*, 1995) found a chronic and dramatic decline in law-enforcement operating budgets throughout the range states surveyed. Compounding this is the large number of endangered species located in poor tropical countries. Bilateral and multilateral funding such as the GEF may help here, as long as the funding is directed into enforcement and conservation.

International political pressure such as economic sanctions against non-complying countries and consumer states could improve the effectiveness and compliance of CITES. For instance, in 1994, the United States imposed trade sanctions against Taiwan for wildlife trade violations. In response, Taiwan introduced stronger wildlife legislation (Baum, 1994; Hemley, 1995). However, such actions may be in violation of many free trade agreements (Moyle, 1998e).

Split-listing of species is another measure to overcome illegal over-harvesting. This is especially relevant when a species can be commercially bred on ranches. Wild populations could still be listed in Appendix I (total trade ban) but commercially bred stocks could have an Appendix II listing (controlled trade). Owners of the commercially breed species use wild species to augment their breeding stock thus they have an incentive to conserve the wild stock. Further, split-listing should allow countries with large populations that are not threatened to carry out sustainable harvests while the threatened populations are still listed in Appendix I. There are however, constraints with split-listing. For instance, there needs to be ways to identify and differentiate between wild and captive stocks and to stop unsustainable and illegal stocks from entering the country, strict border controls are necessary.

In summary CITES can be improved by increasing budgets, improving the level of commitment and enforcement of the regulations, split-listing of species, and using trade sanctions. Evidence however suggests that as the current legislation is not working effectively then tougher measures may also fail. As stated, the illegal trade is well organised and very fluid, so it may be impossible to stop it altogether (Burgess, 1994; Wijnstekers, 1995). Martin (1997b) also suggests that the benefits CITES has generated could have been achieved through other means.

Importantly trade bans should not be treated in isolation. For instance, if illegally harvested quantities of a species have declined this does not mean that the trade ban has worked. It could mean that the smaller and isolated populations of the species are now harder to locate. This may raise the harvesting costs of poachers above the opportunity costs of alternative income and as a result harvesters shift out of poaching. Further, with the return of wildlife to local communities they now have an incentive to protect their resources, consequently enforcement is increased and poaching is deterred.

CITES has a top-down approach to species conservation, which conflicts with the trend of returning wildlife management back to local people. This may be problematic especially at CITES meetings if local people are not represented (Martin, 1997b). Therefore the improvements to CITES mentioned above have limitations. In contrast, sustainable use and legal trade have been proposed as conservation alternatives.

5.3.2 Commercial utilisation of wildlife

Wildlife utilisation is an innovative but controversial conservation proposal. There are a variety of management option, some consumptive and others non-consumptive. For example, trophy hunting is consumptive while wildlife viewing is non-consumptive.[5] The scale of use can range from large-scale international trade to supplying the domestic market or for small-scale subsistence use. Importantly, commercial use allows natural ecosystems to compete with alternative land uses on a profit basis. In many instances, this type of use is more profitable than conversion. For example, in a rainforest in Peru the returns from non-timber harvesting of fruits and latex, plus selective cutting of timber easily exceeded the returns from alternative options such as clear-cutting timber, plantation harvesting, and cattle ranching (Barbier, 1992a). This will be reinforced if local communities are involved and derive positive economic benefits.

> *the Southern African region is able to demonstrate real conservation successes in contrast to many other regions both in Africa and the rest of the world. These successes come about through enlightened decentralisation policies which confer rights and responsibilities to rural landholders and promote the economic value of wildlife* (SACWM, 1996, 18).

Importantly it offers the following advantages.

> *Sustainable use as a biological conservation strategy assumes that the value of wild resources will foster development that is compatible with conserving biodiversity. That is, if local people are allowed to value wild species for tourism, subsistence, trade and so on, development can occur without sacrificing as much diversity as would be lost if the potential value of the wild resource were not realised* (Furze *et al.*, 1996, 180).

Local communities can proposer in terms of economic development while a proportion of the profit from wildlife utilisation could be invested back into conservation programs.

A legal well-regulated wildlife utilisation system may be a more effective way to reduce poaching and maintain populations of wild species. A legal market may drive down the price thus crowding out the illegal market and discouraging poaching (Moyle, 1997). In a legal situation with well-defined property rights local people have an incentive to work with the enforcement agencies, such as informing on any illegal harvesting activities (Moyle, 1997) and carrying out their own monitoring and enforcement.

Swanson (1992b, 1994a) argues that wildlife utilisation is a good conservation tool. It makes it possible to develop natural areas while maintaining the biological diversity of habitats. That is, it generates revenues and incentives for conservation. Swanson (1994a) cautions however that revenue raised from sustainable utilisation may not be enough to guarantee conservation. Rather, it is the flow of *rents* from the

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[5] For a comprehensive survey of the role of wildlife tourism and protected areas see Barnes *et al.* (1992) and Ceballos-Lascurain (1996).

sustainable offtake that will create incentives for the maintenance and conservation of wildlife. To guarantee a secure flow of rents will require well-defined property rights.

A sustainable use strategy has to go hand-in-hand with a good management regime (Swanson, 1994a; Freese, 1998). The objective of a management regime is to maintain the wild stock and regulate trade when necessary. Management would also have to be flexible to (re)enforce controls or bans should problems become evident. This will require information on the biological and economic status of the species. Funding assistance could come from the major consuming countries, this being the cost they pay for sustainability (Swanson, 1994a). Debt-for-nature swaps are one example. However, this still is a top-down approach to conservation and there are difficulties and limitations with international funding mechanisms. For instance, contributions to international funds are largely voluntary, and many countries have been neglectful in making their contribution (Swanson, 1992b). As biodiversity has a global public good characteristic, the free-rider problem emerges and countries that do not contribute to the fund benefit.

Furthermore, a system to discriminate between which states (good) are investing in their wildlife resources and which states (bad) are not needs to be implemented (Swanson, 1994a). That is, to differentiate between the sustainable legal offtake (from the good state) and unsustainable illegal catches (from the bad state). However, it is difficult to perfectly monitor countries and they may hold back information. This raises the problem of asymmetric information, and the possibility of a principal/agent problem (see Moyle, 1998b). This may require an external authority to audit, monitor and make recommendations. The TATIS concept document proposes a solution to this problem (see section 5.4.3).

Swanson (1994a) discusses the necessary components of such an international regime that discriminates between sustainable and unsustainable producers. In short, the price the producer receive from legal and sustainable trade is the premium for investing in conservation while it is the penalty paid by the unsustainable producer who is banned from trading. Such a system may encourage other countries with poor enforcement and management of their wildlife to develop better systems. Regional trade agreements have the ability to overcome some of the above-mentioned problems (Box 5.2).

Box 5.2: Regional trade agreements

CITES is a large multilateral trade treaty. As previously stated many party members are not that interested in proposal's from other parties that do not affect them directly. In addition, environmental groups from their own country often influence delegates. However, each party has the right to vote on trade proposal at the CoP's meetings. For these reasons the voting structure of CITES could have an adverse effect on the species and their long-term survival. Regional trade agreements are one option to overcome this constraint.

Regional trade agreements are economic coalitions among countries within a geographic region. These agreements are usually characterised by liberalised internal trade and uniform restrictions on external trade, designed to promote regional economic integration and growth. Such groupings can also promote long-run development by enabling nations to block certain forms of trade with the more powerful developed nations. In short, these trade agreements offer the opportunity for countries to peruse regional conservation and economic development for their own interests.

For instance, the five range states of the polar bear (Norway, Russia, USA, Canada, and Denmark/Greenland) concluded an international treaty in 1973 between them on the conservation of the polar bear. This treaty allowed for the continued hunting (for customary and commercial use) in some countries and the ban from such activities in other range states. Freeman concludes that there have been multiple benefits associated with this regional treaty. 'The real success is that polar bear numbers have increased from an estimated minimum 5,000 in the 1970's to around 21,500 currently.'

The harvesting of swiftlets nests are a good example where a regional trade agreement rather than a multilateral treaty may be more appropriate. The main producing and consuming countries are in South East Asia. Demand for the birds' nests has increased with the rise in income in many of the countries in this region. This has raised concern that harvesting to meet this demand may be excessive and endangering the species. The population is however estimated at approximately 50 million individuals, CITES has set a criterion of 5000 individuals for a small population threatened with extinction. Nevertheless, in 1994 Italy (a non-producer or user state) using incorrect data, proposed that all species of swiftlets should be listed on Appendix II of CITES. This was not successful. Should the listing have been successful rather than helping in the conservation of the swiftlets it may have been detrimental and created major conservation problems.

In short, certain aspects of CITES, specifically the voting behaviour can create an unequal environment between parties. Whereas regional trade agreements are tailored to the local needs rather than being constrained by a distant global agreement. In addition, knowledge is available at the regional level. Hence, regional trading offers conservation and economic development benefits. Sources: Regional trade: Todaro, 1994; polar bears: Freeman, 1998 and swiftlets: Er *et al.* 1995.

