UN GLOBAL BIODIVERSITY ASSESSMENT – "UNSUSTAINABLES"

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What is "Unsustainable"?

By Freedom Advocates Staff

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The Global Biodiversity Assessment report directed by the United Nations Environment Programme (UNEP) calls for urgent action to reverse the effects of unsustainable human activities on global biodiversity, including but not limited to the following (PAGE REFERENCES from: Heywood, V.H. (ed). The Global Biodiversity Assessment. United Nations Environment Programme. Cambridge University Press: Cambridge, 1995. pp. xi + 1140):

337 Ski runs

350 Grazing of livestock: cows, sheep, goats, horses

350 Large hoofed animals: compaction of soil, reducing filtration

351 Disturbance of the soil surface

351 Fencing of pastures or paddocks

351 Road and trail construction

728 Fossil fuels - used for powering various kinds of machines

728 Agriculture

728 Modern farm production systems

728 Chemical Fertilizers

728 Herbicides

728 Building Materials

730 Industrial activities

730 Human-made "caves" [i.e., buildings, structures] of brick and mortar, concrete and steel

730 Paved and tarred roads, highways, rails

730 Railroads

730 Floor and wall tiles

733 Aquaculture

733 Technology improvements

733 Farmlands, rangelands

733 Pastures, rangelands

733 Fish Ponds

733 Plantations

738 Modern hunting

738 Harvesting of timber

749 Logging activities

755 Dams & reservoirs; straightening of rivers

757 Power-line construction

763 Economic systems that fail to set a proper value on the environment

763 "Inappropriate" social structures

763 Weaknesses in legal and institutional Systems

766 Modern attitudes toward nature - Judaeo-Christian-Islamic religions

767 Private property

771 Population growth - human population density

773 Consumerism

774 Fragmentation of habitat - cemeteries, derelict lands, rubbish tips [i.e., trash dumps], etc.

774 Sewers, drainage systems, pipelines

782 Private property

783 Land use that serves human needs

838 Modern attitudes toward nature - Judaeo-Christian-Islamic religions

969 Fisheries

970 Golf courses

970 Scuba diving

728 Synthetic drugs

990 Fragmentation of habitat through agricultural development, forestry, or urbanization, with resulting impervious surfaces; fragmentation through construction of roads, railroads, powerlines, and pipelines.

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The above list is comprised of summary excerpts from the hard copy edition of the UNEP Global Biodiversity Assessment report. For more internet information, one can view the UN Global Biodiversity link: http://earthwatch.unep.ch/biodiversity/assessment.php. However, the page numbers referred to above will not match the internet version.

Additional information:

IPCC Chair says Western Lifestyles are Unsustainable

http://www.guardian.co.uk/environment/2009/nov/29/rajendra-pachauri-climate-warning-copenhagen

Global Biodiversity Assessment

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masking snow and reducing albedo. This would act as a positive feedback to regional climatic warming, an effect that would be most pronounced at high latitudes, but could extend to the tropics (Bonan *et al.* 1992). The current net CO_2 efflux observed in Arctic ecosystems may depend directly on soil drying (Oechel *et al.* 1993) in the wet Arctic of North America (with diversity playing little role), but the increased CO_2 efflux in the drier Russian Arctic could reflect a reduction in cover of pollutant-sensitive mosses, whose insulative properties govern the soil temperature regime (Zimov *et al.* 1993b). Arctic wetlands and associated loess sediments are large terrestrial sources of methane (Reeburgh and Whalen 1992; Fukuda 1994), and changes in landscape diversity, as a result of soil drying, could reduce methane efflux. Changes in the abundance or species composition of sedges, which transport most methane from Arctic soils to the atmosphere (Torn and Chapin 1993), could alter fluxes of this greenhouse gas and, therefore, the role of methane in atmospheric warming (Whalen and Reeburgh 1992).

6.1.1.6 Landscape and waterscape structure

Human impacts on biodiversity. Human agricultural (alpine) and industrial (arctic) developments have substantially altered landscape structure and diversity. For example, in the Arctic, the building of roads and pipelines has altered patterns of water drainage, and the relative abundance of waterlogged and well-drained soils. In the alpine regions, the construction of ski runs has smoothed the landscape and introduced new plant communities. Such land-use change can alter hydroelectric yield. These indirect impacts of energy development are many times larger than the direct impact of development (Walker *et al.* 1987).

Ecosystem consequences of impacts. For ecosystem consequences, see the above sections on soil structure and water distribution.

6.1.1.7 Biotic linkages and species interactions

Human impacts on biodiversity. Increasing demand by non-Arctic people for Arctic animal products, combined with increasing hunting efficiency, has resulted in a greater human harvest of marine and terrestrial mammals, in many cases causing or contributing to population declines. This is often combined with changes in human social structure which might otherwise have placed limits on the exploitation of these animal resources (Young and Chapin 1995). In addition, human-induced climatic warming is altering the competitive balance and diversity of plant species within the Arctic (see productive capacity) and could decouple the phenology of plants and their pollinators, leading to elimination of plant species that may have important ecosystem effects (Inouye and McGuire 1991).

Ecosystem consequences of impacts. In cases where animals (e.g. sea otters) are keystone predators, over-hunting produces effects that propagate through the entire ecosystem. Human hunting of the Pleistocene megafauna may have triggered the change from grass-dominated steppe to less productive moss-dominated tundra at the end of the Pleistocene (Zimov *et al.* 1995). Changes in abundance of reindeer or herding practices in Russia and Scandinavia greatly influence lichen cover and, therefore, the vegetation structure and productivity of these landscapes (Andreev 1978). Geese and other waterfowl determine productivity, nitrogen input and cycling rates, and disturbance regimes, in Arctic salt marshes (Jefferies and Bryant 1995), and recent changes in the abundance of geese have totally altered the structure and dynamics of these coastal ecosystems. Little is known about the ecosystem impacts of possible changes in pollinator abundances. Insect-pollinated species are concentrated in areas of vertical relief, which contribute little to carbon storage or methane flux but are important in slope stability.

6.1.1.8 Microbial activities

Human impacts on biodiversity. Human impacts on microbial activity are mediated primarily by changes in species composition and litter quality (see productive capacity) and secondarily by the introduction of contaminants from oil spills and pollution.

Ecosystem consequences of impacts. Most Arctic and alpine ecosystems have a similar spectrum of enzymatic potentials to degrade common substrates such as lignin, cellulose and proteins, despite large differences in litter chemical composition (Schimel 1995). Ecosystems do differ, however, in their capacity to produce or consume methane, petroleum products and many anthropogenic pollutants. Changes in microbial diversity are thus more likely to be important in the production and degradation of unusual substrates than in the normal processing of plant litter and soil organic matter.

6.1.1.9 Summary and relevance to human activities

Arctic and alpine ecosystems are particularly vulnerable to human impacts on species diversity, because there are few species in the most widespread vegetation types, so the loss or gain of even one or two species has a large proportional impact on diversity. Furthermore, landscape diversity is easily altered by human impact, due to the sensitivity to disturbance of steep alpine slopes and the sensitivity of Arctic soils to permafrost degradation. Resulting ecosystem changes affect local inhabitants primarily by reducing the productivity of various animal species (reindeer, marine mammals, fish) on which they depend. In alpine and down-slope ecosystems, these changes influence run-off, landslide danger and the quality of drinking water. Other effects of human-induced changes on Arctic and alpine ecosystems are indirect, resulting from potential positive

Schwartz 1991). Biodiversity, reflected in species richness, is moderately high in semi-arid regions and declines with increasing aridity for most taxa (Shmida 1985; Pianka and Schall 1981; O'Brien 1993). Certain taxa are diverse relative to other biomes (e.g. predatory arthropods, ants and termites, grasshoppers, snakes and lizards, rodents, annual plants), but there is substantial variation in the richness of particular taxa among the deserts of different continental areas. The abundance and activity of desert organisms are 'pulsed' in correspondence with episodes of high moisture availability (Noy-Meir 1973; Louw and Seely 1982); while the prevalence of dormancy, cryptobiosis, aestivation and other modes of escaping harsh conditions means that most of the biodiversity of arid regions can be impossible to census or sample during most time periods.

6.1.4.2 Productive capacity, biomass, decomposition and nutrient cycling

Human impacts on biodiversity. Human activity has caused changes in the biodiversity of arid lands primarily through the use of arid and semi-arid systems for grazing of livestock. Introductions of domestic and game animals have altered the character and the magnitude of animal consumption (Oesterheld *et al.* 1992), and these have been accompanied by creation of water points, introduction of non-native plants, and removal of predators and of burrowing and herbivorous animals seen as threats to livestock. Plant species diversity decreases when the local extinction of grazing- or trampling-sensitive species exceeds the establishment of new grazing-tolerant or weedy species (Westoby *et al.* 1989; Milchunas and Lauenroth 1993); after a time lag, loss of native plant species may result in a loss of animal taxa (Jones 1981; Jepson-Innes and Bock 1989; Heske and Campbell 1991). In some regions the intensive cultivation of irrigated croplands has eliminated large portions of native ecosystems (Jackson *et al.* 1991).

Ecosystem consequences of impacts. In North America and Africa, semi-arid grasslands have often been converted to shrublands by grazing and by the dispersal of shrub seeds by livestock (e.g. Peinetti *et al.* 1993), leading to a different structure and display of biomass as well as altered species composition (Schlesinger *et al.* 1990). Production will be reduced if grazers remove leaf area and cause an increase in the proportion of water lost to evaporation, rather than being used by plants. However, net primary productivity or NPP (the amount of plant material produced by photosynthesis per unit area over a time period) is not necessarily changed by changes in plant species composition induced by grazing (Milchunas and Lauenroth 1993). Decreases in NPP are most pronounced where there has been no long evolutionary history of intense grazing (Milchunas and Lauenroth 1993). Some semi-arid and arid ecosystems comprise diverse assemblages of different plant growth forms, physiologies and life histories, which form distinct guilds with respect to water use because of correlations among morphological, phenological and physiological traits (e.g. Golluscio and Sala 1993). Where the members of a guild (e.g. perennial grasses using shallow water during the hot season) respond similarly to a disturbance (e.g. are all grazing-sensitive), the elimination of that functional group will have direct influences on the structure and functioning of the ecosystem. In addition, the importance of macro-organisms in determining microbial populations (Gallardo and Schlesinger 1992) means that removal of plants or alteration of the distribution and abundance of plant roots will alter the pool and rates of activity of decomposers, thus altering rates of decomposition as well as of organic matter inputs. Termites and nest- or burrow-building mammals provide micro-environments that enhance decomposition and nutrient cycling. Their removal by humans has the potential to slow decomposition and increase the chances of loss or transport of nutrients from the surface (Whitford 1991).

Conversion of semi-arid or arid lands to agriculture (usually by irrigation) may increase local NPP values, but the effects on biodiversity are severe. Native plant communities are displaced, and are notoriously difficult to restore after cessation of cultivation (Jackson *et al.* 1991). In India the birds of semi-arid regions have proved especially likely to disappear following cultivation and fragmentation of native habitats (Daniels *et al.* 1990). It is not clear how human activities have altered the role that fire plays in semi-arid regions. In some places fires may have maintained semi-desert grassland, suggesting that grazing (causing the reduction of fuel loads) and fire suppression have contributed to the conversion of grasslands to shrublands. However, in other systems the perennial grasses appear to be more sensitive to fire than the woody plants (Wright 1980), and there is no clear guideline for the use of fire in maintaining grassland or manipulating species composition (Bock and Bock 1992).

6.1.4.3 Soil structure and nutrient pools

Human impacts on biodiversity. Introduction of hoofed livestock to regions lacking a recent evolutionary history of ungulate grazing (e.g. Australia, southwestern North America) has been the primary human effect on the biodiversity of semi-arid regions. This has been accompanied by reduction of populations of native burrowing herbivores (e.g. North American prairie dogs, Australian marsupials), thus reducing the soil-disturbing activities of these animals.

Ecosystem consequences of impacts. The introduction of large hoofed animals to regions previously lacking them has caused changes in the compaction of soil, reducing infiltration (Roundy *et al.* 1992), while also churning up dry surface soil and increasing its vulnerability to erosion.