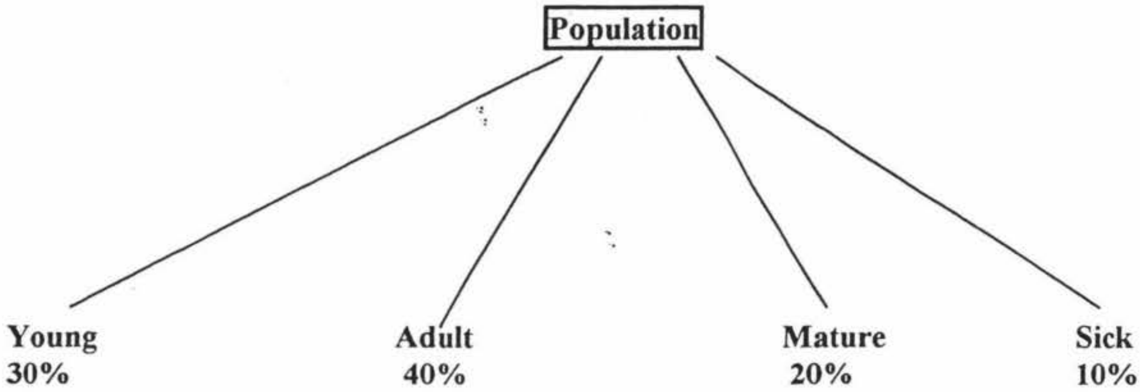
5.3.3 Comments on sustainability

The majority of the poorest people live in ecologically fragile areas, and are dependent on wild foods (McNeely *et al.*, 1990; Barbier, 1992a; Freese, 1998). Under conditions of extreme poverty a household's major concern is securing sufficient means for survival today (Streeten, 1985; Barbier, 1992a; Todaro, 1994). This could affect the sustainable level of harvests, especially if an open-access regime operates. Further, sustainability is compounded by a high degree of uncertainty over sustainable harvest levels and stochastic shocks such as drought, other natural disasters or natural die-offs. Nevertheless, property rights can generate incentives to conserve wild species when these situations arise, as the following hypothetical example illustrates.

Natural die-offs although they can be very severe do not affect the population proportionately. The following example illustrates first, the effect of a stochastic shock on a population and changes in the growth rate. Second, these rates are then compared with a financial rate of return. The main point is that species have comparable or better rates of return.

Assume that the population prior to a shock may be distributed as follows:

Fig 5.1: Hypothetical population distribution of a wild species.



If each adult has stochastic rate of reproduction of 0.25, the intrinsic rate of reproduction for the population as a whole is 10 percent. Resource owners have the choice of harvesting the resource and investing the returns into alternative assets or holding onto the resource to reap future income. In the above situation *E(10%) = r* for the owner to have an incentive to maintain the species. That is, if *r* is less than 10 percent the species should be conserved.

Assuming in a drought or other stochastic shock the sick, young and old members of the population will probably die-off first. This leaves the adult population, which is also the most valuable for breeding and population recovery. Assuming the same stochastic rate of reproduction, the core adult population would now have an intrinsic rate of reproduction of 25 percent. Now *E(25%) = r* for the owner to continue maintaining the wild stock. Twenty-five percent is a relatively high rate of return on an asset. Hence, the adult breeding population is now very valuable to the owner and consequently it is in the owners best interests to invest in protection for the species. Note that as the population recovers the intrinsic growth rate will decline.

For an accurate comparison, financial assets should incorporate inflation and tax components. Assuming in the economy the nominal rate of interest is 12 percent; the tax-rate on nominal interest is 33 percent and inflation is 4 percent. This leaves a real rate of return on financial investments of 4 percent. In this example, the species must have an intrinsic population growth rate greater than 4 percent for the owner(s) to invest in the conservation of the species. In both hypothetical scenarios this was achieved.

Many species have relatively high population growth rates. Natural population growth models such as the logistic growth curve show that at low levels of population many species have high growth rates (Hanley *et al.*, 1997). For example, Barbier *et al.* (1990) used a 10 percent population growth in a study on the Africa elephant. Er *et al.* (1995) presents the following estimates of recruitment success for swiftlets. The overall recruitment success of *Collocalia fuciphaga* was calculated at 14.4 percent and for *Collocalia maxima* was 10.0 percent. Saltwater crocodiles have an estimated population rate of increase of 6.5 percent p.a. (Webb and Manolis, 1993).

Assigning property rights to the species will also help to generate incentives for the owner to invest in management and protection of the remaining adult population rather than over-harvest the population. This could include erecting fences, patrolling and creating conditions to maintain the population growth. For example, Canadian Inuit voluntarily cut back on hunting polar bear when it was revealing the population was in decline although it increased economic hardship in the short-term (Freeman, 1998).

Sustainable use strategies usually focus on individual species, essentially commercial valuable species. For instance, the Zimbabwe, Botswana, and Namibia sustainable use proposals to CITES were specifically aimed at their elephant populations. However, the harvesting of a species should not be looked upon in isolation. That is, the inputs used to harvest one species could be switched to harvesting other similar species or used to maximise joint output. Dublin *et al.* (1995) uncovered a relationship between rhino and elephant poaching. That is, rhino poachers will continue to hunt rhino, however once the population is severely reduced they frequently turn to elephants (Dublin *et al.*, 1995). Moreover, Milner-Gulland and Leader-Williams (1992) while researching illegal poaching in Zambia identified that it was not profitable for poachers to hunt rhino alone but very profitable to hunt both rhino and elephant together.

Therefore legalising trade in certain endangered species (elephants) and not others (rhino) may not deter rhino poachers from killing elephants should they came across any while out hunting for rhino. Rhino's are still being illegally poached although a trade ban was implemented in the 1970's. For example, South Africa has a large white rhino population of about 6000 (Simmons and Kreuter, 1990). Poachers, especially organised gangs may target these populations and with some probably shoot any elephants they encounter. In short, if poachers are willing to take the risk of poaching rhino, well-defined property rights and the legal trade based on a sustainable offtake of elephants may not stop the poaching of these elephant populations.

Furthermore, hunting for the domestic market and local subsistence is a major factor in species sustainability. In a study on mammal species hunted in Equatorial Guinea the researchers found harvest levels to be up to 28 times sustainable levels (Fa *et al.*, 1995). There is however successful examples of species recoveries combined with commercialisation (Box 5.3).

Box 5.3: Crocodiles: a successful example of a species recovery program based on commercial incentives.

In the Northern Territory of Australia commercial use of wild crocodiles; the saltwater species *Crocodylus porosus* and the freshwater species *Crocodylus johnstoni* have generated positive conservation benefits and development opportunities for indigenous peoples. From the mid 1940's until the early 1970's saltwater crocodiles were commercially hunted with little or no controls. Consequently the population rapidly declined. In response, the species were officially protected from 1971.

Importantly because suitable natural habitat remained intact the species recovered relatively fast. By the late 1970's populations were abundant and by 1994 the population was estimated to be 60,000. Further, public opinion began to change about the commercial use of crocodiles. In particular private landowners should benefit from the use of crocodiles on their lands. 'The deliberate fostering of the commercial use of crocodiles, through farms, allowed landowners to gain commercially from crocodiles.' Private owners now invest in protection and maintenance of the populations on their land. In addition, crocodiles exist in areas occupied and owned by aboriginal people. The commercial use of crocodiles has generated economic opportunities such as employment in harvesting wild eggs, farming and tourism for these people.

Sources: Webb and Manolis, 1993; Webb *et al.*, 1996; Martin, 1997b; Moyle, 1997.

5.3.4 Problems limiting the effectiveness of legal trade and markets

The arguments supporting the commercialisation of wild species have some limitations and gaps. Many of these problems however are encountered in any other market for goods or services. That is, there is nothing from an economic perspective that will greatly abate markets for wildlife from functioning. However, it is worthwhile to review some of these constraints.

The literature seems to have focused largely on instituting the right conditions for the sustainable supply of wildlife. For instance, creating incentives for investment in wildlife, implementing trading systems, and estimating sustainable yields *inter alia*. In theory, it appears to be a relatively simple and straightforward task. However, there are two sides to trading systems and markets. Equally important is the demand or consumer side.

Demand destruction campaigns have been relatively successful in Western developed countries. Taking the trade in elephant ivory as an example Swanson (1994a) and Dublin *et al.* (1995) report that all available evidence to date shows that the pre-ban markets for elephant ivory, such as Europe and the United States, have largely disappeared. This was due to active campaigns by preservationists and animal rights movements in suppressing demand (Barnes, 1996). Further, highly emotive campaigns in Europe and the USA effectively destroyed the seal skin markets (Freeman, 1998). Finally, evidence from America suggests that re-establishing markets for American alligators *Alligator mississippiensis* Alligatoridae has been difficult (Simmons and Kreuter, 1990). Barrett and Arcese (1995, 1078), caution that, 'too little is known about demand for wildlife products to determine how to produce and market them so as to maximise net revenues, particularly if one wishes to jointly optimise herd size.'

Moreover, for political reasons as opposed to scientific, countries that have argued for greater protection, enforcement and trade bans in the past may be reluctant to (re)open commercial markets for wildlife products. For instance, a report on the commercial utilisation of Australian native wildlife received 341 submissions of which 192 opposed any commercial use of Australia wildlife (Senate Rural and Regional Affairs and Transport References Committee, 1998, xii). Further:

> ***Animal rights and welfare groups opposed to any use of wildlife by people have exerted political pressure on their governments to vote in ways that further their own philosophical goals. That is, they have tried and often succeeded in using CITES to stop or confound trade in any species, regardless of its real conservation status and the level of threat posed by international trade* (Er *et al.*, 1995, 4).**

For these reasons then, re-establishing or creating new consumer demand and markets may be very difficult. However, as demand destruction closes down some markets, new markets may emerge. For instance, the NIC's of South East Asia are now the major markets of elephant ivory. In short, the main issue is not closing down markets but generating new markets and demand that in turn benefits species conservation.

There may also be a risk of the market price decreasing when legal trade is established. That is, legalising trade may increase the supply of wildlife and products without increasing demand. However, the benefits of trade such as less wastage should outweigh this decrease in price if any. For instance, there is a high mortality rate among smuggled species. In a legal trading environment harvesters and traders will have to meet certain official regulations such as safe and humane transportation standards, this should result in a decline in unnecessarily deaths.

Trade could still be viable if prices fell. For instance, although raw elephant ivory reached US$300 kg before the trade ban, TATIS (1998) have calculated that even at US$100 kg this price generates incentives for marketing elephant ivory. In Australia the red-tailed black cockatoo *Calyptorhynchus banksii* competes with commercial rice cropping for land (Vardon *et al.*, 1997). International trade in this species is banned so its natural habitat is often converted for rice cropping. However, a single bird may be worth between AUS$9,000- $12,000 on international markets (Vardon *et al.*, 1997) whereas rice cropping is worth around AUS$150 per hectare per annum (Vardon *et al.*, 1997). If private landowners were able to legally harvest and sell the birds this should generate a much higher rate of return than rice. In consequence, natural habitat and the species are conserved. Even if the price fell the species should still be able to compete with alternative land uses (Vardon *et al.*, 1997).

Another issue is that owners may try to domesticate commercially valuable wild species and create conditions that increase the productivity of these species (Freese, 1998). This may well lead to habitat conversion into specialised use and result in a reduction in diversity that could reduce the stability of natural systems (Johnstone, 1995). For example, predators or non-commercial competitors may be purposely eliminated (Meffe and Carroll, 1994c). Owners may also try to manipulate the genetic diversity of marketed species to increase productivity or commercial quality (Freese, 1998). This is particularly true for some captive breeding and farming operations. In sum, although the species is preserved, genetic, species, and ecosystem diversity may decline. Nevertheless, captive breeding may be more optimal than extinction for the species. If the species is lost, then overall biological diversity declines.

According to Luxmoore and Swanson (1992, 180), 'the most non-intensive, non-specialised uses of the wilds render a straightforward conservation benefit when they are practised sustainably.' These actions are also very productive (Luxmoore and Swanson, 1992). Less intensive use provides a greater range of resources to be utilised. Consequently, there is a wider portfolio of species (Swanson, 1994a). Finally, indigenous species are better adapted to local conditions, and there is less environmental degradation (Luxmoore and Swanson, 1992).

Although the commercialisation of wild species poses some challenges, they are not unique problems. Commercial use offers additional benefits for conservation above those generated by protected areas and trade bans. More importantly, this use creates economic incentives such as increased income that rewards owners who wisely use the biological resources they own.

5.3.5 Summary on the commercialisation and trade in wildlife

Trade bans effectively reduce the economic value of the species. This will only lead the species down one of the other paths to extinction (Swanson, 1994a). In contrast, allowing wildlife to be traded the species will be able to compete with domestic agricultural species for scare base resources (Swanson, 1994a; Yu, 1994). Evidence suggests that wildlife utilisation is profitable and often greater than traditional mono-agriculture (Luxmoore and Swanson, 1992). This should lead then to greater investment in conservation and counteract the biggest threat to extinction, that of human disturbance.

Furthermore, legal trade in wildlife offers several other benefits. For instance, enforcement costs may be reduced if smuggling is crowded out. The official authorities and agencies are able to monitor species and products entering the country more effectively. It also allows for the transfer of information (biological and economic) from the local level.

5.4 CONCLUSION

In theory partnerships and sustainable use of wildlife provide possibilities for conservation of biodiversity and the economic development of local communities. Nevertheless, like any preservation, trade ban or development option there will be costs and benefits. Economic analysis is necessary to calculate the net benefits and to compare them to the other options and returns. In practise all strategies must first create incentives for investment. This will require some form of ownership regime. If the conservation strategy lacks a well-defined property right regime, unsustainable use, degradation, and species extinction may continue to occur.

Effective long-term conservation of biodiversity has to involve local communities. Moreover, it must provide them opportunities to increase their standard of living. The utilisation of local wildlife provides such an opportunity. International trade is important to the success of such a strategy. Trade also allows countries to diversify their range of exports. However, for long-term sustainable use, secure and well-defined property rights are an essential precondition. Without property rights wildlife will be quickly exploited for short-term gain on the international and domestic markets. An open access regime functions. To be effective, property rights must be assigned to local communities.

Importantly, commercial use does not mean only consumptive use. Wildlife has many other uses and values. Hence, the job of the economist is to identify and quantify these values. The following conservation case study illustrates the combination of local communities in Zimbabwe owning local biological resources and a sustainable utilisation strategy involving the African elephant.

5.5: A CASE STUDY OF SUSTAINABLE USE – THE AFRICAN ELEPHANT & LOCAL COMMUNITIES.

5.5.1 Introduction

The African elephant *Loxodonta africana* Elephantidae ranges over 37 countries in tropical Africa from open savanna to rainforests (Simon, 1995). This large mammal is a particularly interesting example with which to compare traditional protection strategies with the more recent innovative conservation initiatives of sustainable use and comanagement.

African elephant numbers have been in decline this century. The rate of decline accelerated in the 1970's and 1980's. Current population estimates (as of the mid 1990's) are in the region of 600,000, a decline from over 2 million in the early 1970's (Ehazi, 1997). However, these estimates need to be viewed with some discretion. According to the African Elephant Specialist Group (AfESG):

> *Most information on elephant numbers on the continent has been guesswork and has not formed a suitable basis from with to determine population trends over time, or to discern the effects of policy or management* (Said *et al.*, 1995, cited in Sharp, 1997, 111).

Rather, the AfESG classify numbers into four categories of descending reliability: definite, probable, possible, and speculative (Sharp, 1997). Regional population estimates are presented in Table 5.1. In short, reliable information on elephant numbers is lacking. This fact was also noted by Dublin *et al.* (1995). The WCMC in the Red Data Book has classified all populations on the African continent as vulnerable (Simon, 1995).

The African elephant was listed on Appendix II at the initial CoP's of CITES in 1976 (Barbier *et al.*, 1990). This listing however did not stop the rapid decline in numbers. In response many African countries imposed stricter restrictions on the hunting of elephants and the trade in ivory (Barbier *et al.*, 1990; Swanson, 1994a). For example, Zambia banned licensed hunting in 1980 (Leader-Williams and Albon, 1988). In 1985, a Management Quota System was adopted (Barbier *et al,* 1990; Wijnstekers, 1995). However, by the late 1980's ivory was still being illegally traded and the decline in numbers was not arrested. Consequently the African elephant was transferred to Appendix I in 1989, effectively banning all trade in elephant products between CITES members[6] (Simmons and Kreuter, 1990). This listing however has not stopped the poaching or decline of elephant, or prevented ivory being traded (Sharp, 1997). Rather, a study by Dublin *et al.* (1995, 84) concluded that, 'the illegal killing of elephants for their ivory remains a problem in the vast majority of range states across Africa' (See Appendix X).

Table 5.1: Selected African elephant population estimates.

Regions (no. range states)	Definite number	Probable number	Possible number	Speculative number	Total	% total
Central (7)	7,320	81,657	128,648	7,594	225,219	37.77
Eastern (8)	90,482	16,707	19,999	1,084	128,272	21.51
Southern (9)	170,837	16,402	18,983	21,825	228,047	38.25
West (13)	2,760	1,376	5,035	5,554	14,725	2.47
Continental total	286,234	101,297	155,944	36,057	579,532	

Source: Said *et al.*, cited in Sharp, 1997, 112.