Vegetation influences the 'roughness' of the surface, which in turn influences the movement and erosive power of wind and water (e.g. Abrahams *et al.* 1994); thus the activities of livestock (or humans) have direct influences on the rates of erosion of surface layers. Erosion and transport of surface soil particles will in turn influence the transport and loss of mineral nutrients from the site (Schlesinger *et al.* 1990). The soil-binding properties of plant roots are especially critical in dunes, where species that can stabilize sediments serve a critical role in determining ecosystem structure (e.g. Klopatek and Stock 1994). Harvesting or loss of these species may shift a stable substrate (offering habitat for plant and animal populations) to a much harsher migrating dune system. Disruption of a microbial 'crust' on the soil surface (e.g. by hooves) alters inputs of nitrogen by these N-fixers (see 6.1.4.8). Burrowing animals (termites, ants, rodents, marsupials) create and reinforce heterogeneity in soil structure and nutrients (Whitford 1993).

6.1.4.4 Water distribution, balance and quality

Human impacts on biodiversity. Human and livestock activity have altered plant species composition and vegetation cover in many regions, both by reducing native vegetation and by introducing non-native species, either deliberately or accidentally. Aquatic habitats (riparian zones, springs) have been especially vulnerable to invasions of non-native plants. Populations of burrowing organisms have also been reduced. Road construction and other activities have altered drainage patterns, with resulting changes in water distribution and hydrologic regimes: these changes then impact the distribution and behaviour of organisms.

Ecosystem consequences of impacts. Vegetation cover modulates the impact energy of raindrops, reducing the amount of sediment dislodged and transported during heavy storms (Wood *et al.* 1987; Rogers and Schumm 1991). Rooted plants provide root channels which in turn enhance deep percolation of water into the soil profile (e.g. Greene 1992), and the nature of the plant canopy influences the proportion of rainfall that is intercepted and that falls either as throughfall or stem-flow (West and Gifford 1976; Navar and Bryan 1990; Tromble 1987). Deep roots of shrubs, and transpirational losses, are a strong influence on soil water content in the lower parts of the soil profile and thus on the depth of carbonate deposition, leading potentially to alterations in the effective rooting depth of the soil (Schlesinger *et al.* 1987). Removal of shrubs would then be expected to have potentially strong effects on soil moisture content and other characteristics. On the other hand, Dugas and Mayeux (1991) and Carlson *et al.* (1990) found that increased herbaceous cover following shrub removal resulted in little net change in water distribution or in total evapotranspiration from dry rangeland sites over the short term. Introduction of phreatophytic (deep-rooted) plants, especially the genus *Tamarix*, has dramatically altered hydrology in some riparian systems (Blackburn *et al.* 1982), even leading to the elimination of surface water from previous spring sites due to its high evapotranspiration). Reduction in populations of termites (small burrowing animals) has a dramatic influence on infiltration and surface runoff (Elkins *et al.* 1986).

6.1.4.5 Feedbacks to atmospheric properties

Human impacts on biodiversity. At the global scale, human activity is expressed primarily through direct and indirect alterations of vegetation cover and disturbance of the soil surface.

Ecosystem consequences of impacts. Vegetation cover is negatively associated with albedo. Arid lands are significant determinants of the Earth's overall albedo (Otterman 1989) and thus of its global radiation balance. Arid lands are also significant contributors of dust, and reductions in vegetation cover caused by grazing or other human activity (e.g. roads) increase these contributions (Pewe 1981; Pye 1987). While it has been proposed that arid-zone termites contribute substantial amounts of methane to the atmosphere, recent work suggests that arid regions are actually a significant sink for methane (Striegl *et al.* 1992). Conversion of semi-arid grassland to shrubland or woodland may increase carbon storage, affecting the global carbon cycle (McPherson *et al.* 1993).

6.1.4.6 Landscape structure

Human impacts on biodiversity. Human alterations to semi-arid landscapes are generally to facilitate grazing by livestock: they include fencing of pastures or paddocks, creation of new water points for animals, and construction of roads or trails for transport. Human and livestock activity often creates gradients of disturbance or of alteration (a 'variegated' landscape), rather than the conspicuously patchy or fragmented nature of landscapes in other ecosystem types (McIntyre and Barrett 1992).

Ecosystem consequences of impacts. Creation of new watering points has perhaps increased local rates of NPP. Conversely, disruption of normal drainage patterns (e.g. by road construction or by diversion of water) alters the hydrology of intermittent streams and playas, and decreases the productivity of vegetation dependent on that water flow (Schlesinger and Jones 1984).

6.1.4.7 Biotic linkages and species interactions

Human impacts on biodiversity. Importation of non-native plants for improvement of forage or as weeds in some regions has reduced local plant diversity by replacing native species. Humans have attempted direct removals of plant species considered to be undesirable forage species (e.g. large-scale removals of native shrubs in southwestern US rangelands). Populations of some tree species in semitribal societies (Hickerson 1965). For example, *Arthasastra*, the Indian manual of statecraft composed some 2000 years ago, prescribed strict protection to wild elephant populations in forests at the boundaries of kingdoms (Kangle 1969).

Such conflicts can have a variety of effects on biodiversity. When a New Guinea highland tribe defeats another in a war, it does not immediately take over the territory of the vanquished. Instead, the winners cut down the fruit trees raised by them, perhaps to reduce the chances of the defeated people attempting to reclaim their territory (Rappaport 1984). As conflicts have intensified with the march of civilization, they have called for large investments in both defence and offence, putting greater demands on the natural world. Apart from these indirect demands, conflicts have directly inflicted much damage, for instance in the pursuit of the scorched-earth policy by warring armies. Defoliation in Vietnam, damage to marine life in the Kuwait war, and destruction of agricultural land in the Horn of Africa are recent examples of the impact of international conflicts on nature. Internal insurgencies also affect biodiversity, in part positively by reducing the pressures of commercial exploitation, in part negatively by allowing unregulated hunting.

This phase of human existence provides the first examples of intensive human alteration of and major destructive impact on the environment. It also provides the first examples of how environmental degradation became a main contributing factor to the collapse of vast societies.

11.1.5 The modern high-energy phase

The steadily expanding global trade in biological resources under increasingly centralized control of nation states and business corporations has been accompanied by an industrial revolution brought about by a series of technological advances. These advances have deployed large amounts of energy towards production, processing, transport and distribution of increasing quantities of goods. Until a few centuries ago, humans for the most part depended on their own muscle power, and that of their domesticated livestock, for production. Thus horses and oxen ploughed the land and pulled the carts, and artisans depended on their own muscle power to weave the cloth, forge swords and ploughshares and build houses and bridges. Wood fuel fired kitchen hearths and metal foundries, and wind power moved ships over oceans, but these energy sources could only be used at very moderate levels of efficiency.

Enormous supplemental sources of energy came about with the development of the steam engine, the internal combustion engine, electricity turbine, nuclear fission and solar cells. An associated development is the ability to synthesize a variety of chemical molecules serving a wide range of functions (i.e. building materials, fertilizers, pesticides, synthetic drugs and plastics). All of this is contributing to even more profound changes in global biodiversity.

This modern high-energy phase is the dominant ecological and economical phase in the modern world today. It is characterized by very high and increasing rates of use of material resources and of energy (mainly in the form of fossil fuels) which is used for driving various kinds of machines. The basic ecological characteristics of farming remained unchanged until the nineteenth and twentieth centuries, but the industrial transition unleashed a new era in farming, with fundamental changes in agricultural practice. First, the use of machines powered by fossil fuels began to perform work previously done by humans or draft animals (e.g. ploughing, sowing and harvesting). Second, application of artificial fertilizers (mainly nitrogenous, phosphate and potash fertilizers) has progressively increased. And third, the genetic variability of cultivated plants has gradually declined.

Historically, as agriculture intensifies there is a tendency to emphasize a smaller number of crops (Boserup 1965). Thus in Asia the less intensive systems of cultivation involve several species and landraces of millets, amaranths and beans along with hill rice. With the introduction of wet paddies this gives way to stands of a few high-yielding varieties of rice and blackgram (*Phaseolus mungo*). This is because the technologies of more intensive crop production based on creation of relatively homogeneous environments tend to diffuse over wide areas.

Modern systems of agricultural production create extensive stands of single crops of low genetic diversity in the fields, while maintaining large amounts of genetic diversity so essential for continued success of plant breeding in *ex situ* storage. The modern farm production systems focusing on maximizing profits do so by wiping out environmental heterogeneity and by bringing in irrigation water and more recently chemical fertilizers, herbicides and pesticides. In these highly modified environments production tends to concentrate on genetically homogeneous stocks of a small number of species that may be grown most profitably in any given locality. These processes have led to a gradual loss of variety and genetic heterogeneity of cultivated plants being grown in any given locality, a process that is continually gathering pace.

The widespread use of synthetic pesticides to control parasites and diseases, and the cultivation of new high-yielding varieties of certain crops, are other changes that have resulted in considerable increases in yield per unit area, at the cost of losses in biodiversity.

The great increase in yield per hour of human labour is perhaps a more striking change. In the hunter–gatherer and farming phases, typically enough food was harvested to support each family. In the high-energy phase the situation

For humankind the possibility of resource scarcity is far less dangerous than that of 'choking in its own wastes', or severely damaging the productive processes of the biosphere. Broadly speaking, two kinds of waste are produced by high-energy societies. First are what can be called the biometabolism products of society which are the organic waste products (i.e. sewage, unused food and organic fibre). Usually little of this type of waste is returned to the soil so natural nutrient cycles are disrupted. The second type of waste consists of those waste products resulting from the technometabolic aspects of society, including industrial activities, such as plastic production, the use of machines, and the by-products of extrasomatic energy use. As discussed previously in this volume, human activities are also resulting in the discharge into the atmosphere of gases that contribute to global warming. The chlorofluorocarbons (CFCs), the synthetic products of our industrial society which are responsible for the thinning of the ozone layer are, volume for volume, many thousands of times more potent as greenhouse gases than carbon dioxide.

Needless to say, the control and use of material resources, and the material benefits derived from them, are far from evenly distributed within the populations of most high-energy societies. The disparities that exist in the intensities of resource use between populations in the developed and developing regions of the world are even more striking. The industrially developed world, which contains about one-fifth of the total human population, uses 90% of the non-renewable resources produced, meaning that on a per capita basis the populations in these countries are using about twenty times more non-renewable resources than populations in the developing countries. For example, the United States, which has 5% of the world's population, uses 27% of the materials extracted. Per capita, this is 36 times more than in the developing world. It is not easy to provide an accurate picture of the changing rates of use of different mineral resources either by humankind as a whole or by separate societies. However, it stands to reason that increasing industrial productivity involves increasing use of resources. The impact of this on biodiversity is unclear, but it seems inevitable that expanding human consumption of resources that means fewer resources will be available for other species.

In addition, in this phase, urbanization processes have increased, with a higher degree of artificialization. Simple shelters that once involved wooden poles and hides, or just shallow caves, have developed into enormous human-made caves of brick and mortar, concrete and steel. Simple beaten footpaths have been converted into kilometre after kilometre of paved and tarred roads. Such habitats especially favour a few commensals of humans; rock doves and swifts fond of rocky crags are at home in a forest of human-made crags as are black rats and house mice in human-made burrows. Cities have their own biodiversity-rich areas in gardens and parks where high levels of diversity may be created by artificially collecting species from around the Earth, often in environments specially created to suit them (Section 13.3.8.1).

Apart from directly moulding large areas, urban concentrations, highways and railroads affect natural diversity through making ever-growing demands on resources of far-flung areas. Pink granite is mined in south India, destroying extensive dry deciduous forests on its hill tracts, to make tiles for the bathrooms of the urban middle class and facings for buildings in western Europe. The fuelwood demand for baking bricks to support the unceasing expansion of Calcutta is leading to the slow demise of the world's largest mangrove forest in Sundarbans.

Urban metropolises may appear to be burgeoning, crowded, overstressed habitats with little room for nature. But hidden within the concrete maze, a surprising amount of wildlife still survives (Section 13.3.8.1) A small lake in the centre of Hanoi in Vietnam harbours the only surviving population of the giant freshwater turtle (*Pelochelys bibronii*) (Quy 1995). Another outstanding example of such survival is the Delhi Ridge Forest which runs through the capital city, and is covered by dense dry deciduous forest and thorn scrub. This 7770 hectare sliver of vegetation harbours a large amount of biodiversity, including nearly 200 species of birds.