5.5.2 Causes of decline: poaching and other factors

Poaching has been identified as the primary and immediate cause of the decline in elephant populations. The elephant's tusks are the main target of the poacher (Barbier *et al.*, 1990; Simmons and Kreuter, 1990; Dublin *et al.*, 1995; Sharp, 1997). Poaching was prompted by the strong and large international demand for ivory. Around 80 percent of ivory was destined to the USA, Europe, and Japan (Barbier *et al.*, 1990). According to Burgess (1994), the export of ivory increased during the 1980's reaching a peak of 1,031,934 kg in 1983 but declining in tonnage after that. Barbier (1991) has estimated the annual value of ivory exports from Africa in the 1980's to be around $US50-$60

[6] Also see http://www.traffic.org/brf_elephants_cites.html

million. An indirect cause of decline is competition between elephant and local agro-pastoralists for base resources such as land and water.

Biological factors have helped in the decline of the elephant. For instance, while females first calve at about 13 years and have a five year interbirth period (Barbier *et al.*, 1990) male elephants do not become active breeders until about 35 years of age (Simon, 1995). Further, it takes approximately 6 and 9 years for male and female elephants respectively to generate a 1 kg tusk of ivory (Barbier *et al.*, 1990). Although the tonnage of poached ivory has decreased, the actual numbers of elephant killed has increased (Barbier, 1991). This is because by the late 1980's most of the big tusk bull elephants had already been shot. Poachers shifted their attention then to cows and calves which have smaller tusks (Barbier, 1991).

Many other factors have also helped accelerate the decline in elephant numbers. These include a high price/cost ratio. For example, the price of raw ivory increased from $US60 to $US150 per kilogram during the 1980's reaching $US 300/kg before the trade ban was introduced (Burgess, 1994). However, the price paid to the local hunter for the cost of harvesting was in the order of $US5-10/kg (Swanson, 1994a). Barbier estimated that African countries only captured about 5 percent of the value of raw ivory exports during the height of the ivory trade (Brown *et al.*, 1993). Hence, intermediaries and exporters/importers may have opportunities to make large profits from the illegal trade.[7] Compounding this is the ready availability of weapons in Africa that contribute to the large number of potential poachers (Barbier *et al.*, 1990; Simon, 1995).

Further, management costs for elephant protection are high. That is, elephants are migratory (fugitive) animals thus it is difficult and expensive to monitor them (Barbier *et al.*, 1990). Combined with a lack of management expertise, limited budgets, and weak and poor enforcement, effective conservation was difficult to achieve. In addition, maintaining habitat is associated with high opportunity costs, especially for local communities. This is in the form of loss of access to resources in protected areas and crop damage caused by elephants as well as human-elephant conflicts. In consequence, disenchanted local communities may decide to work with commercial ivory poachers (Dublin *et al.*, 1995).

Corruption is also widespread in the ivory trade. This goes up to the highest official and government levels. For example, Kenya's top wildlife civil servant, John K.

[7] This can not be concluded with much certainty unless costs are known.

Mutinda was involved in poaching and smuggling during the mid 1970's (Simmons and Kreuter, 1990). During the Angola war, the Cuban military became involved in ivory smuggling (Simon, 1995). The South African Defence Forces were also heavily embroiled in killing large numbers of elephants and rhinos in Mozambique (Simon, 1995).

Accordingly, the decline in elephant numbers in some Africa states can then be attributed to the many factors mentioned above. In short, biological factors, high price/cost ratio (in relation to the discount rate), open access and other issues such as level of enforcement and detection create conditions so that it may be optimal from the standpoint of the poacher to harvests the population as quickly as possible (as per the Clark extinction model). Further, the trade bans eliminates valuable consumptive use values of the elephant, in consequence the natural habitat for the elephant is converted into alternative land uses, adding to the demise of elephant populations.

Elephant populations, the degree and intensity of poaching however vary across the range states. For example, in the Selous Game Reserve (50,000 km^2) located in southern Tanzania one of the largest wildlife sanctuaries in Africa, the elephant population numbered some 110,000 in the early 1970's but due to poaching they were reduced to about 30,000 in the late 1980's (Simon, 1995). In aggregate numbers, the elephant population in Tanzania decreased from 316,300 in 1979 to 61,000 in 1989 (Swanson, 1994). In the Tsavo National Park in Kenya, poaching has reduced elephant numbers from 45,000 in 1965 to 6,000 by 1985 (Simon, 1995). Overall, the elephant population in Kenya has fallen from 65,000 in 1979 to 19,000 in 1989 (Simmons and Kreuter, 1990). In Zaire, numbers have decreased from 377,000 in 1979 to 112,000 in 1989 (Swanson, 1994a). Nevertheless, although poaching is rife throughout Africa, the decline in elephant numbers is not widespread across all the elephant range states (Simmons and Kreuter, 1990; Burgess, 1994).

In contrast, some states most noticeably the southern African states their elephant populations have increased. Zimbabwean numbers have increased from 30,000 in 1979 to 43,000 in 1989 (Simmons and Kreuter, 1990). In Botswana, surveys undertaken between 1989 and 1992 indicate that numbers have increased by 25 percent since 1989 (Barnes, 1996). In South Africa, they have brought their elephant populations up from under 1000 in 1900 to 10,000 in 1994 (Sharp, 1997). The southern range states now account for almost 40 percent of the total population on the African continent (Table

5.1). Research should be carried out to discover why some elephant populations are healthy and increasing while others are near to extinction.

Evidence to date suggests that the different management regimes are responsible for the decline and recovery of elephant populations. That is, the conservation problem is not biological but of an economic nature. It appears that some states have invested in protecting their elephant populations while in other states with poor enforcement, poaching is rampant and in consequence populations are declining (Swanson, 1994a). For instance, Zimbabwe has comprehensive wildlife legislation; offenders are dealt harshly (Dublin *et al.*, 1995) and poachers are shot on sight (Simmons and Kreuter, 1990). These enforcement actions may increase the probability of detection and the costs of the illegal activity. In turn, this acts as a deterrent to the poacher, who may now reduce their illegal harvesting effort.

Although poaching is a direct cause of the decline (and the activity that a large percentage of conservation resources are allocated to halting) the fundamental reason is a lack of incentives to invest in long-term conservation. Harvesters have no incentive for long term investment due to the open access nature of elephants. As stated, the current trade ban reduces the economic value of elephants, which will act as a barrier to landowners to investment in elephant management. Hence, the poacher, the landowner and also the state have little incentive to invest in elephant conservation. For long-term conservation other strategies are needed (Barnes, 1996).

Elephant are seen as valuable, having important use and non-use values. Banning the international trade in elephant products reduces the value of elephants and creates a disincentive for investment. Swanson, (1994a, 72) argues that, 'elephants do not demonstrate the sort of characteristics that make an asset worthy of the substantial investment (of natural and government resources) that this species requires for its sustenance.' A total trade ban also discriminates against those countries that have invested in elephant because they will not receive a return on this investment, thereby signalling that they have chosen the wrong option and disinvestment may now occur. Leader-Williams and Albon calculated that for a zero population change in elephant numbers it would cost about $US215/sq. km. Few of the Africa states have invested this amount of money, for example, Tanzania invested $US18/sq. km. in 1986, while Zimbabwe invested $US194/ sq. km. in 1986. Dublin *et al.* (1995) report that law enforcement costs are likely to have doubled since the mid 1980's while at the same time law enforcement budgets are declining.

Today, law-enforcement budgets in the majority of Africa's protected areas are less than 5% of the US\$200/km² figure required to guarantee the integrity of protected areas and the safety of the "flagship" species within them....Staffing levels in parks and protected areas have fared no better (Dublin *et al.*, 1995, 6).

Ironically in the states that have invested in elephant's protection the population increase is causing problems. South Africa and Zimbabwe have had to introduce population control measures such as culling or the removal of live elephants to other areas or have had to purchase more land to be set aside as wildlife areas. In addition, the large and increasing populations are destroying habitat. This is a cost to these African states and specifically local communities. Nevertheless, they are banned from selling any ivory (recovered from culling or natural deaths) which may be used to offset some of these expenses or provide funds for conservation and local development projects. According to Simon (1995), it makes no sense to deny local people the benefits if elephants have to be shot. Finally, all range states have ivory caches stockpiled from confiscated illegal operations but are unable to sell the ivory (see Dublin *et al.*, 1995 for individual states' stockpiles).

In summary, the lost of potential revenue from the trade in elephant ivory acts as a disincentive for states to continue investing in the future conservation of their elephant populations. In the long-term this may also have adverse effects for other species that rely on the elephant as a keystone species performing its ecological function.

5.5.3 The Southern African sustainable use proposal

Since the uplisting to Appendix I in 1989 several African countries have campaigned to downlist their populations to Appendix II. In 1997, at the 10th CoP's conference the African elephant was downlisted to Appendix II, but only for the Botswana, Namibia and Zimbabwe populations, and only under certain strict conditions (Pendry, 1997; Sugal, 1997). Trade will commence in 1999 for a two-year test period with an experimental quota to Japan only (Pendry, 1997; TATIS, 1998). The experimental export quotas have been set for unworked (raw) ivory. Zimbabwe is also allowed to export a limited quantity of hides, leather goods and ivory carvings (Sugal, 1997).

This experimental trial has been criticised, from several points of view. For instance, opponents of restabilising a limited trade argue that it will legitimises the illegal trade, and this could see an increase in poaching (Sugal, 1997). Allowing only three countries to trade may create a loophole where ivory from unsustainable harvests carried out in other countries may be smuggled into and then legally re-exported from those three countries. Further, after the two year test period the trade ban may be reinstated. In consequence, producer and consumer states may take a short-term (finite) view and continue to sell and purchase illegally poached ivory (Moyle, 1998d). This is because if the parties believe that the market will be closed after the trial countries have an incentive not to cooperate but to cheat and continue purchasing poached ivory. However, if the trial and the market are of infinite length it is in the best interests of the countries to cooperate. These suggested outcomes follow those from game theory

TATIS (1998) has developed an International Ivory Exchange System (IIES) for the monitoring of the international ivory trade. It addresses some of the problems that have been raised about the experimental quota. The IIES comprise the following components to monitor the trade effectively and efficiently (TATIS, 1998, 5):

- An internet-based electronic auction floor - a transparent and effective means of maximising the revenue generated from the sale of ivory;
- A marking system for tusks - the use of low-cost two-dimensional bar code technologies to tag elephant tusks with tamper proof bar codes containing full documentation relating to each tusk;
- A networking environment - providing complete transparency and reconciliation potential at every stage in the process.

The technologies proposed should counter the past abuse and corruption associated with the ivory trade (TATIS, 1998). The transparency would make any abuse difficult to hide, in that each stage of the trade would be monitored and the information available to all the parties (TATIS, 1998). There would also be checks before moving onto the following stage of the transaction (TATIS, 1998). That is, from the auction to the purchase to the certificate processing and shipping stages. In short the trail would be 'marked' (Moyle, 1998d).

5.5.4 Value of elephants

Elephant have other values apart from that derived from ivory (a consumptive use). For instance, Brown and Henry (1993) estimated the viewing value (non-consumptive use) of elephant from tourists in Kenya to be between $US22-30 million annually.[8] They used both travel-cost and contingent valuation methods to calculate this value. [9] This is as much as ten times the value of its (poached) ivory exports (Barbier 1991). The surveys also concluded, that should poaching of elephant continue with a corresponding decline in population this would have a large negative effect on tourist numbers. It would also result in a large finance loss from lost tourists' expenditure. They calculated that overall tourism expenditure would decline between $US52-103 million annually should elephant numbers decline between 25-50 percent (Brown and Henry, 1993).

Barnes (1996), has analysed the effects of the CITES trade ban and the changes in economic use value of elephant in Botswana. Barnes calculated that the trade prohibition has reduced these (use values) to half (53 percent) of what they were in 1989. He concluded that the trade ban has jeopardised the future of the elephant in Botswana (Barnes, 1996). The main effect will be conversion of elephant habitat to domestic livestock use in the future. Barnes offers a solution to improve the conservation of elephant. This involves maximising the TEV, i.e., the direct use, indirect use, and non-use values. In maximising the TEV, elephants will be seen as competitive to alternative returns on the use of the land and investment in the conservation of elephants should increase (Swanson, 1994a; Barnes, 1996; Freese, 1998).

These reports show that the African elephant have a variety of values. The Kenyan survey calculated just the viewing value, while the Botswana survey showed the detrimental effects a trade ban could have on the value of elephants. In Kenya should poaching of elephant continue this could effect tourism. While in Botswana a country that has invested in the protection of their elephants the trade ban effectively penalises them. The loss of economic use value results in less revenue for the two countries and this leads to less investment in elephant conservation.

Elephants also have value in the functioning of ecosystems (Burgess, 1994). Elephants keep the forest from encroaching on the savanna, and importantly in the dry season they enlarge water holes where other animals come to drink. This can be positive and negative depending on the population size however. For instance, increasing numbers destroy vegetation for other species, put pressure during the dry season on

[8] Data from 1988/89 was used in the analysis.
[9] These non-market valuation techniques are used to estimate values when no actual or surrogate market exists.

habitats and can be a nuisance for local farmers. Large elephant populations can change forest landscapes towards open grasslands. This reduces available habitat for woodland mammals such as black rhino *Diceros bicornis*, the lesser kudu *Tragelaphus imberbis* and the gerenuk *Litocranius walleri* (Simon, 1995). Whereas, in the south-east of the Tarangire National Park in Tanzania uncontrolled poaching and hunting of elephants and rhino has led to bush encroachment. This is turn caused an increase in tsetse flies, which reduced the population of domestic livestock affecting agricultural productivity in the area (McNeely *et al.*, 1990).

Further, elephant society is also based on small and strongly matriarchal family units (Simon, 1995). The poaching of mature females for ivory (mature bull males having already been poached for their larger tusks) orphans calves, leaving them alone, weak, and vulnerable, lessening their survival chances (Simon, 1995). In addition, the loss of the social structure provided by the females discipline has resulted in behavioural problems in surviving calves. It has been observed in private game parks that imported young elephant without adult guidance have, 'formed gangs; they terrorise other wildlife; and, they hide from the tourists' (Anonymous, 1996). The problem is these values are difficult to measure. There needs to be more research into both the elephant's ecological function and how to calculate these values. In addition, local communities must value elephants as a resource to conserve.

5.5.5 Sustainable use of elephants and local communities

Elephants have several commercial uses. These include consumptive uses such as trophy hunting, and non-consumptive uses such as wildlife viewing. However, sustainable use of elephant will not be successful unless rural Africans are involved in management (Simmons and Kreuter, 1990). According to Metcalfe (1995, 276), 'a way exists to maintain Africa's large mammal diversity in savanna land use by facilitating the establishment of local wildlife management systems, linked with the government protected areas.' Barbier (1992b, 105) details some advantages of community-based wildlife management. These are as follows:

- potential to promote rural development;
- increase local income;
- improve standard of living;

- strengthen local community structures; and
- empower local communities.

In sum, if local communities can benefit economically from elephants then they will value them and this creates incentives to protect the elephant from illegal poaching. However, for a successful sustainable use strategy and for markets to operate efficiently a property rights system needs to be implemented. The next section examines a community based property rights system and how wildlife conservation and local communities can benefit.

5.6 A PARTNERSHIP IN CONSERVATION: CAMPFIRE

In Zimbabwe wildlife-based tourism is a major foreign exchange earner. In 1990 safari and tour operators earned the country around US$52 million (Ceballos-Lascurain, 1996). This was derived from consumptive and non-consumptive use of wildlife. Further, it involved operations not just on national parks, but importantly private and communal lands. In fact, there has been a rapid increase in land devoted to wildlife since the 1980's (Metcalfe, 1995; Ceballos-Lascurain, 1996). CAMPFIRE is a pivotal part of these achievements.

In Zimbabwe it is thought the best way to protect elephants and other wildlife is to give local communities the opportunity to benefit from their presence (Simmons and Kreuter, 1990; Pye-Smith *et al.,* 1994). CAMPFIRE is an example of sustainable use of wildlife by local communities in Zimbabwe. CAMPFIRE provides a framework and support for developing community-based wildlife management on communal land that covers 42 percent of the country (Freese, 1998). CAMPFIRE was established in the mid 1980's and is a comanagement conservation partnership between the state, the Department of National Parks and Wildlife Management (NPWM) and rural communities(Metcalfe, 1995). By 1993 nearly 400,000 people on some 30,000 sq. km. of land had enlisted in the program (Freese, 1998) (see Appendix 3).

CAMPFIRE represents efforts by rural communities to manage wildlife on an economic basis and to secure direct financial benefits (e.g. employment, cash) from wildlife (mainly elephant, Cape buffalo, several antelope species) (Barrett and Arcese, 1995). The communities are the main stakeholders, in that they legally own the wildlife[10] (Lewis and Alpert, 1997). By taking responsibility for wildlife on their land they can directly benefit from managing those resources. However, the state still has a supporting role, the NPWM acts in a technical role and local authorities have a legislative role (Metcalfe, 1995).

> *The CAMPFIRE program argues strongly in favor of sustainable use being the springboard for large mammal diversity integration into communal land use practices. A wise use approach outside a protected area is effectively subsidized by the preservationist approach of the protected zone. The park may be perceived as an ecobank supplying interest in the form of a renewable supply of wild animals. The softening of the "hard edge" between zones has always been a goal of buffer zone approaches, but too often the relationship between people and park has been asymmetrical and not a genuine meeting of land uses and authorities* (Brown 1991, cited in Metcalfe, 1995, 273).