11.1.6 History of conservation traditions

The consequences of advances in human technology through the ages, for the life experience of humans and for the relationships between human society and the biosphere, have been multiple and wide-ranging.

The preceding discussion supports the conclusion that the Earth has witnessed an ever-accelerating erosion of biological diversity over the past 10 000 years as numbers of people and their abilities to affect the natural world have grown steadily. But people can and do modify their behaviour on the basis of their perceptions of the long-range consequences of their actions. At many different times and in many different cultures, people have apparently been motivated to conserve, even enhance, biological diversity and have evolved a variety of cultural practices that seem to contribute to this end. But establishing a connection between specific practices and conservation or enhancement of biological diversity is by no means a simple matter; for the overtly declared purpose of a practice which seems to help conserve biological diversity may in fact be quite different. Thus, in South Asia many sacred ponds have helped conserve the indigenous fish fauna. But people leave these ponds alone out of respect for some deities, not with an expressly declared purpose of conserving fish diversity. It is then quite possible that many practices that seem to promote conservation may have originated from different village community-based forest management was officially reinstated in a small number of villages, including Kallabbe. A survey in the 1980s revealed that the Kallabbe village forest was among the most biodiversity-rich tracts in the district (Chandran and Gadgil 1993).

However, not all colonies were the same. A few recognized the negative impacts of settlement and sought to correct them. On Mauritius, the French colonial government passed an ordinance in 1769 which stipulated that 25% of all land-holdings were to be kept as forest, particularly on steep mountain slopes, to prevent soil erosion; all denuded areas were to be reforested; and all forests within 200 metres of water were to be protected. In 1803, clearing of forest was forbidden higher than one-third of the way up a mountain side (Grove 1992).

In some parts of Europe, the community control of common property remained strong through the period of rampant deforestation of Europe. By the mid-nineteenth century Switzerland retained only about 4% of its land under forest cover, leading to serious problems of landslides and siltation. The communities then rolled back the tide of forest loss, and successfully built back the forest cover to the present 25% (WRI 1994). As of today these secondary forests remain under the control of powerful local community-based governments. But such examples of effective community control, whether of ecosystem people as in Kallabbe or of biosphere people as in Switzerland, are exceptions. Over much of the world, biodiversity-rich common lands and waters are largely under control of the state, and have often been permitted to lose much of their biodiversity.

11.2 The impact of human activity on biodiversity

11.2.1 Introduction

The current impacts of humans on biodiversity are both direct and indirect. The direct mechanisms include habitat loss and fragmentation, invasion by introduced species, the over-exploitation of living resources, pollution, domestication and selection, global climate change, local and industrial agriculture and forestry. But these are not the root of the problem. Biotic impoverishment is an almost inevitable consequence of the ways in which the human species has used and misused the environment in the course of its rise to dominance: the factors that have led to the expanding ecological niche of humans are indirect causes of the loss of biodiversity. Sections 3 and 8 have assessed the current status of biodiversity and showed that humans have been a major force in determining this status. Section 6 has assessed human impacts on ecosystem functioning in various biomes. Chapter 11.1 has assessed knowledge about the history of human impact on biodiversity. This chapter will assess the mechanisms of human impacts – both positive and negative, direct and indirect – on biodiversity.

The 'Global Biodiversity Strategy' (WRI, IUCN, UNEP 1992) identified both direct and indirect mechanisms that affect current levels of biodiversity, nearly all of which have significant human components. The direct mechanisms, following Soulé and Wilcox (1980), Diamond (1985) and Pimm and Gilpin (1989), include:

- exploitation of wild living resources;
- expansion of agriculture, forestry and aquaculture;

Key messages

- Humans have endeavoured, rather successfully, to acquire their growing biomass needs from intensifying the productivity of a small number of domesticated species. Populations of the favoured species have then reached high densities in limited areas manipulated as farmlands, pastures, fish ponds or plantations.
- Humans have been engaged over historical times in steadily improving technology and expanding the range of biological resources useful to people. This has inevitably been accompanied by a retreat of the natural world; and an erosion of biodiversity.
- As natural forests and fish stocks have declined, the historical trend is to devote greater effort to plantations and aquaculture. These replace large tracts of natural diverse ecosystems with species-poor systems supported by high levels of technological inputs; they also promote extensive use of pesticides and other poisonous substances resulting in more widespread negative impacts on biodiversity.
- As human technological capacities have increased, so have inequities within and between societies. The powerful social segments within nations have access to natural resources from wide catchments, suffering few of the negative consequences of environmental degradation and erosion of biodiversity. Rather, they have a strong vested interest in continued growth of the artificial at the cost of the natural, often in another country. This disrupted link between maintenance of biodiversity and the quality of life of those who ultimately decide the course of economic activity is at the base of the growing pace of erosion of global biodiversity (Shiva *et al.* 1991).

high harvest rates, however, removal of dead wood implies a reduction of food and substratum for large groups of species (e.g. Siitonen 1994). Moreover, picking of berries at extreme harvest rates may imply a change in selection pressure or reduction of recruitment rates for the species in question, as well as removing energy which would otherwise either be available to native fruit-eating birds, mammals and insects or form part of the nutrient cycle in the forest.

Hunting has exterminated many endemic species, particularly on islands (Olson 1989; Diamond 1989). However, hunting may also lead to the demise of species in less confined habitats (Caughley 1994), and circumstantial evidence indicates that prehistoric humans were instrumental in the disappearance of many large mammals in the Americas, the Mediterranean, Madagascar and Australia (Janzen and Martin 1982; Martin and Klein 1984; Owen-Smith 1987; Burney 1993). Although modern hunting based on scientifically developed management plans may actually increase population numbers, it is usually selective, implying increased mortality within sex or age groups, altered population structures, life histories and genetic structures and often the removal of some of the most fit males (Skogland 1990).

Harvesting of timber from a natural forest has profound effects on the biodiversity of the forest ecosystem (e.g. in tropical forests: Sayer and Whitmore 1991; Whitmore and Sayer 1992). Harvesting of timber reduces diversity in terms of tree species and structural variation (Järvinen *et al.* 1977; Kouki 1994), although young, early successional stands which develop soon after harvesting may have high biodiversity (Kimmins 1992). Forest trees compete with other plants, provide food for various animals, and dominate the physical structure and microclimatic conditions of the ecosystem, creating habitat and substratum for other species. In unharvested forests, old, dead and decaying trees are food and substratum to many late succession organisms (Whitmore1984; Maser, 1988; Väisänen *et al.* 1993). Harvesting for timber and fuelwood removes these old and dead trees, and may lead to the disappearance of a number of the specialist species (Raphael and White 1984; Angelstam and Mikusinski 1994; Siitonen 1994).

In addition to these effects, removal of biomass may also influence the nutrient cycling in ecosystems. Clear-cutting removes biomass from the ecosystem and changes its physical structure to an extent that alters microclimate, nutrient cycles and nutrient availability (Bormann *et al.*,1977; Saulei 1984; Malingreau and Tucker 1988), influencing regeneration processes and thereby species diversity in all subsequent successional stages. This aspect may be more pronounced in moist tropical forests than in temperate forests (Maser 1988), as tropical forests have a larger proportion of their biomass stored in the living vegetation (Whitmore 1984).

Harvest of biomass from natural ecosystems may also be indirect, through the grazing of domestic animals. It is well known that heavy grazing may seriously alter ecosystems in subarctic, temperate and tropical regions (e.g. Bond 1993). The effects of grazing depend on the species of grazing animals, for example domestic goats may destroy the plant roots, promoting soil erosion, whereas cattle only eat the above-ground parts of the plants (although, being heavy animals, they can cause soil erosion indirectly through soil compaction resulting in higher runoff).

While grazing may directly alter plant species composition and productivity, it may also indirectly affect invertebrate species which often play an important role in the maintenance of stability in ecosystems. Mott and Tothill (1994), reviewing the losses in diversity in Australian savannahs, document the large changes in abundance and diversity in ants, soil detritivores and termites as a result of grazing practices. Even at low stocking rates substantial changes in the fauna have been recorded: free-ranging, low-density livestock have been associated with a 60% reduction in small rodents and marsupials in the Central Australian region since pre-European times (Morton and Baynes 1985). A study in China showed that changes in plant communities through grazing by domestic animals directly affected the species composition of grasshoppers, with moderate grazing preserving more diverse grasshopper populations with a lower proportion of pest species (Le 1994). On the other hand, Dodd (1994) discusses the need for caution in explaining the vegetational changes in Africa's arid lands in terms of overgrazing of domestic livestock. While the high densities of animals near high-volume watering points and villages and on rangelands closely linked to crop agriculture in Africa can permanently and severely alter vegetation, he found no scientific evidence that either nomadic or commercial use of livestock has caused irreversible changes in range vegetation away from these areas.

Among wild ungulates, harvesting can affect population growth by influencing reproductive traits such as age at maturity, twinning rate, and proportion of reproducing females (Jonsson *et al.* 1993). These reproductive traits vary with body size and age of the animal, and show geographical variation reflecting differences in environmental conditions and adaptations. There is a negative correlation between age at sexual maturity among moose and life-time reproductive success; cows that mature earlier begin to produce twins sooner than cows that mature later and as a result have higher reproductive output.

Over-exploitation can severely disrupt ecological communities. While it is generally agreed that whaling represents one of the most dramatic alterations of mammalian species diversity by humans, the quantitative effect of whaling on deep-sea biodiversity is difficult to in the artificial propagation, the changed selection regime during rearing may change the genetic structure of the recipient population (Ryman and Ståhl 1980; Waples 1991), though the implications of this for biodiversity remain unclear.

11.2.2.2.5 Forestry. Global closed forest is estimated to have covered three million hectares in the early 1980s, roughly evenly split between temperate and tropical areas. In the most authoritative statement on forest status globally, FAO (1993) reports that between 1980 and 1990, an annual average of 15.4 million ha of tropical forests were cleared, amounting to an annual loss of about 0.8% and a total loss over the decade of tropical forests of almost three times the size of France. Alarming as these figures are, it is difficult to determine whether 1970–80 rates of loss were significantly different from the 1970s because FAO used different methodologies in its later assessments. FAO also reports that the area of tropical forest plantations increased from 18 million ha in 1980 to more than 40 million ha in 1990, with about three-quarters found in Asia.

Temperate closed forests are thought to be in a relatively stable state at present, or even showing a slight increase (WRI 1994), although Dudley (1992) warns that this general trend obscures some important regional variations and significant losses in old-growth forests.

Since tropical forests are thought to contain at least 50% of the species of the globe, it is hardly surprising that much attention has been focused on tropical deforestation rates and species extinction.

The documented evidence of species extinction through tropical forest loss is, however, limited and although much controversy surrounds the likely level of extinctions (see Section 4.3 and various papers in Whitmore and Sayer 1992) it is generally agreed that disturbance of forests will alter the relative frequency of animal and plant species and, where severe, disturbance will cause a 'commitment to extinction' although the time to reach the new equilibrium state is unknown (Heywood and Stuart 1992; Reid 1992).

Deforestation has three major effects: habitat loss; habitat fragmentation; and edge affects at the boundary zone between forested and deforested areas (Skole and Tucker 1993). The data presented by these authors on deforestation in the Brazilian Amazon emphasizes the relative importance of the edge effect which may extend 1 km into adjacent forest resulting in a net loss of 'interior' plant and animal species in the edge area which may be occupied by a different suite of species (Turner *et al.* 1991). Skole and Tucker's figures indicate that simply looking at deforested areas alone masks the broader effects: deforestation affected a total of 230 324 km^2 in their study areas, creating 16 228 km^2 of isolated forest and an edge effect covering 341 052 km^2, resulting in a total area affected by deforestation activities of 587 604 km^2.

Deforestation is a dramatic alteration of habitat but many other levels of degradation, not considered as deforestation, can lead to long-term and perhaps permanent changes in species composition. Logging for international and domestic consumption, although only one of the causes of forest destruction, has proved to be of great importance because it opens up the forest to further encroachment by agricultural settlers, in itself believed to be the single greatest cause of forest destruction (Poore and Sayer 1991; Callister 1992).