Local communities are given ownership and control of wildlife on communal land, (i.e., land outside national parks). In essence they paid the costs of managing the wildlife, such as the loss of land for possible agricultural use, but retain the benefits of preserving the land as natural habitat. CAMPFIRE specifically addresses the problem of communal resource ownership, it provides for a more equitable allocation of natural resources and places a value on them that had in the past been absent for the communal land residents (Barbier, 1992b). In essence, CAMPFIRE is about land use and the appropriate and equitable use of wild resources.

Revenue from safari hunting totalled US$1.5 million in 1995 (Freese, 1998). Specifically to hunt a bull elephant safari operators may charge around US$1,000 a day to hunters, and an additional US$7,500 trophy fee (which goes to the local community) for the elephant itself (Pye-Smith *et al.*, 1994). Recreational hunting accounts for some 93 percent of total revenue from all forms of use (Freese, 1998). Money from tourism, selling hunting licenses and big-game hunting fees goes directly to the local community. For example, in the Kanyurira ward of the Guruve distinct each individual household received a benefit of US$384 in 1993. In contrast, the statutory minimum monthly wage is US$40 (Furze *et al.*, 1996). Additional benefits include local employment opportunities, infrastructure investment (e.g., schools, clean water, etc). In consequence, local communities attitude towards wildlife and conservation is positive (Lewis and Alpert, 1997). Importantly for the wildlife, the success of CAMPFIRE has brought about a doubling of the elephant range in Zimbabwe to 30 percent of the country (Anonymous, 1997b) and elephant numbers are increasing (Pye-Smith *et al.*, 1994).

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[10] Unlike ADMADE (Box 5.1) where the Zambian government owns the wildlife.

In addition, private land in Zimbabwe since the 1960's has also been utilised for wildlife conservation with commercial farmers deriving a return from wildlife use (Barbier, 1992b). For example, of Zimbabwe's 4,500 farmers, approximately 500 are now actively involved in wildlife production, such as sports hunting, meat production, and eco-tourism covering 2,700,000 hectares (Pye- Smith *et al.*, 1994). In 1990 US$3.1 million in revenue from safari hunting on private commercial ranches was generated (Freese, 1998).

Table 5.2: Land area (km²) used for wildlife conservation and utilisation in Zimbabwe.

Year	National parks	Safari areas	Forest areas	Communal lands	Commercial farms	Total	% of Zimbabwe
1930	17,500	0	0	0	?	17,500	4.48
1940	10,583	0	0	0	?	10,583	2.71
1950	11,075	0	0	0	?	11,075	2.83
1960	11,800	0	0	0	350	12,150	3.11
1970	26,073	7,494	0	0	30,000	63,567	16.26
1980	22,799	18,576	5,541	3,356	30,000	80,272	20.54
1990	22,799	18,576	4,963	12,806	27,000	86,144	22.04

Source: Cumming 1990 cited in Barbier, 1992b, 108.

In summary, wildlife has to save itself. Zimbabwe has shown that private and communal ownership combined with commercial sustainable utilisation of wildlife can successfully combine conservation and economic development. Such projects improve the cooperation of local communities towards conservation. In areas where local people receive direct benefits and actively participate in CAMPFIRE wildlife conservation is much better than in areas where local institutions are weak (Gibson and Marks, 1995). However, it will be of interest if this model based on the use of mainly large mammals that attract large numbers of tourists can be applied to other species or other areas.

CAMPFIRE can not claim to have achieved all its objectives (Metcalfe, 1995, 273). There are some problems with CAMPFIRE that need addressing. This includes the distribution of revenue, between districts and between the partners. For example, the

CAMPFIRE areas with high wildlife population densities and consequently higher economic rewards have had a human population increase (Freese, 1998). People are immigrating to those areas to take advantage of the socioeconomic opportunities. This could cause land use and distributional conflicts that could result in a breakdown of the existing institutions governing the use of local wildlife.

Although CAMPFIRE districts own the local wildlife they have to share the revenue with other stakeholders. In consequence, many communities and individuals still practise livestock farming. This is because all the benefits from this economic activity accrue to the individual. Again this may lead to conflicts over land use. However, if local communities received the full financial benefits from wildlife then this problem may be solved.

Nevertheless, CAMPFIRE has established a framework for the integration of local community management of wildlife and conservation. CAMPFIRE demonstrates 1) community participation; 2) links conservation objectives to economic development; 3) to achieve this linkage property rights are emphasised; 4) community institutions are reinforced from the bottom-up but with a comanagement partnership with the state. In summary, CAMPFIRE has generated benefits for wildlife in Zimbabwe, specifically elephants, and as well as increasing the economic welfare of local communities.

6
Conclusion

But arguably the single most direct measure of the planet's health is the status of its biological diversity....biodiversity is the basis for our existence **(Tuxill and Bright, 1998, 41).**

The maintenance of biological diversity depends on the integration of social, biological, and economic factors. The future requires a new approach to basic needs that encompasses physical and emotional human needs as well as the needs of the ecosystem that sustains both **(Metcalfe, 1995, 277).**

The human species face two major challenges on Earth: first to survive and second to improve the quality of life. Both require the appropriation of biological resources. It is important that these resources are efficiently and optimally utilised. Well-defined and enforceable institutions are necessary to legitimise rights and duties, to ensure efficient resource allocation and long-term sustainability.

Biodiversity is utilised for its direct and indirect use as goods or services in economic development activities. In the past, economic growth and development adversely affected the natural environment. This is especially true with respect to the status of many wild species and in turn overall biological diversity. The consequences of these impacts are largely unknown. What is known is that species extinction are irreversible and that environmental quality and biological diversity does not have to be needlessly sacrificed.

Therefore it is in our best interests to conserve natural habitats and a diverse range of species. Maintaining habitat in its natural form is crucial and the most cost-effective way to ensure the survival of the greatest number of species and the continuation of ecological services (Furze *et al.*, 1996). Strategies such as protected areas and trade bans were thought to be best for conserving biodiversity. Most often they involve a centralised absolute protectionist and anti-use approach to conservation. Institutions are created to enforce the rules and regulations protecting the remaining wild species. At the country level the state assumed control and management of large areas of natural habitat, while at the global level, CITES regulates the trade in endangered species. However, in many instances these institutions are weak, poorly enforced and have been neglected.

There is now real concern that such strategies are unsuccessful in preventing the decline in biological diversity. This study has identified issues and constraints that limit the effectiveness of these strategies. They were discussed in Chapter 4. The main points are; first, protected areas are now largely inadequate in physical size to protect the diverse range of wild species. Second, there is a lack of sufficient funding for effective management and enforcement. Finally, CITES has not effectively stopped the trade in endangered species.

Governments in all countries have a limited capacity to efficiently and effectively protect biodiversity. For example, Australia has banned the international trade in its native wildlife, yet wildlife has been smuggled out of that country undetected for years. These problems are especially acute in developing countries. In many developing countries wildlife laws are weak and poorly enforced, corruption is rife, and the returns from poaching and smuggling are very attractive to many individuals compared to alternative income opportunities. As institutions regulating the use and protection of biodiversity breakdown this may lead to an open access situation. Hence, it is difficult to halt the illegal encroachment and poaching from protected areas and over-exploitation results, as per the economic extinction models.

Compounding these problems are large numbers of people on land surrounding protected areas who rely on the natural environment for both subsistence and as a source of income. These are mainly rural communities in developing countries whose relative poverty is a development concern. They are also the most affected by any conservation policy or program designed to protect wildlife.

State management of biodiversity also creates externalities. That is, the protectionist approach imposes cost onto others. These costs include the opportunity cost of not converting the land to agricultural production for example, and the cost of crop and livestock damage caused by fugitive wild species. Typically, it is the local communities that bear most of these costs. Externalities are also generated from an open access regime. That is, the quantity of wildlife harvested may be above the socially optimal level. Conservation policy needs to find ways to internalise these externalities.

In essence, as long as wildlife remains in the public domain, private individuals and community groups will not invest in conservation. If protectionist conservation strategies are failing, new institutional arrangements must evolve to replace the existing institutions. Conservation management must therefore change, it must equate to "wise use" and private individuals and communities must take more responsibility. In developing countries protected area management, local people and the use of wildlife must all be integrated. If incentives can be generated to change the behaviour of the exploiters towards conservation then the other mentioned difficulties in protected area management may be reduced.

The challenge is to develop and implement strategies that restores the natural environment back to a healthy state while balancing the needs of economic development and expanding the economic opportunities for local communities. A major institutional shift requires that individuals or groups have access to wildlife before they will invest resources into conservation. Partnerships and property right institutions offer solutions to improve the management of biodiversity.

Comanagement of habitat and wildlife is one option that involves partnerships between the state, and private or communal owners. ICDP's are an example of comanagement initiatives. These approaches delegate rights, obligations and responsibility to the co-owners. In general, local communities own the land while the state owns the biological resources. The state should also have the responsibility to legitimise and enforce property rights and duties, promote markets and promote incentives for conservation and disincentives that degrade and over-exploit the natural environment. In addition, the state has to manage the economy so that there are no perverse incentives and no barriers to restricting trade and the creation of wealth.