Logging activities have both direct and indirect effects on biodiversity. Loggers are usually selective in their choice of species and, although extinction of exploited timber species has not been documented, Oldfield (1988) gives examples of several island endemic species that are on the brink of extinction and lists 41 other timber species

***Table 11.2-5:** Causes of tree mortality during logging (from Johns 1992).*

	Percentage Loss of trees (>30 cm girth)				
	Ponta da Castanha, Brazil	**Nigeria**	**S. Tekam, Malaysia**	**S. Pagai, Malaysia**	**G. Tebu, Malaysia**
Killed					
Timber trees	0.6	1	3	8	10
Destroyed during construction of access roads and landing sites, etc.	60.4	25	8	46	55
Destroyed during felling and dragging			39		
Remaining	39	74	49	46	35

Table 11.2-11: Global status of land degradation in drylands of the world (Biswas 1994).

Continent	Irrigated lands			Rainfed croplands			Rangelands			Total agriculturally used drylands		
	Total 10^6 ha	Degraded		Total 10^6 ha	Degraded		Total 10^6 ha	Degraded		Total 10^6 ha	Degraded	
		10^6	%		10^6	%		10^6	%		10^6	%
Africa	10.42	1.90	18	79.82	48.86	61	1342.35	995.08	74	1432.59	1045.84	73.0
Asia	92.02	31.81	35	218.17	122.28	56	1571.24	1187.61	76	1881.43	1341.70	69.7
Australia	1.87	0.25	13	42.12	14.32	34	657.22	361.35	55	701.21	375.21	53.6
Europe	11.90	1.91	16	22.11	11.85	54	111.57	80.52	72	145.58	94.28	64.8
N. America	20.87	5.86	28	74.17	11.61	16	483.14	411.15	85	578.18	428.62	74.1
S. America	8.42	1.42	17	21.35	6.64	31	390.90	297.75	76	420.67	305.81	72.7
Total	145.50	43.15	30	457.74	215.56	47	4556.42	3333.46	73	5159.66	3592.17	69.0

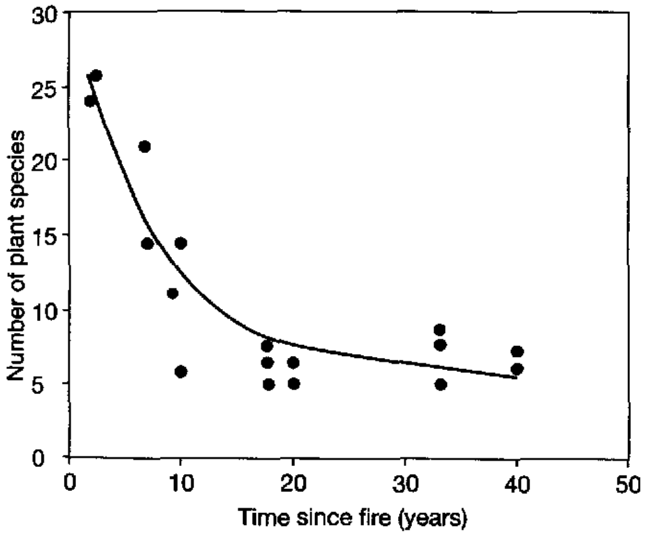

Figure 11.2-4: Numbers of plant species within a 100-metre radius plot with time since last fire in desert vegetation near Lake Mackay, Western Australia (*from* Burrows and Christensen 1990).

(Naiman 1992). Riverine systems have been profoundly influenced by damming, channelization, and other engineering works that reduce their diversity and dynamism. As of 1988, some 39 000 large dams had been constructed, creating reservoirs larger in total than the area of California or France (International Commission on Large Dams 1988); more than 100 dams with heights greater than 150 metres have been constructed, creating reservoirs that cover 600 000 km^2 (an area greater than the North Sea) and have a capacity of 6000 km^3 equivalent to 15% of the annual runoff of the world's rivers (Pierce 1992). The reservoirs created by dams therefore represent substantial new aquatic habitats, though these tend to be occupied by common and widespread species. The new reservoirs also tend to replace much more diverse habitats, including waterfalls, rapids and floodplain wetlands, leading to the loss of the numerous species of plants and animals specific to running waters (Dynesius and Nilsson 1994). As a result of habitat destruction and obstruction to the dispersal of fish and other organisms, many riverine species may have been extirpated over large areas and others have become fragmented and therefore more prone to future extinction.

While dam construction is the most obvious human intervention leading to the loss of wetland habitats, other engineering works also cause problems. For example, straightening rivers decreases the retention of matter and energy, and naturally functioning floodplains provide wildlife habitat and help reduce or buffer non-point-source pollution (Petts 1984).

Wetlands – including estuaries, mangroves, open coasts, floodplains, freshwaters (streams and rivers), lakes, peatlands, and swamps – have been lost due to a wide range of factors (Table 11.2-12).

The industrialized countries have had an especially profound impact on rivers. Dynesius and Nilsson (1994) found that 77% of the total water discharge of the 139 largest river systems in North America north of Mexico, Europe and the Republics of the former Soviet Union is strongly or moderately affected by fragmentation of the river channels by dams and by water regulation resulting from reservoir operation, inter-basin diversion and irrigation. Rivers are also being influenced through human activities in their catchments, which are being influenced by embankments, draining, deforestation, urbanization and industry (Turner *et al.* 1990). The remaining free-flowing large river systems are relatively small and nearly all situated in the far north, as are the 59 medium-sized river systems of Scandinavia. Modifications to riverine systems resilient and are unlikely to be adversely affected by climate change (Wilkinson and Buddemeier 1994), they are vulnerable to chronic stress from sources such as pollution. While data are sparse, Wilkinson (1993) concludes that 10% of the coral reefs of the world have already been degraded beyond recognition; 30% are in a critical state such that they will be lost or severely damaged within the next 10 to 20 years; another 30% will be grossly damaged in 20 to 40 years; and about 30% appear likely to remain healthy into the far distant future. The status of coral reefs is clearly responsive to human management action (Phongsuwan and Chansang 1993; Lidell and Ohlhorst 1993).

More generally, a decline in biodiversity has been associated with specific land-use changes associated with economic development, such as power line construction (Nickerson *et al.* 1989), urbanization (Leidy and Fiedler 1985), colonization adjacent to protected areas (Neumann and Machlis 1989), and forest fragmentation (Harris 1984; Hanson *et al.* 1990). However, economic development in urban and peri-urban habitats does not invariably mean loss of all of the original biodiversity and, in addition, new habitats such as urban parks, urban forests, urban wetlands, domestic gardens, roadside plantings, etc. may result in increases in both population densities (in certain species) and local biodiversity (Tables 11.2-13 and 11.2-14). Urban

Table 11.2-13: *Impact of urbanization on biodiversity: dependence of bird biomass and diversity on urbanization in Finland (Nvorteva 1971; Goudie 1993).*

	City (Helsinki)	Near rural houses	Uninhabited forest
Biomass (kg/km^2)	213	30	22
No. of birds (km^2)	1089	371	297
Number of species	21	80	54

Table 11.2-14: *Population estimates of birds in different habitats (from McClure 1969).*

Place	Habitat	Birds per 40 ha
Kuala Lumpur	Urban and gardens	1100
Subang	Secondary forest	450
Rantau Panjang	Cococut plantation and mangrove	800
Ulu Gombak Forest Reserve	Extraction track in logged forest	400
Ulu Gombak	Virgin jungle reserve	400

biodiversity is discussed in greater detail in 11.2.3.3 and Section 13.3.8.1.

In conclusion, habitat loss clearly has been a major proximate cause of the loss of biodiversity, though surprisingly little accurate information on habitat loss is available globally. Changes in the distribution and quality of most habitat types are difficult to determine, even with the availability of satellite imagery. The problem is especially difficult in developing countries due to problems such as inadequate ground-truthing, less comprehensive satellite coverage, and the difficulties in identifying and classifying regional habitat types (Sisk *et al.* 1994).

11.2.2.4 Indirect negative effects of species introduced by humans

11.2.2.4.1 Introduction. As early as 1958 Elton recognized the effect of invasive species as 'one of the great historical convulsions in the world's fauna and flora'. The history of invasions is of course older than the history of humankind, some mixing of species certainly occurring without human intervention, but this section is concerned only with human-induced invasions of species. Historical aspects of invasions were dealt with in Chapter 11.1.

Species are introduced into alien habitats by people for a number of reasons. Levin (1989) identified three major categories: (1) accidental introductions; (2) species imported for a limited purpose which then escape, and (3) deliberate introductions. Many of the introductions relate to the human interest in providing species that are especially helpful to people. This is particularly true of agricultural species; indeed, in most parts of the world, the great bulk of human dietary needs are met by species that have been introduced from elsewhere (Hoyt 1992). Species introductions in this sense, therefore, are an essential part of human welfare in virtually all parts of the world. Further, maintaining the health of these introduced species of undoubted benefit to humans may require the introduction of additional species for use in biological control programmes which import natural enemies of, for example, agricultural pests (Waage 1991).

As a biodiversity issue it is not always possible to identify invasions as inherently 'bad'; di Castri (1989) asserts that overall, the central European flora has undergone an enrichment of diversity over historical time as a result of human-induced plant invasions. Britain's mammalian fauna totalling about 49 species includes some 21 introduced species, including eight large mammals (wild goat *Capra hircus*, fallow deer *Dama dama*, Sika deer *Cervus nippon*, Indian muntjak *Muntiacus muntjak*, Chinese muntjak *Muntiacus reevesi*, Chinese water deer *Hydropetes inermis*, Bennett's wallaby *Macropus rufogriseus bennetti*, reindeer *Rangifer tarandus*). It is thus highly likely that, due to human influence, the mammalian fauna of Britain is more species-rich now than at any time sulphide in lake and marine sediments (Mannion 1992; Goudie 1993) (Box 11.2-6). Aquatic life is also threatened by organic, chlororganic and other micropollutants from polluted rain, pesticides from agriculture, drainage from dumps and fillings, industry outlets, airport drainage, etc. Marine environments are polluted by organic waste and chemicals (e.g. antibiotics) from fish farms and inorganic (e.g. from antifouling paint) and organic waste products from ships, runoff and land-based discharges and airborne pollution from distant sources (e.g. Atlas and Giam 1981; Anon. 1989; Muir *et al.* 1992), etc. Table 11.2-17 summarizes the relative contribution of all potential pollutants from human activities which enter the sea as estimated by GESAMP (1990).

Chemicals added to ecosystems by human action can have profound ecosystemic effects. Peterson *et al.* (1985), for example, report that a river in northern Canada was transformed from a heterotrophic condition to an autotrophic condition through the addition of phosphorus from air pollution. Higher nutrient input from agricultural runoff (and also from human wastes) increases the primary production of coastal waters (80–90% of nutrient input is taken up by primary production in estuarine and nearshore waters; GESAMP 1990) and has, for example, increased primary productivity in the Baltic Sea by 30% (Hammer *et al.* 1993). However, this has not been followed by an increase in decomposition and the net result has been the production of anoxic conditions in deeper waters, either impoverishing or completely eliminating benthic communities. These conditions have also substantially reduced the spawning area for cod – which require, for successful spawning, a minimum salinity only found in the deeper waters of the Baltic. But there have also been other repercussions of high nutrient runoff: the higher turbidity resulting from runoff has caused a reduction in the *Fucus* community in deeper waters and an increase in filamentous algae which have taken over their role. The balance of species within the Baltic community has changed, with fishes such as bream and roach increasing at the expense of whitefish, pike and perch (Hammer *et al.* 1993). The effects have therefore been far-ranging, affecting both biodiversity and the integrity of the ecosystem.

Table 11.2-17: *Relative contribution of all potential pollutants from human activities which enter the sea (GESAMP 1990).*

Source	Contribution %
Offshore production	1
Maritime transportation	12
Dumping	10
Runoff and land-based discharges	44
Atmosphere	33

11.2.2.6 Global climate change

In coming decades, a massive 'side-effect' of air pollution – global warming – could play havoc with the world's living organisms (Peters and Lovejoy 1992). Human-caused increases in 'greenhouse gases' in the atmosphere are likely to commit the planet to a global temperature rise of 1–3 °C during the next century, with an associated rise in sea level of 1–2 metres. Each 1°C rise in temperature will displace the limits of tolerance of land species some 125 km towards the poles, or 150 m vertically on mountains. Satellite altimeters are becoming sufficiently sophisticated to measure changes in sea level, indicating rates of sea level rise of 3.9 ± 0.8 mm per year, substantially more than had earlier been estimated (Nerem 1995). Many species will not be able to redistribute themselves fast enough to keep up with the projected changes, and considerable alterations in ecosystem structure and function are likely. In the United States rising seas in the next century may cover the entire habitat of at least 80 species already at risk of extinction. Many of the world's islands would be completely submerged by the more extreme projections of sea level rise – wiping out their fauna and flora, not to mention human habitations. And protected areas themselves will be placed under stress as environmental conditions deteriorate within them and suitable habitats for their species cannot be found in the disturbed land surrounding them. As the effects of climate change will be felt most profoundly in the future, this topic is covered more fully in Chapter 11.4.

11.2.3 Forces driving human impact on biodiversity

The root causes of the loss of biodiversity are not in the forest or on the savannah, but are embedded in the way societies use resources. They lie in human social organization, burgeoning human numbers, the way in which the human species has progressively broadened its ecological niche and appropriated ever more of the Earth's biological productivity, the excessive and unsustainable consumption of natural resources, a continuing reduction in the number of traded products from agriculture and fisheries, economic systems that fail to set a proper value on the environment, inappropriate social structures, and weaknesses in legal and institutional systems (WRI, IUCN and UNEP 1992). These are discussed in more detail below.

11.2.3.1 The rules governing the use of biological resources

11.2.3.1.1 Introduction. The way human beings view, use and maintain elements of biodiversity is greatly dependent

Clearly, some societies adapt to changing conditions better than others (Davis 1977; Laughlin and Brady 1978; Fürer-Haimendorf 1982; Eder 1987). Kuran (1988) has criticized the several theories seeking to account for such variability, which generally ascribe lack of adaptation to personal conservatism or collective conservatism – both forms of attachment to past choices. The key factor in successful adaptation appears to be the presence of feedback mechanisms which allow consequences of decisions to influence the next set of decisions, which enable societies to adapt to changing conditions.

When local people are part of a local ecosystem, their behaviour directly affects their own survival. But cultural mechanisms that have been developed as adaptations to the environment over tens or hundreds of generations are quickly cast aside when trade or new technology frees people from traditional ecological constraints, changing them from what Dasmann (1975) calls 'ecosystem people' who are adapted to their local ecosystem, into 'biosphere people' who can draw from the resources of the entire world.

While changes in traditional attitudes towards nature can be a result of internal dynamics (e.g. population increase), it is perhaps more often a result of outside influences: interaction with a modern culture, intrusion of the market, and so on. Redford (1990) has argued that in South America, at least, traditional resource use patterns may be sustainable only under conditions of low population density, abundant land, simple technology and limited involvement with a market economy, so that when confronted with market pressures, higher population densities, new technologies, and increased opportunities, few indigenous peoples can maintain the integrity of their traditional methods. This argument also applies to traditional fisheries management systems in the insular Pacific (Johannes 1978). In tropical Asia, on the other hand, some traditional systems have supported high populations and intensive agriculture for centuries. However, at least some such groups have fought hard to maintain their identity, and are seeking expanded international support for their efforts.

One of the lessons from recent work in ethnobiology and traditional ecological knowledge is that some indigenous views of conservation may differ from conventional biological views but nevertheless are legitimate in their own right (Berkes 1987; Alcorn 1993; Gadgil *et al.* 1993). One of the main differences between these two sets of views is that traditional concepts of conservation tend to be user-orientated; there have been no documented cases of a 'traditional preservation ethic'. Yet many indigenous groups have practices that help maintain ecological processes and the species that mediate these processes (Alcorn 1989). Thus, the area of common interest between Western conservation scientists and indigenous wise-use conservationists is biological sustainability. A major paradigm change among some Western conservationists is that some kinds of human use are accepted as part of conservation planning, as done in the updated World Conservation Strategy (IUCN/UNEP/WWF 1991). According to this view, biodiversity cannot be conserved in the longrun without the support of indigenous and other rural peoples, and without attention to their views and needs (e.g. Gadgil *et al.* 1993). As Alcorn (1993) puts it, indigenous peoples' goals match the broader goals espoused by many conservationists who recognize that most of the world's biodiversity is found in landscapes already occupied by people.

The influence of modern attitudes towards nature is now global. Stressing the place of humans outside nature, and the need for and possibility of technology mastering nature, these attitudes tend to treat all biodiversity elements as material resources created for human use. Therefore, nothing is 'wrong' in the extinguishing of other life-forms if the pursuit of material well-being (called 'progress') is served by it. Also lost in much of modern thinking are notions of inter-generational equity, as the discount rates used by economists render long-term considerations non-viable (Daly and Cobb 1989). Such modern materialist views have directly or indirectly led to considerable over-exploitation of nature, and consequently to the loss of biodiversity (WRI, IUCN and UNEP 1992). However, it can be argued that the modern scientific world view is also essential in using resources sustainably, since this provides a better capacity to understand natural processes and products, and to use them rationally. Rolston (1993) believes that spirituality and science need not be at odds. The biological sciences describe what is the case in nature and enable us better to appreciate and conserve it. Ecologists, for example, often solve problems with numerous variables and make predictions based on those solutions (The Crucible Group 1994). Perhaps science does this better than folklore, mythology and religious beliefs. Science observes ecological reality in the world, and does not choose or assign this reality (though of course scientific tradition affects the way scientists perceive reality; Sprugal 1991). People may choose to conserve natural things because they are useful, but also because they marvel at the intricacy, diversity, complexity, beauty, order, natural history and creativity present in nature. Ethics is informed by the facts about nature, and these facts influence value judgements.

In a way, the human world-view of nature is coming full circle in parts of the modern conservation movement. Respect for nature, inter-generational equity, and other such values which characterized many traditional societies are again gaining ground, not just in the deep ecology (Wilson 1984) and animal rights groups (Regan 1983; Singer 1991), but also in more mainstream efforts such as the UN General Assembly's *World Charter for Nature* (1982), the WWF's Network in Conservation and Religion, the IUCN's Working Group on Ethics, Culture, and Conservation, the World Commission on Environment and Development's *Our Common Future* (WCED 1987), the document *Caring for the Earth: A Strategy for Sustainable Living* (IUCN, WWF, UNEP 1991), the *Earth Charter* adapted by the UN Conference on Environment and Development (1992), and the Australian CSIRO Conference on *Nature Conservation: The Role of Networks* (in press). This is further explored in Section 13.

There is in all this an increasing realization that cultural and biological diversity are intimately and inextricably linked. Attitudes toward nature and toward fellow human beings are part of a society's culture, and the enormous diversity of cultures around the world has arisen in response to diverse biological, historical and physical environments, and has in turn influenced the management of these environments. As cultural homogenization sweeps across the world in the wake of modernization and westernization, the vast range of human knowledge, skills, beliefs and responses to biodiversity is also washed away, leading to great impoverishment in the fund of human intellectual resources. Loss of cultural diversity leads to loss of biological diversity by diminishing the variety of approaches to human, plant and animal coexistence that have been successful in the past, and by reducing the possibility of imaginative new approaches being developed in the future (Burger 1990; Mayberry-Lewis 1992; Suzuki and Knudtson 1992).

11.2.3.1.3 Property rights and the use of biological resources. Human societies display an enormous range not only of moral and other attitudes, but also of organizational forms, interpersonal and inter-community relations, and political and economic systems. Each of these systems tends to have a somewhat different impact on the way humans relate to and use resources, and therefore on biodiversity. While no comprehensive comparative work on such impacts is available, the following provides examples indicative of the complexity of the situation.

The object of the property relationship is often a biological resource: land, trees, wildlife, crops and so on. The property relationship may then be based around obtaining, guaranteeing, or controlling access to the resource, essentially securing the rights to deriving value from it by either use or exchange. Property rights are not absolute and unchanging, but rather a complex, dynamic and shifting relationship between two or more parties, over space and time. Numerous property regimes have been distinguished, including open (unregulated) access, communal (regulated) property, private property (including corporate property), and state property (Berkes and Farvar 1989). How each of these relates to the conservation and use of biodiversity and the impacts of changing from one to another regime, appears to be highly variable (Berkes 1989).

In considering the property regimes that affect access to resources in most parts of the world, the complications are many. For instance, more than one set of laws or legal systems may be in operation at any one time and place, affecting the same common property resource. The dynamics of land use and transformation at a particular site are significantly influenced by competition between different sources of legitimate authority, such as 'customary' or 'traditional' law and 'formal' legal systems (Moore 1986; Fortman 1990; Peluso 1992). Further complicating the matter, conflicting perceptions and negotiations over the meaning of land and other natural resources may affect the form of property relations (Dove 1986; Posey 1989). In addition, scholars have pointed to the importance of an individual's social position, derived in part from participation in various types of social networks, as central to an appreciation of the ways in which property relationships are formed and operate in any particular place (Berry 1989, 1991).

Painter (1988), for example, found that the crucial issue underlying the loss of biodiversity in Latin America was the gross inequity in access to resources. In eastern Bolivia, smallholders are locked in an intense struggle for development resources (such as access to markets, agricultural credit, transport facilities and similar services) with large-scale commercial agriculture enterprises, lumber companies and other interests. This competition appears as an inter-ethnic conflict between people who are native to the area and migrants from Bolivia's highland and Andean valley regions.

Following up on this point, Lele (1991) has made a convincing argument that differential access to resources and the resulting affluence for some, in the form of over-consumption, may be linked much more directly to environmental degradation than is poverty *per se*, in either the North or the South. It is much easier for an economically powerful country to gain access to the resources of less powerful countries by creating an economically powerful market demand. This can then encourage the countries who are hoping to tap into this market to harvest their resources at unsustainable rates, while the richer country can protect its resources for a later day. Although it does not necessarily make environmental sense to do so, many tropical countries have taken an economically defendable position of harvesting their forest as quickly as possible, thereby cashing in on the stumpage value of the standing timber and investing the capital in opportunities that earn a higher rate of return, at least in the short term. This tendency explains the boom-and-bust logging cycles that have characterized many small tropical countries. Timber concession policies are partly to blame, because they fail to combine forest tenure with capture of stumpage value, thus preventing either the government or concessionaires from detecting and responding to rising possibility of conservation-orientated use of resources would be greater with decentralized political structures, everything else being equal. However, little literature is available to support this hypothesis. Nor is it possible to prove unequivocally, at the current state of knowledge, that democracies are better at conserving biodiversity than dictatorships: experience from Africa, for instance, suggests that either may be possible. However, given that democracies are more amenable to other aspects of social relations which may facilitate conservation (e.g. access to information, secure land tenure and resource rights, participation in decision-making, flexibility in dealing with diverse situations, etc.), such a relationship may be posited. Reed (1992) shows how the mismanagement of natural resources in Côte d'Ivoire, leading to severe deforestation, is a result of attempts by the ruling elite to maintain political power, and how this was made possible by a one-party system which repressed any challenge to official policies.

11.2.3.2 Growth in human population and natural resource consumption

The Earth's human population is increasing. Some 5.6 billion people currently occupy our planet and that number is projected to double within the next 60 years (UN 1992). More people means a greater need for agricultural and industrial produce, settlements, transportation, and infrastructure. More and more natural areas will be modified, though to what extent will depend partly on the number of people they support (Boserup 1965). This growth will have serious impacts on the levels of biodiversity that can be maintained.

In most countries with high fertility rates, about half the population is under the age of 16. The resulting demographic momentum – that is, high birth rates in coming years due to the large number of people who will be reaching their reproductive years – means that global population will continue to grow for at least the next half century and probably longer, barring catastrophe. Another billion people are likely to be added to the world population for each of the next three decades (UN 1992). The rates and magnitude of this growth and the eventual size at which the global population stabilizes – critical considerations for biodiversity – depend on social and economic measures, especially on the rate of economic development in the developing countries.