Conservation partnerships also delegate responsibilities to local communities. They need to be consulted and involved in issues regarding ownership and the use of local resources. A major factor overlooked with a centralised approach is that solutions to many conservation problems and management constraints may exist at the local level. According to Hayek the most relevant information is likely to be held at the individual, firm or household level (Lal, 1985). Local people have valuable information and knowledge on species that they have traditionally hunted and harvested. The use of this knowledge could help improve conservation management. Conservation partnerships can also assist local communities to develop and increase their standard of living. Important to achieving this objective is a wildlife utilisation strategy that can generate a flow of revenue. This will require some form of ownership (private or common property) that sanction local communities' rights to the base resources and/or wildlife. These rights may include the right to use the land, to harvest wildlife, or ownership of the informational value of biodiversity.

Socially sanctioned institutions legitimises owner's rights for control over the benefit stream arising from the use of the resource and non-owners duties to refrain from illegally taking the benefits. Property rights will be effective in this respect as long as an authority system exists to enforce the rights and duties. Authority systems need not be centralised government agencies; they can evolve at the local level. This process has been observed in many fishing communities (Berkes *et al.,* 1989). Further, some Inuit communities decided at the local level to reduce hunting of polar bears when numbers were decreasing, they also established their own rules regulating the hunting of the polar bear (Freeman, 1998). The successful outcome of these local actions is that polar bear numbers have increased. In short, property rights allocate responsibility into the hands of those who earn the benefits and pay the costs. Owners if acting for their own self-interest should utilise the resource efficiently with due consideration for the future, thereby avoiding resource exhaustion.

Private property or common property appears to be best at creating these incentives. Although common property and private property are very similar i.e. nonowners can be excluded, the current economic and political situation in many developing countries favour common property regimes especially in rural areas. A reason that favours common property in developing countries is the difference in transaction costs between private and common property regimes. However, should these conditions change then private or even state management may become the more appropriate regime.

Property rights give owners the choice in how they wish to use the resources they own. Secure property rights while contributing towards encouraging prudent management of the world's scare resources do not guarantee long-term well-being (Mendelsohn, 1994). Also needed are commercial opportunities to exchange property rights that generate profits for owners. Simply, to be conserved, wild species must compete with alternative uses of the habitat. Markets offer this opportunity.

If these rights can be exchanged in a market this will reinforce the value of the resource, and there may be efficiency gains. For instance, the Inuit can exchange their cultural right to hunt polar bears for the price that a trophy hunter would pay to hunt a polar bear (Freeman, 1998). The main point here is that the Inuit as owners have the choice in deciding who has the right to use the resource (polar bear). If owners have control over the decisions pertaining to the future use of the resource, this generates incentives for a long-term sustainable outlook.

Wild species have many values including utilitarian and intrinsic values, use and non-use values, some which have commercial value. Markets are needed so that these values can be maximised. In turn, this should generate incentive for conservation. Most important for conservation is that the species TEV is greater than alternative uses of the base resources. This is why economic analysis researching and identifying the different values, and use opportunities of biological resources is important. Examples presented in this study indicate that the TEV of biological resources are often greater than alternative uses. However, thin markets, the public good aspect of biodiversity, and trade bans often limit maximising TEV. In consequence, some biodiversity will be lost.

Large populations of species are also located on habitat outside protected areas that is privately or communally owned. Although these species may have potentially valuable commercial uses, in many instances owners are prevented from maximising this value. For example, an Appendix I CITES listing bans any commercial trade in that species. Wildlife now imposes a cost onto the base resource owner's. Simply, those who have wildlife on their land should receive some economic benefit.

A major theme throughout this study is that local communities should have economic opportunities to develop. Sustainable use offers a practical means for local communities to increase their income and wealth. Sustainable use is a biological conservation strategy that assumes the value of the wild species will foster development that is compatible with conserving biodiversity (Furze *et al.*, 1996). This study has presented evidence and case studies illustrating the benefits that can be generated from the commercialisation of wildlife. Community-based initiatives such as CAMPFIRE are usually both efficient and effective, generating conservation and development gains.

Sustainable use has several other benefits at the local level. First, its commercialisation may lead to wage employment and increased income. Second, it acts as a quasi-option value, that is, it is a form of insurance. For instance, should wage employment declines and individuals return back to the subsistence sector, they have the option of harvesting wild species to meet their basic needs. Further, the use of wildlife is just not conserving biodiversity it is also conserving cultural diversity. In short, give local people ownership of local resources, allow them access to markets and they will change their behaviour towards wildlife and manage it for their own self-interest.

Local communities have traditionally meet their basic needs from utilising the local natural environment for thousands of years. However, for local communities to increase their standard of living, income, and wealth will require some form of economic development. Commercial use and trade of local wild species offer an opportunity. This will also require access to outside markets. However, current conservation strategies act as barriers to this exchange. The development challenge is to link local communities, wildlife and markets.

CAMPFIRE is an example of institutions developed at the local level for managing wildlife resources. CAMPFIRE empowers local communities to take responsibility for the costs and benefits (rent) generated from conservation and management. CAMPFIRE has achieved positive conservation gains such as increased habitat for wildlife and the recovery of elephant populations in Zimbabwe. Further, CAMPFIRE has improved the standard of living for the communities involved in the project. This includes, increased income, education and health facilitates, in general the communities basic needs. In essence, what makes CAMPFIRE successful is the decentralisation of property rights and the allocation of wildlife to local communities, combined with the commercial use of local wildlife.

There is concern that economic growth and development pressures will accelerate the loss of biodiversity. However, if biodiversity were accurately measured and valued for its contribution to productivity and overall economic growth then there would be investment in conservation. As biodiversity becomes scarce, its use value and also option and existence values should increase. Hence, countries endowed with biological diversity (natural capital) will become wealthier. The economic growth models could emphasis this and incorporate natural capital as a source of economic growth. In consequence, these countries would view conservation as an investment into the future.

Moreover, with a comanagement approach protected areas can act as an eco-bank, maintaining wild species as a form of wealth. In addition, these areas can supply a sustainable flow wildlife to surrounding areas which is then utilised in consumptive and non-consumptive ways by the owners. In turn these uses create a stream of revenue that can then be invested back into wildlife conservation generating more wealth. In essence, the returns generated from the harvesting and use of wildlife on surrounding lands is the opportunity cost of maintaining the area in a natural state.

Legal commercial use offers further benefits. First, compared to the illegal market, the legal trade is much more transparent. This will in turn free resources that had previously been allocated to enforcement. Second, developing countries have a CA in biological resources. Trade theory points out the advantages gained from countries trading in CA goods and services. This does not necessarily mean the physical trade of biological goods. Rather, developing countries could trade the rights to use these resources. For instance, this could include the right to retain tropical forest for their carbon storage value and the informational value contained in biodiversity. The Nature Conservancy for example purchases natural habitat in developing countries. What The Nature Conservancy is purchasing is the right to use that habitat, in The Nature Conservancy case this is to leave it in a natural state and conserve biodiversity.

The limitations, effectiveness, and criticisms of CITES as a wildlife trade treaty were discussed in Chapters 4 and 5. The Convention has the mechanisms for the sustainable trade in wild species such as using Appendix II. The constraint on the effective use of these mechanisms is the political process by which species are listed and the voting system for sustainable use proposals.

Regional trade agreements are an approach that has the ability to overcome these constraints of CITES. These regional trade agreements decentralise trade decisions to between producers and consumers of common goods. For instance, the swiftlets bird nests are produced and consumed in Southeast Asia. Hence, it is in the best interests of the producing and consuming countries that this species is conserved as it generates benefits for the whole region.

Microeconomic theory points out the efficiency benefits, gains and discovery process of markets. For instance, the commercialisation of crocodiles has generated positive conservation benefits. Wild reptile eggs collected for commercial farming and ranching and the return of juveniles can increase their rate of survival by 5000 percent (Swanson and Barbier, 1992). In Zimbabwe the conversion of agricultural land from beef production back to wildlife production has seen the natural system recovery very rapidly. Annual and perennial grass have come back, springs have reappeared, and wildlife is returning (CPC, 1999). In turn, tourism has increased and other countries in Africa are purchasing wildlife from Zimbabwe to restock depleted populations (CPC, 1999). The amount of money from wildlife is two to four times above that from cattle and it is sustainable (CPC, 1999). In short, if markets did not exist it is doubtful that these conservation benefits would have been discovered.

Sustainable use has largely focused on commercially valuable species e.g. elephants, crocodiles and parrots. These species only represent a very small percentage of total biological diversity. There is a need for biological research on the sustainable use of other species and how current sustainable use affects other species. Nevertheless, a bottom-up approach that has institutions, property rights and markets can generate positive feedback on such issues.