The problems associated with population growth and distribution and loss of biodiversity are reaching critical proportions in many parts of the world. In all but one of the seven mega-diversity countries which together contain over 54% of the world's flora and fauna – Brazil, Colombia, Mexico, Zaire, Madagascar, Indonesia and Australia – population growth exceeded the world's average for the period 1950–90. In these countries, population growth is expected to continue, and if present trends persist, population increases are likely to lead to higher deforestation, degradation of land and loss of biodiversity. But the picture is seldom simple: Brazil, for example, has been experiencing an absolute decline in rural population since 1975 as rural people have moved to the cities. Declines in rural population can mean increased biodiversity in at least some cases where demand for biological resources also declines.

The impacts of population increase are felt in both terrestrial and aquatic habitats. Lundin and Linden (1993) estimate that already 66% of the Earth's population lives within 60 km of coasts and that population growth here is faster due to migration from inland areas. This has resulted in densities of 1000–2000 people per km^2 in the coastal zone of Asia, Central America and the Caribbean. Coastal migration affects developed and undeveloped countries alike: in recent decades migration to coastal areas of the Pacific Coast and the Gulf of Mexico has increased such that 50% of the US population now lives within 70 km of the sea (WRI 1994).

The average number of working-age people entering the labour force is about 35 million annually in developing countries, exacerbating the already serious problems of underemployment and unemployment. In the next 10 years, developing countries must generate 30 million new jobs each year just to absorb into the workforce the children already born. This means that the natural resource base of these countries will be increasingly under pressure if countries are unable to provide employment in industry and agriculture. For example, in India, almost 10 million entrants a year are projected in the coming decade. Since 73% of the Indian population is currently in rural areas, it is possible that agricultural lands and forests will need to absorb about 7 million new workers each year, placing additional pressures on already degraded resources (Bose 1991).

Population growth affects biodiversity: directly, through increased resource consumption; and indirectly, through fuelling the processes of poverty and migration, and causing a breakdown in social institutions determining the management of natural resources. However, the ways in which population pressure affects natural resources and habitats vary, and therefore it is necessary to understand the causes of such pressure before designing interventions for its control. One of the major causes is the effect of natural population growth on the regeneration rates of resources. Another cause is unsustainable resource consumption patterns, including excessive commercial use of resources, urbanization, and human uses of valuable or critical species. A third cause is increasing migration of poor people into ecologically fragile areas. Resource degradation can also occur when the population exceeds the social capacity of institutions to cope with environmental changes.

Human population density is clearly linked with wildlife habitat loss. The 20% of countries which have lost most like). In contrast, in the industrial world today, and among the large middle classes of the developing world, consumerism is all-pervasive: indeed it defines what constitutes 'the good life' (Durning 1992). An indication of the magnitude of the growth in consumption can be gained by several indicators in the growth of global consumption of resources since 1950 (Brown *et al.* 1992) (Table 11.2-18). Particularly remarkable is the order of magnitude increases in fertilizer use and natural gas production.

Whittaker and Likens (1975) have estimated that an 'agricultural world' in which most human beings are peasants, should be able to support 5 to 7 billion people, probably more if the large agricultural population were supported by an industry-promoting agricultural activity. In contrast, a reasonable estimate for an industrialized world society at the present North American material standard of living would be one billion. At the more frugal European standard of living, 2–3 billion would be possible. These figures represent not just the contrasting material demands of developed and developing societies but also their contrasting dietary habits. Huxley (1984) considers that dietary habits lie at the root of many of our troubles and calculates that the average Westerner consumes 65 kg of grain per year while the meat consumed accounts for over 900 kg of grain per year. The average Chinese consumes 160 kg of grain and under 20 kg of meat per year. In short, the range of food required to feed 200 million in the West would feed 1500 million Chinese. However, the current change in the economic status of China and Southeast Asian countries will result in a shift towards Western levels of consumption with drastic implications for world food reserves and future demand, and resulting impacts on biodiversity.

Most Asian and African nations have predominantly rural populations (70.1% and 67.3%, respectively). In contrast, in developed countries and in Latin America about 75% of the population is urban. Over the past 40 years, there has been an unprecedented growth in the world's urban population and by the year 2000 it is estimated that nearly half of the world's population will be urban (WRI, 1994). Urbanization affects biodiversity in four main ways:

- Geographical expansion of settlements and infrastructure displaces the existing vegetation and diversity through land conversion.
- Urban activities indirectly have a significant impact on hydrological and atmospheric systems at both local and global levels.
- Urban dwellers plant many species of plants around homes, along avenues and in parks. These are largely ornamental and often introduced species which displace the native vegetation, while adding to overall diversity.
- Urban demands for biomass require fuelwood, industrial wood, sawnwood and other products such as fruits and flowers from surrounding areas. Around cities, plantations of genetically similar trees are displacing the local vegetation to meet the urban demands for biomass.

Urbanization and its effects on biodiversity are discussed in more detail below (11.2.3.3) and in Section 13.3.8.1.

Another way in which population change affects biodiversity is through population movements. Environmental degradation is both a cause and a consequence of frontierward migration. It is a *cause* when decreasing ecological capacity forces people to move elsewhere, often entailing clearing forests for settlements and agriculture. It becomes a *consequence* when increasing populations exert pressure on resources for livelihood; short decision-making time horizons prevent many poor farmers from investing in soil or forest conservation techniques, especially when payoffs are not immediate (Shaw 1989).

Resource extraction activities have motivated movement into frontier sites (Cruz *et al.* 1992), including those initiated or supported by governments and aid agencies. More than half the developing countries in the tropics with annual deforestation rates of over 90 000 ha have populations in excess of 55 million and average annual population growth rates of 2–5% since the mid-1970s. Close to 30 million people reside in forests and protected areas in India and Indonesia. In the Rondonia area in Brazil, the population of small-scale cultivators has increased by over 15% per year since 1975, a rate that is many times higher than Brazil's annual population growth rate. Similar mass movements into tropical forests and protected areas have occurred (WRI, UNEP and UNDP 1990).

While the magnitude of population pressures on the environment can be measured, and in some cases predicted, another dimension of population pressure which is relevant in the management of natural resources is the social and cultural characteristics of the population. These characteristics determine the capacity of groups in dealing with changes in resources and access rules.

Because of accessibility to markets and increased immigration, which are often linked to population pressures, very few frontier sites have culturally homogenous populations and even areas, and can show striking differences among groups in their population growth rates, migration patterns, and their land-use and resource consumption patterns. Three aspects of the composition of populations have especially important impacts on the management of biological resources: recognition of ethnicity; gender-specialized roles in conservation; and differences in socioeconomic status. In any particular habitat or ecosystem, the ways in which resources are used and managed will differ among ethnic groups, between men and women, and between different socioeconomic groups. It is important to distinguish interventions in terms of which population sub-groups are most vulnerable to changes in resources (e.g. indigenous peoples, women, the poor).

11.2.3.3 Urbanization and biodiversity

The urban environment is a mosaic of human-made, natural and semi-natural habitats with climatic and hydrological conditions that distinguish it from adjacent rural areas (Berry 1990; McPherson 1994). While only 18% of cities are open space (Nicholson-Lord 1987), as much as 40–70% may be green and photosynthesizing (Nicholson-Lord 1987; Ignatieva 1994; Loucks 1994).

Around 45% of the world's population is urbanized but this is unevenly split between the developed countries (over 70%) and the developing countries (just under 40%) (Berry 1990; WRI 1994). However, the gap is closing and the urban growth rate in the latter is currently four times faster than in the developed countries and their urbanized area is predicted to double over the period 1980–2000 (UNEP 1992). Some general trends in city growth include:

- an increase in cities of over 1 million inhabitants (increasing from 190 in 1975 to over 400 by the year 2000; Berry, 1990);
- 'counterurbanization' in the developed countries with substantial numbers of people leaving core city areas for the less built-up suburbs (OECD 1985; Nicholson-Lord 1987; Berry 1990; Skibniewska 1994) leading to more disturbance of green-belt areas and more dispersed populations;
- higher numbers and growth rates of cities in coastal areas where 60% of the world's population currently lives (Walker 1990; Lundin and Linden 1993).

11.2.3.3.1 Effects of urbanisation on biodiversity. These can be considered from two perspectives: the direct effects urbanization has on biodiversity (the loss of habitat; the fragmentation of habitat; the creation of new human-made habitats such as cemeteries, derelict lands, rubbish tips, etc.); and the indirect effects it has through its impact on hydrological systems and the atmosphere. Another indirect effect which is considered elsewhere and will not be discussed further here is the effect on the rural environment of the urban demands for resources.

Indirect effects: Urbanization covers the urban landscape with impervious surfaces (one US study of an average, urban area estimated this at 12–37% of total urban area: Loucks 1994) and these have a dramatic effect on runoff, an effect which is exacerbated by extensive sewers and drain systems (Binford and Buchenau 1993; Goudie 1993). A comparison of contaminant profiles of urban runoff and raw domestic sewage, indicates that urban runoff contributes more suspended solids, pesticides, chlorides and heavy metals while the sewage is the main source of nitrogen and phosphorus (Goudie 1993). The less soil available, the more concentrated are these chemicals when they arrive in the rivers.

Most African and Asian urban centres have no sewerage systems at all (UNEP 1992) and in developing countries human sewage is the most important pollutant of freshwater and coastal zones (Markham 1994). Sewage, both treated and untreated, contains high nitrate and phosphate loads and, together with the nutrients and contaminants from urban runoff, produces an assault on the aquatic environment that has resulted, at best, in eutrophication, and at worst, in its almost total destruction and associated loss of biodiversity.

Highly polluted systems are characterized by the loss of fish life and small, specialized benthic life (composed of chironomid larvae and tubificid worms) as was recorded for the River Thames in the 1950s (GESAMP 1990; Allan and Flecker 1993) and other rivers (Whiteley 1994). Eutrophication of rivers, nearshore marine ecosystems and semi-enclosed seas (such as the Baltic, Black Sea and Mediterranean which are naturally oligotrophic), although producing less profound effects on species diversity, alters species composition and destroys the integrity of the ecosystem (Caddy 1993; Hammer *et al.* 1993).

Urban centres have substantial daily freshwater needs. Abstraction can have serious effects on species diversity and composition, both within the river, where lower water levels affect the fauna and riparian vegetation (Binford and Buchenau 1993) and in the nearshore waters where rivers meet the sea. Reduced water flows have caused increased salinity in San Francisco Bay, the Caspian Sea, the Black Sea and the Mediterranean with effects including a reduction in fish populations and changes in the composition of fish communities (GESAMP 1990; Caddy 1993; Goudie 1993).

A common phenomenon near and within urban areas as a means of flood control, channelization drastically alters the physical characteristics of a stream, increasing water velocity and reducing habitat diversity and riparian vegetation (and thus nutrient input to the stream). As a result some species are eliminated and species composition is altered (Allan and Flecker 1993; Binford and Buchenau 1993; Goudie 1993).