The trend to return wildlife management back to local communities is quite recent. Hence, research and a proper understanding of local social, cultural and economic issues are necessary. Most important is the relationship between local people and wildlife and how commercial use will alter these relationships. Further research is needed on institutions, rights and duties. Institutions are dynamic, as the economic and biological situation change, institutions will have to evolve or adapt to these changing conditions. For instance, what social institutions can effectively protect biodiversity, especially in light of the growing importance of the informational value contained in biodiversity.

There is no one ideal management plan or any easy answers. Each species (even separate populations) have different natural histories, biological requirements, population dynamics, cultural, and economic values. Importantly suitable habitat must be available to the species for it to recover. Past conservation strategies overlook many of these issues and consequently failed. It is clear that change and adjustment in conservation thinking is necessary, but the transition is complex and complicated.

This study is not a comprehensive discussion on all issues related to the natural environment; they are many and varied. Rather it focused on the loss of biodiversity, conservation and suggested strategies to address this decline. Conservation will not be favoured if:

1. there are no markets for the biological resource; and
2. governments intervene in the market and distort economic signals in favour of activities that involve biodiversity loss.

Conservation will be favoured if:

1. biodiversity demonstrates economic value; and
2. mechanisms are implemented whereby values can be appropriated and captured.

Wildlife can save itself. The experiences of CAMPFIRE, the Inuit, and crocodiles indicate that provided it is used wisely and marketed effectively and the rents appropriated to the land they came from and therefore to the owners wildlife will be conserved. This study has also argued and presented examples showing that common property as an institution can be efficient in allocating resources for conservation.

This study has also shown that change is possible, that the new strategies can be effective, efficient and compatible. Economic analysis is important to these issues and has added valuable insights into how to implement and sustain these new initiatives. By changing the behaviour of individuals such as assigning them property rights to wildlife it alters the owners perception towards conserving biodiversity. These strategies must however involve and integrate social, economic and ecological factors. Partnerships in conservation especially where local communities have property rights to wildlife and are allowed to exchange these rights in commercial markets can generate conservation and economic development benefits.

Appendix 1

RESPONSES TO THE QUESTION, "ARE YOU AWARE OF ANY POACHING FOR IVORY CURRENTLY GOING ON IN YOUR COUNTRY?"

	Yes	No	Don't Know
East	**5**	**0**	**0**
Ethiopia	X		
Kenya	X		
Sudan	X		
Tanzania	X		
Uganda	X		
West	**7**	**0**	**1**
Burkina Faso	X		
Ivory Coast	X		
Liberia			X
Niger	X		
Nigeria	X		
Senegal	X		
Sierra Leone	X		
Togo	X		
Central	**5**	**0**	**1**
Cameroon	X		
C.A.R.	X		
Congo	X		
Gabon	X		
Rwanda			X
Zaire	X		
Southern	**8**	**0**	**0**
Angola	X		
Botswana	X		
Malawi	X		
Mozambique	X		
Namibia	X		
South Africa	X		
Zambia	X		
Zimbabwe	X		
Overall Total	**25**	**0**	**2**

Source: Dublin *et al.*, 1995, 85.

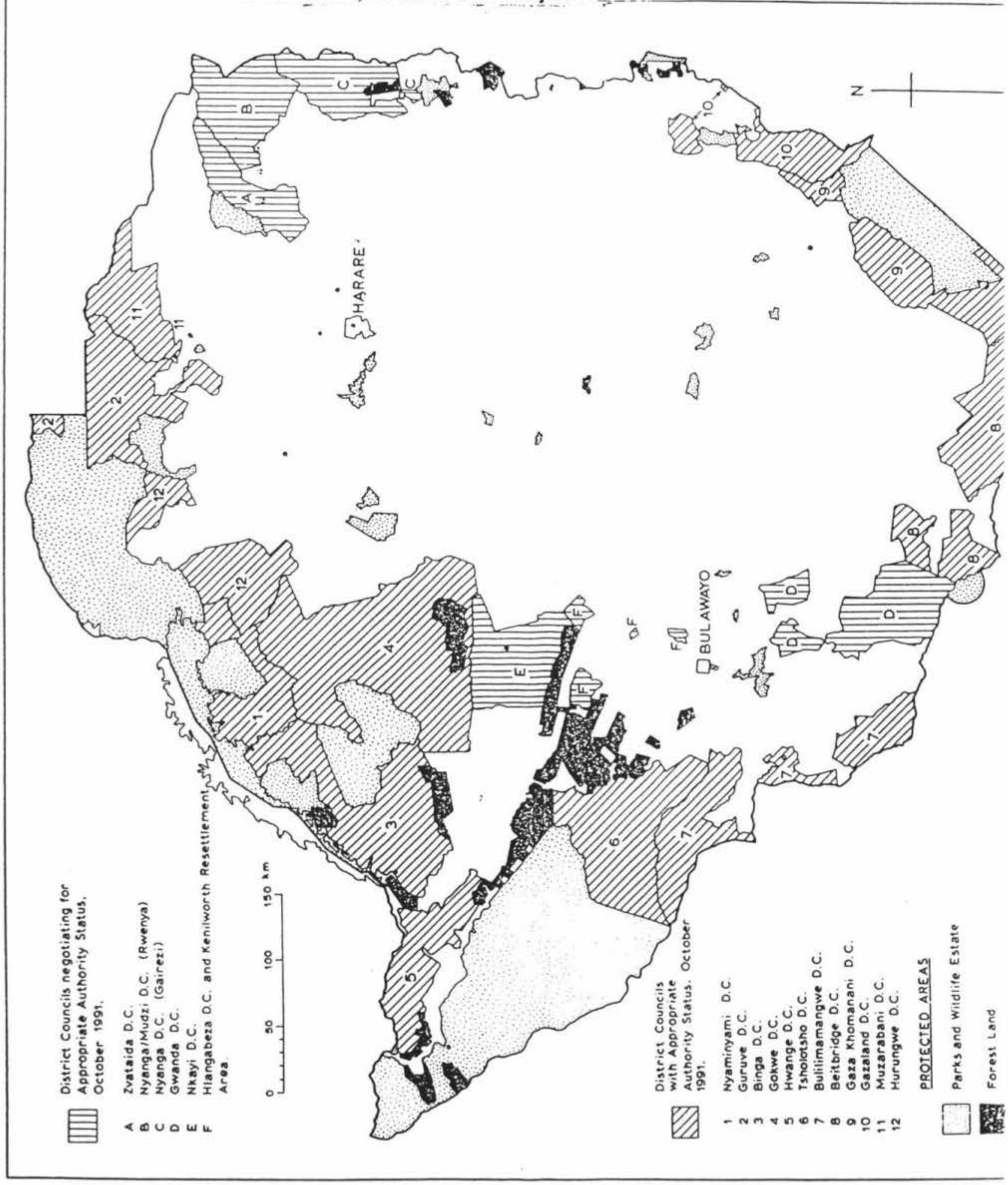
Appendix 2

WHEN CITES WORKS	WHEN IT DOES NOT
1. CITES ENVIRONMENT	
(a) When applied under centralised wildlife management systems.	When applied to Parties who have decentralised wildlife management systems.
(b) When there is a strong State capacity for species protection and control of illegal trade.	When Parties have limited Government capacity for law enforcement.
(c) When CITES takes into account the full social and economic effects which its decisions may have on stakeholders.	When CITES provisions are applied in a vacuum without consideration for the effects on stakeholders.
2. FUNDAMENTAL POLICY ISSUES	
(a) When CITES is seen primarily as an extension of Parties national conservation efforts.	When the treaty is seen by some Parties as a mechanism to force other Parties to comply.
(b) When emphasis is on incentives to promote sustainable use and improve wildlife management.	Where the emphasis is on banning trade without improving management.
(c) When sustainable harvests from the wild are preferred to captive rearing.	Where farming and captive breeding are preferred over wild harvests (including ranching).
(d) Where the Convention limits its considerations to species genuinely threatened by trade.	When the Convention is seen as a catch-all preservation treaty aimed at preventing trade.
(e) Where stricter domestic measures are applied primarily by range states to limit exports.	Where stricter domestic measures are applied primarily by he importing states.
3. IMPLEMENTATION	
(a) When CoP meetings are primarily technical in their functioning.	When the CoP is primarily political in its approach to the business of the treaty.
(b) Where perceived threats to a species do not automatically result in the banning of trade.	Where the inflexibility of the treaty results automatically in a trade ban.
(c) Where adaptive management is the chosen mode of implementation.	Where there is insistence on a *priori* demonstration of sustainability.
(d) When illegal trade is reduced.	When illegal trade thrives despite CITES.
(e) When CITES promotes a high value for specimens in legal trade.	When CITES focuses on destroying markets.

Source: Martin, 1997b, 4.

Appendix 3

Zimbabwe, CAMPFIRE Projects



Source: Ceballos-Lascurain, 1996, 250.

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