City-produced contaminants (such as CO_2, SO_2, nitrous oxides, ozone, etc.) have effects within the city, close to the city and globally (Berry 1990; Hawksworth 1990; Westman 1990). Lichens have proved to be excellent monitors of sulphur dioxide pollution which has caused the absence of any species in the central areas of some major cities and their severe reduction in cover and diversity

Key Messages

- Virtually every decision people make can have an influence on biodiversity, either positive or negative. Losses in biodiversity can affect the sustainability of society. Continued policy failures can lead to domestic turmoil.
- Loss of biodiversity at the hands of people will continue and even accelerate, because the amount of space on our planet and the natural resource base are fixed, but both consumption and human population are expanding, leading inevitably to increasing pressure on limited resources. Therefore, these resources need to be managed more effectively, and the policies adopted need to be supportive of the Convention on Biological Diversity.
- Although it is by no means clear whether poverty, with its pressures to survive, or affluence, with its pressures to consume, ultimately leads to greater loss of biodiversity, it is obvious that the rural poor cannot conserve their biological resources if this is in conflict with their immediate survival needs.
- Human activity is not necessarily incompatible with the maintenance of biodiversity, and many human actions tend to foster greater biodiversity (especially components which people consider desirable). However, some important components of biodiversity are most likely to prosper in areas that are remote from human influence; where extreme environmental conditions prevail; or are associated with conditions provided by humans (for example, within protected areas, agricultural lands, and other systems that are managed by people).
- Tropical islands can provide a preview of the environmental situation that is likely to become more prevalent on the world's continents in the future. These islands typically have high population densities, exhibit highly fragmented landscapes, and have already experienced significant extinction events. On the other hand, their landscapes are now enriched by the addition of imported plants and animals that form new combinations of communities and ecosystems (which sometimes replace native species and communities which may have been unique in the world).
- Economics provides a useful perspective for understanding human relationships with biodiversity. Property rights is a crucial element in understanding why biodiversity is lost. Goods can be bought and sold only when property rights to them are well defined, where the seller truly owns the goods, and has the right to transfer these goods to others. Biodiversity, on the other hand, is a public good that is provided to everyone, rather like law and order and defense. Market economies, if left to themselves, typically under-provide public goods. Thus property rights work well for bread as a private good, but much less well for a public good such as genetic variation in wheat types. The full social benefits of tigers, rhinos, portfolios of germplasm, marine resources of a global commons, and so forth, are public goods beyond appropriation by markets, even when market value is fully enhanced by all the devices of the law. Therefore, new policies are required which help enable public goods to be managed for the benefit of society.

Efforts to develop alternative environmental accounting systems are at an early stage, but the process of refining existing accounts and proposing alternative accounts is helping to sharpen discourse and illuminate critical assumptions and issues.

11.2.3.6 Inequity in the ownership, management and flow of benefits from both the use and conservation of biological resources

In most countries, ownership and control of land and biotic resources, and all the benefits they confer, are distributed in ways that work against biodiversity conservation and sustainable living. The rapid depletion of species and the destruction of habitats are the norm in many countries where a minority of the population owns or controls most of the land. Profits from logging or fishing flow to the few, while the local communities dependent on the continued production of the resources pay the price (see also 11.2.3.1.3).

A second problem arises from the concentration of resource control and responsibility for environmental policy decisions primarily in the hands of urban men. In many societies women manage the environment and possess far greater knowledge of biodiversity's value to farming and health.

A third issue is the way international trade, debt and technology transfer policies and practices foster inequities that resemble – and often reinforce – those found within nations. By 1988, developing countries were transferring \$32.5 billion net to industrialized countries, excluding other implicit resource transfers not involving direct financial flows (United Nations 1989). (At the beginning of the decade, $42.6 billion had been flowing to developing countries.) Between 1986 and 1993 developing countries paid $1253 billion to serve a growing foreign debt that reached around $1550 billion in 1994 (IMF, 1994). If the developing countries continue to be shut out of markets, deprived of access to technology, and burdened with debt, they will have neither the means nor the incentive to conserve their resources for the future.

11.2.4 Conclusions

The major cause of biodiversity loss in recent historical times is human action, primarily land use that alters and degrades habitat to serve human needs (Pimm and Gilpen1989; Freedman 1989). Yet the ability to forecast the impact of specific actions on biodiversity is not yet well developed, and practical techniques for conducting such analyses are at only a very preliminary stage (OTA 1987; Soulé and Kohm 1989). Machlis and Forester (1992) have pointed out that while a large number of conceptual and predictive models of the interactions between humans and nature exist, explicit models of biodiversity loss are sparse and incomplete. While many of the generic models treat 'environmental change' or 'ecosystem alteration' as the dependent variable, it is not at all clear that biodiversity loss can simply be substituted for these more general factors. Biodiversity loss is a special case of environmental change, and the socioeconomic factors that influence it may not have generic impacts. For example, biodiversity loss measured as a reduction in species richness may be so dependent upon the original number of species at a specific locale that certain generic models will fail to explain, much less predict, even the most dramatic levels of biodiversity loss. The importance of habitat in the preservation of biodiversity may suggest that spatial relationships at the local scale will play a more significant role in biodiversity models than, say, models of climate change.

However, it seems apparent that the issue of scale is crucial, as biodiversity loss is embedded in a complex human/environment system that operates at several hierarchial levels; socioeconomic factors important at one scale may be less important at another, and at each different scale, new variables and relationships may emerge as critical driving forces.

The driving forces of human-induced change will vary with the type of change involved, and forces that drive some changes may lessen others (Meyer and Turner 1992). For example, rising agricultural prices may provide an incentive for clearing forests, while also providing an incentive to adopt soil conservation measures. Second, the same kind of land-cover change can have different sources in different areas, with deforestation in some areas primarily for timber extraction, in others for shifting cultivation, and in others for establishment of plantations. In the dynamics of underlying causes, no agreement yet exists on the level at which adequate explanation is achieved. For example, some may consider that deforestation by agricultural expansion is driven by population growth, while others would contend that agricultural expansion helps to cause population growth; others will suggest that population growth needs to be explained in terms of the socio-political and economic conditions that promote it. Ehrlich and Ehrlich (1990) have attempted to provide a single comprehensive approach to the question of driving forces, using the equation I = PAT where I represents environmental impact, as the product of P (population), A (affluence) and T (technology). Thus, human impact is a product of the number of people, the level at which they consume, and the character of material and energy flows in production and consumption. Meyer and Turner (1992) have pointed out that this formula suffers from the handicap of a mismatch between its categories of driving forces and the categories customarily used in the social sciences. Neither 'affluence' nor 'technology' is associated with a substantial body of social science theory.

Today's pressures on the natural world mean that the genetic diversity of many species is being reduced because the total sizes of populations are decreasing and they are often being split into small, widely separated, subgroups which cannot interbreed. Others might argue that this is one of the processes of speciation, with humans serving as a new isolating mechanism.

11.3 Information requirements for the sustainable use of biodiversity

11.3.1 Introduction

Effective action must be based on accurate information, and the more widely shared the information, the more likely it is that individuals and institutions will agree on the definition of problems and solutions. However, the current state of knowledge is still largely inadequate to evaluate precisely what are the impacts of human activities in different ecosystems, and to understand what are the relationships between economic activities, development and conservation of biodiversity. Gaps in knowledge may have at least three origins.

First, the lack of information resulting from an insufficient research effort, especially for the inventory of species and ecosystems (see 11.3.2.4), for understanding how components of ecosystems fit together and interact with one another, for information on traditional use and knowledge of biodiversity, and for changes in ecosystem use. A significant increase in funding and man-power could fill most of these gaps. However, while some scientists argue that until we understand the natural environment, it will be difficult to understand how human societies interact cultivation (Lenski and Lenski 1978). In what follows we indicate how this view differs from others that have existed in the past and that still exist in some societies. This world view is characteristic of large-scale societies, heavily dependent on resources brought from considerable distances. It is a world view that is characterized by the denial of sacred attributes in nature, a characteristic that has its roots in Greek philosophy, and became firmly established about 2000 years with the Judaeo–Christian–Islamic religious traditions (White 1967).

For much of human history, people have lived in small-scale, kin-based societies; attempting to obtain a subsistence from their immediate surroundings. All of humanity lived in such societies of hunter–gatherers or horticulturalists until the beginning of irrigated agriculture 6000 years ago. Much of the Earth's surface was still covered by them until the European expansion beginning five centuries ago. (Crosby 1986; Cavalli-Sforza *et al.* 1994). It is only now that such societies are finally disappearing (Lee and DeVore 1968; Diamond 1992). The world view of such societies tends to be strikingly different from the modern world view. In particular, the values assigned by such societies to biodiversity are grounded on very different premises.

The stylized facts are as follows. In such societies a relatively small number of people, many of them related to each other, tend to be in face-to-face contact over long periods, often their whole lifetimes. Their relationships to each other are moulded by ties of altruism and reciprocal help (Trivers 1971; Wilson 1978). With hunting, gathering, fishing and low-input agriculture and animal husbandry as the mainstay of subsistence, the people are also closely tied to the natural resources of their immediate environments. They therefore tend to view themselves as members of a community that not only includes other humans, but also plants and animals as well as rocks, springs and pools. People are then members of a community of beings – living and non-living. Their relationships with other community members, be they trees, birds or mountain peaks are moulded by the relationships with other human members of the communities, as recipients of altruistic or mutualistic favours. Thus rivers may be viewed as mothers; the Ganges of India is Gangamai, mother Ganga to inhabitants of the Gangetic plains, and has probably been viewed so for millennia. Animals may be treated as kin; thus antelopes are brothers to the followers of the Bishnoi sect of Rajasthan and Haryana in northwestern India (Sankhala and Jackson 1985). The Koyukan people of Alaska believe that the key to the success of a good bear hunter lies in the respect he shows towårds his victims (Nelson 1993).

The manifold restrictions and taboos on the way nature is to be treated (grass burnt, honey collected or deer hunted) are an expression of respect for other non-human members of the community of beings. Anthropologists have documented such practices; that they are a significant component of the ways in which hunter–gatherer–subsistence cultivator–pastoral people relate to their environment is widely acknowledged (McNeely and Pitt 1985). What is disputed is whether such respect, such restraints, are best viewed as irrational, superstitious practices with no significant implications for conservation of biodiversity, or whether they are to be viewed as practices evolved by societies through a long-term process of trial and error actively to promote sustainable use of biological resources and conservation of biodiversity. In favour of the first view is the mounting evidence of many extinctions of larger birds and mammals by hunter–gatherer–horticulturalist colonists of the Americas, Madagascar, New Guinea, New Zealand and the Polynesian islands (Diamond 1994). But, such extinctions by new colonizers may also be reflections of a long period of striking roots in a locality and of gradual accumulation of experience of how nature responds to human use that must elapse before societies can culturally evolve effective traditions of restraints on resource use (Gadgil and Guha 1992; Gadgil, Berkes and Folke 1993; Gadgil 1995). Thus the early settlers of Madagascar were probably responsible for extinction of some of the largest species of lemurs, but the remaining species then came to be treated as sacred animals, protected against hunting, and so survived to the modern times when the influence of Christianity led to the gradual loss of traditional protection (Jolly 1980) (see Section 11.1.2 for further discussion). Indeed, on balance it appears reasonable to conclude that the many restraints on the use of natural resources that include protection everywhere to keystone resource species such as fig trees protection to highly susceptible stages such as birds breeding communally at a heronry, and protection to entire biological communities of sacred sites such as sacred groves or ponds, may have evolved culturally in response to the need to ensure more sustainable use of biological populations and conservation of biological diversity (Gadgil and Berkes 1991) (also see Sections 11.1, 11.2 and 13.6 for further discussion).

Nelson (1993) and Diamond (1993) have observed that such restrictions on the use of natural resources are often against the short-term interests of individuals – as are environmental regulations in modern societies. There exists, therefore, a private incentive to flout such social conventions, to violate the taboos, and this cannot be contained solely by respect for non-human fellow members of a community of beings. Compliance in such societies is typically assured through two devices: fear of the wrath of offended nature spirits and social sanctions against offenders. Thus traditionally Gangtes, a group of shifting cultivators in the state of Manipur in northeastern India, feared that violation of taboos against cutting of sacred groves may lead to illness or death, even if no other country's best-known wildlife reserves and protected areas (Norton-Griffiths and Southey 1995), although Kenya has not diversified its wildlife sector in ways – sport hunting is banned, for example – that could enhance revenues (Edwards and Allen 1992). This illustrates one of the major difficulties in the management of wild resources: the price of a single species product in most cases cannot reflect habitat management costs. However, amortizing these costs across an array of species and products, over a number of years, may make the investment economically feasible. Multiple uses of wild species or habitats add value to the resource. Likewise, given the vicissitudes of market demands for wild products, it is not advisable (nor probably sustainable) for a management programme to be based on a single species. In Zimbabwe, for example, an elephant may be harvested for meat, used to make a variety of wildlife products, marketed for sport hunting, and promoted for tourism. Successful crocodile farming operations in various countries have incorporated tourism and sale of breeding stock into their business activities. Each additional use adds to the potential value of the resource and increases the incentive to maintain and manage the resource.

The value of wild habitats and their species to rural communities is often overlooked in national policy. Rural people can be (and often already are) allies – not adversaries – in the sustainable management of wild resources. Government agencies and NGOs will need to invest heavily in working with local communities to educate and train residents to be wildlife managers and to use wildlife to meet their social and economic development goals. In the few places where this has been done, such as Zimbabwe's Campfire Programme (Child 1994), wildlife and biodiversity have not only held their own but actually increased.

13.3.6 *Biodiversity prospecting*

The search for wild species whose genes can yield better crops and new medicines – sometimes called 'biodiversity prospecting' – has created a rapidly growing new industry. Already, pharmaceutical companies are screening the genetic storehouses found in Costa Rica, Brazil, Micronesia, China and other biologically diverse countries, as well as forest habitats in temperate countries and hydrothermal vents deep under the sea. Without appropriate legislation, benefit-sharing arrangements and technical guidelines, this 'gene rush' may do little to conserve ecosystems and provide few if any benefits to the people living in or near them. Done well, though, bioprospecting can bolster both economic and conservation goals while advancing medical and agricultural needs to sustain growing human populations (Lash 1993).

For decades, ecologists and environmentalists have argued that agricultural, pharmaceutical and other commercial applications of biodiversity should help justify its conservation. Since the mid-1960s, however, industry investment in natural products research had been small, even declining during the 1970s (Reid *et al.* 1993). As noted above in Section 11, in September 1991, Costa Rica's National Biodiversity Institute (INBio) – a private, non-profit organization – and the US-based pharmaceutical firm Merck & Co., Ltd. announced an agreement that helped reverse this trend. The agreement provides US$1.135 million to INBio from Merck to conduct research and sampling of wild plants, insects and microorganisms in exchange for chemical extracts which Merck could screen for potential pharmaceutical applications. If any commercial products resulted, Merck agreed to pay INBio royalties which INBio will use to further its inventory and research, and to support a fund for the management of Costa Rica's national parks (Reid *et al.* 1993). Since then, a growing number of biodiversity prospecting agreements have been negotiated between industry, research institutions and governments around the world.

The dramatic changes taking place in the use of biodiversity in the agricultural and pharmaceutical industries are primarily the result of advances in gene transfer and biochemical screening technologies. Breeders can now move genes from unrelated species into agricultural crops through genetic engineering. For example, breeders are now screening plant extracts for antifungal or antiviral activity, isolating and transferring the genes responsible for those chemicals into elite crop lines within a matter of days (Reid *et al.* 1995). In the pharmaceutical industry, natural products drug discovery processes have traditionally required substantial quantities of material. Advances in techniques for extraction, screening, fractionation and chemical identification now require much less material and are much less costly. Samples of only 200–500 g of dry plant material are now needed to isolate, elucidate the structure of, and test a novel plant metabolite, instead of the 100 kg or more that may have been needed a decade ago. Where the screening of 10 000 samples would have cost US$6 million in the mid-1980s, it now costs only US$150 000 (Reid *et al.* 1995). And, tissue and cell culture – the 'next frontier' of natural products research – now allows small samples of plants to be collected and screened without any need to return for additional supplies (see Section 10 for discussion of biotechnologies).

The potential of biodiversity prospecting to adversely impact genes, species and ecosystems is much less than that of many practices in agriculture, forestry and fisheries. Unregulated biodiversity prospecting, however, can speed the destruction of a species. In one particularly notable case, the entire adult population of the shrub *Maytenus buchananii* – the source of the anti-cancer compound maytansine – was harvested (27 215 kg) in Kenya by the US National Cancer Institute for testing in its drug development programme (Oldfield 1984). To avoid such problems, research agreements, such as those specified under the Philippines Presidential Executive Order on Biodiversity Prospecting (Reid *et al.* 1995), can specify ecological and population studies to determine limits for sample collections. All collections must be approved in accordance with these limits.

The greater challenges are to implement mechanisms to equitably share the benefits of biodiversity prospecting and to invest some of those benefits in biodiversity conservation. This is complicated by the new ways in which biotechnology generates wealth and the lack of legal and institutional experience to ensure that benefits are equitably shared.

The economic value of wild biological diversity is increasing because of biotechnologies. Any organism is now a source of chemical and genetic innovation with potential application in the agriculture, pharmaceutical or industrial chemical industries. And, the cost of working with new genetic material or identifying and isolating new chemicals is decreasing rapidly. Ironically, while the economic value of genetic resources in general is rising, the same is not true for the commercial value of any given species or extract. This is because the technological advances that make biodiversity prospecting feasible have greatly increased the effective supply of species. Plant breeders no longer have to restrict their search for new genes to crop relatives, or even to the plant kingdom. In the near term, there is competition among firms for access to a relatively small number of quality suppliers of raw biochemical and genetic material. Over the long term, the large supply of material and decreasing costs of natural product research are likely to hold the 'market' value of samples of raw materials very close to the direct labour cost of obtaining the sample (Reid *et al.* 1995). In other words, most of the benefits from biodiversity prospecting will not come from finding raw materials: they will come from developing those materials into commercial applications.

The keys to sharing the benefits that flow from biodiversity prospecting will depend on two issues. First, how can mechanisms be designed and implemented to target benefits to countries and communities that conserve their biodiversity and make it available for biodiversity prospecting? Second, how can countries build the research and commercial capacity to make use of their own genetic resources? The Convention on Biological Diversity addressed these issues by reaffirming the sovereign rights of nations to their biodiversity, establishing the right of nations to regulate access to genetic resources, and creating technology-transfer mechanisms. The legal aspects of regulating genetic access are discussed in 13.6.3.1.

13.3.7 Managing the impacts of tourism on biodiversity

Tourism has become one of the largest economic activities in the world – if not the largest. In developing countries alone, it generated US$62.5 billion in 1990. Throughout the world, international arrivals of visitors were estimated at nearly 450 million people in 1990, generating nearly US$3.5 trillion. This amount is greater than that generated by the global agriculture, automobile or steel manufacturing industries (WTO 1991, 1992). The rapid growth in tourism has produced more infrastructure, increased pollution, put unsustainable demands on local environments, and created adverse impacts on biodiversity. In many coastal areas, for example, strings of high-rise hotels stretch for kilometres along beaches that not long ago might have been diverse habitats of coastal forest, mangroves, seagrass beds and coral reefs.

But tourism, particularly ecotourism or nature tourism, is also seen as a force to help the sustainable use and protection of biodiversity, since the wildlife and natural habitats that draw visitors also generate significant income and foreign exchange in some parts of the world. And this influence will increase, as nature tourism is one of the fastest-growing elements of tourism, increasing at a rate of 7% annually (Filion *et al.* 1992). In Kenya, for example, tourism is the country's largest earner of foreign exchange: nearly all its visitors come to visit the national parks and view wildlife (McNeely *et al.* 1992). Nature tourism is also a tremendous economic force in Costa Rica and a growing number of other countries, and may be valued at US$50 billion world-wide by the turn of the century. Tourism, therefore, is both a benefit and a curse to the sustainable use and protection of biodiversity.

The big challenges in minimizing the impact of 'mass tourism' on biodiversity lie with problems such as sewage treatment in coastal areas, site selection of hotels, pollution caused by large fleets of tour buses, impacts of scuba divers on coral reefs, and environmental issues posed by golf courses, waste treatment of big resort hotels, etc. Ecotourism can contribute to pressures on biodiversity through serious trail erosion, cutting down trees for fuelwood, inappropriately sited infrastructure in protected areas, and waste disposal (see also 13.5.3.3). While these may occasionally seem peripheral to the management of biodiversity, they have caused significant local losses of biodiversity in many parts of the world and – at least indirectly – have contributed to the global endangerment of species. One basic step towards protecting biodiversity from unintended adverse impacts associated with tourism is to use environmental impact assessment (EIA) procedures to identify serious problems before they occur. Siting is the most important consideration in the EIA process. For example, in fragile island environments, all tourism facilities should be placed well away from sensitive habitats (e.g. seabird and turtle nesting areas, mangrove forests, salt ponds, seal and sealion rookeries) and well above the high-water mark since natural erosion and accretion cycles are a feature of many beaches. Careful planning, design, and building guidelines for tourism

Examples of advanced work on bioregional management include the tri-country Waddensee programme along coastal Denmark, Germany and the Netherlands (Wadden Sea Assessment Group and Trilateral Working Group 1991), the Serengeti Greater Ecosystem in Kenya and Tanzania (Ministry of Tourism, Natural Resources and Environment 1991), La Amistad in Costa Rica and Panama (Gobierno de Costa Rica 1990), the Great Barrier Reef in Australia (Murdoch 1992), and Yellowstone National Park in the United States (Rawlins 1994), as reported by Miller (1995).

13.4.2.1.4 Effectiveness of protected areas for maintaining biodiversity. Two issues are critical in assessing the effectiveness of protected area systems for biodiversity conservation. First, do the protected areas have clear biodiversity management objectives and the appropriate boundaries, legal status, funding and personnel to obtain those objectives? Second, do local people benefit economically from protected areas, and do they have incentives to use the resources in surrounding areas sustainably? Assessing the biological effectiveness of a protected area network requires information on the range of biodiversity elements contained within the area, and on the quantity and kind of management inputs going into protected areas management.

While reasonably accurate data on the number and area coverage of protected areas are available for most countries, more specific information on the effectiveness of protected areas in conserving elements of biodiversity is generally not available, even for countries with well-established protected areas. For example, Newmark (1987) documented the extirpation of large mammals during the twentieth century in every national park in the continental United States, indicating the lack of systematic monitoring and protection even of large mammals in the US National Park System. While some countries have attempted to conduct surveys of key species, information on the biodiversity contained within protected areas is inadequate for virtually all groups, except some vertebrates. Monitoring of genetic diversity has received far less attention than it deserves.

In India, which has a protected areas network of over 500 national parks and nature reserves covering nearly 5% of the country's land area, two nationwide surveys have been conducted to assess the effectiveness of the network. On the biological front, the Wildlife Institute of India evaluated the biogeographic coverage of the existing protected areas network and suggested options to plug the major gaps (Rodgers and Panwar 1988). The second survey (Kothari *et al.* 1989), conducted simultaneously by the Indian Institute of Public Administration (IIPA), looked at various management parameters of the protected area network: legal status, research and monitoring, human use and management activities (see Box 13.4-2).

McNeely *et al.* (1994) review protected areas management effectiveness around the world. Priority actions to improve management and increase the ability of protected areas to conserve biodiversity are presented by region (14 regions including coastal marine areas).

Social and economic measures to enhance the compatibility of biodiversity management in protected areas with economic and social development are presented in 13.5. These measures are vital to the success of protected areas management strategies in developing countries, and frequently in developed countries as well.

13.4.2.2 Managing corridors and natural habitat fragments

Fragmentation of natural ecosystems is generally seen to be one of the most important threats to biodiversity world-wide (Saunders *et al.* 1991; Bierregaard *et al.* 1992; Kattan *et al.* 1994). Fragmentation occurs when human activities such as agricultural development, forestry or urbanization remove large proportions of the natural ecosystem and replace them with a greatly modified matrix, within which small remnants of the native ecosystem remain. This process results not only in vastly reduced areas of the natural ecosystems but also their subdivision into small and relatively isolated fragments. Roads, railroads, powerlines and pipelines are also important fragmentation factors in many places, although they may not directly convert large areas of habitat.

13.4.2.2.1 Response of ecosystems to fragmentation. The response of biota to ecosystem fragmentation has received much study and is well documented in the conservation literature. This research has largely centred on species and community responses to changes in ecosystem size and isolation, most of it within the theoretical framework of island biogeography (Diamond 1975; Wilson and Willis 1975; Simberloff 1988; Shafer 1990; Soulé 1991). Considerable debate has focused on such questions as whether a small number of large reserves will maintain native biota better than a larger number of smaller reserves, and whether reserves linked by corridors are better than those without linkages. Much of this debate has been carried on with a paucity of data from real systems. Saunders *et al.* (1991) pointed out that, until recently, relatively little emphasis has been placed on understanding the effects of fragmentation on the structural and functional aspects of natural and managed ecosystems.

Much attention has been focused on the effects of habitat size and isolation on the biota in remnant areas. It was suggested that remnant areas should be similar to islands and that the biota should follow the rules developed by island biogeography (Diamond 1975; Wilson and Willis 1975). While the biogeographic processes operating as a