GLOBALISATION AND NATURE POLICY: AN INTEGRATED ENVIRONMENTAL-ECONOMIC FRAMEWORK

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Abstract

The search for a framework to study globalisation, economics and ecology for nature conservation and biodiversity protection requires the integration of concepts, theories and models from economics and ecology. This allows for the study of interactions between economic and ecological processes, including protection of species and biodiversity, sustainable and optimal use of renewable resources, land use and physical planning, maintenance of nature areas, acquisition of nature areas, and development of outdoor recreation areas. Economic theories relating to nature and ecosystems have focused on notions of capital theory and intertemporal trade-offs, decision making under uncertainty and irreversibility, and marginal valuation and cost-benefit analysis. Recently, the conservation and valuation of biodiversity, and the resilience of ecological and combined ecological-economic systems, have attracted a great deal of attention in the environmental and resource economics literature. In addition, the distinction between local and global costs and benefits of environmental and biodiversity policies is regarded to have significant impacts on international co-operation.

Ecology can be incorporated in economic analyses in various ways, notably by offering information about the hierarchy of dynamic ecological processes, including population dynamics, ecosystem succession and cycles, and long run trends of selection and evolution. Biodiversity has been linked to resilience in the analysis of complex ecological-economic systems. Understanding of ecosystem irreversibility and uncertainty can improve economic analysis of decisions with impacts on ecosystem. Ecosystem performance indicators, such as those proposed around the concept of ecosystem health, can be useful for multidisciplinary modelling and evaluation studies. Finally, monetary valuation studies of goods and services provided by ecosystems can be complemented by detailed information about ecosystem scenarios and functions.

The paper discusses existing approaches to integrate economics and ecology, reviews the most important studies found in the literature, and suggests a number of general frameworks and models to address pressing policy questions relating to globalisation, conservation of biodiversity, sustainable use of natural resources, and sustainable land use.

1. INTRODUCTION

The global expansion of economic activity puts tremendous pressure on ecosystems all over the word. With economic growth and development, mankind has modified or radically altered natural areas on the Earth. Given the growing global interdependence of countries, both economically and ecologically, environmental problems need to be addressed on a global basis. Or, in other words, globalisation – the move towards a global economy where national borders cease to matter – is the new order that binds mankind in mutual interdependence. The term globalisation is mainly heard in connection with economic phenomena, although it has implications far beyond the realm of economics. From an economic perspective, globalisation works in three main areas. First, there is a globalisation of trade, which is defined as the creation of an integrated world market through the process of trade liberalisation. The second area is the globalisation of production whereby transnational corporations set up factories in countries other than their own 'home' country. Finally, there is a globalisation of financial markets, which is the removal of national controls on financial transactions in order to encourage the flow of finance across national borders. In addition to the transnational flow of capital, trade, labour and technology, other drivers of gobalisation are cultural processes (as they are transmitted electronically through the World Wide Web), the spread of diseases, travelling and foreign holidays, and pollution.

Together with increased global human activities, most of the original habitats have been destroyed. Habitat loss is, among others, considered the primary threat to biological diversity and to protect species habitats must be protected (Barbier *et al.*, 1994; McNeely *et al.*, 1995; Primack, 1998). To this end, nature policy is of major concern. Nature, however, is a term that requires a clear and practical definition. Eijsackers (1996) states that nature includes all organisms on earth in interaction with their environment. In addition, nature is considered as not being altered by human beings.

Nature protection can take place in two ways. An attempt can be made to preserve nature by placing limits on its use. This is often done by declaring particular areas as national parks or nature reserves. An alternative to protect nature is to encourage its sustainable use. This means exploiting natural assets in such a way that their stocks do not diminish. The term 'conservation' is used to describe the "... options in which the essential features of the natural habitat are maintained but some of the habitat area or some of its features are traded off for development benefits." (Pearce and Turner, 1990, p. 311). The term is often used to denote the protection of nature for future use and indicates that human activities are not always incompatible with the protection of nature. The term 'preservation', however, is used to "... describe the non-development option in this case." (Pearce and Turner, 1990, p. 311). This means that a given habitat can either be developed or preserved in its natural state. There is, in other words, no compromise. [1]

Protection of endangered species and of the wild lands that are their native habitat has attracted the attention of researchers in both natural and social sciences. As a result, a large body of literature has evolved around this topic. In many contributions it is emphasised that ecology, economics and possibly other disciplines can best work together to improve the understanding of the rapid depletion and degradation of the world's natural resources (e.g., Wilson, 1988; Costanza

[1] Preservationists tend to think that a given habitat is either preserved or destroyed for economic development. They feel that any use of natural assets will lead to destruction of biodiversity. The view of conservationists, however, is that mankind has to demonstrate that conserving nature is economically worthwhile and that incentives to secure such conservation have to be designed (Turner *et al.*, 1994). According to Pearce and Turner (1990) some of the fiercest debates in the environmental literature are between preservationists and conservationists (although the authors do not give references). Nevertheless, a clear distinction between 'preservation' and 'conservation' is often lacking. As a result, the two concepts are regularly used interchangeably (Heywood and Baste, 1995).

et al., 1991; Perrings *et al.*, 1992; Barbier *et al.*, 1994; Primack, 1998). Furthermore, the threatened state of the natural environment has become a major issue in policy making. Governments need to allocate their resources in the most efficient manner. Well-considered arguments for nature protection should not only be based on ecological grounds but also on economic grounds. In other words, nature policies should satisfy both (certain) ecological and (certain) economic criteria. Managing the degradation of the environment requires policy prescriptions for influencing economic activity and incentives.

The search for a framework to study policies for nature conservation and biodiversity protection has stimulated the integration of concepts, theories and models from economics and ecology (van den Bergh, 1996). This requires collaboration among ecologists, economists and related disciplines, i.e. a multidisciplinary approach. Integrated ecological-economic frameworks have to be developed that address particular aspects of economy-ecosystem interactions, such as protection of species and biodiversity, sustainable and optimal use of renewable resources, land use and physical planning, acquisition and maintenance of nature areas, and development of outdoor recreation areas.

This paper aims to present an overview of relevant ideas in ecology and economics that have been integrated or are amenable to integration. This will include a review of the most important studies in the literature. In addition, a number of general frameworks and models are proposed that address pressing policy questions.

The organisation of the paper is as follows. Section 2 starts with a concise description of nature policies at national and international levels. It will be argued that nature policies cut across the disciplines of ecology and economy. The concern of Section 3 is with the former; it addresses ecological concepts and theories, as well as ecological implications of the loss of nature. Section 4 discusses economic concepts and models relevant to the analysis of nature policies. Some of these already integrate economics with ecology. Section 5 goes one step further, and tries to identify further options for integration, based on the elements presented in sections 3 and 4. Finally, Section 6 presents recommendations, a reseach agenda and conclusions.

2. NATURE POLICIES

2.1 International level

While the major nature control and protection mechanisms that presently exist in the world are based within individual countries, increasingly international agreements are being used to protect nature and species. International co-operation with regard to the protection of nature is an absolute requirement for several reasons (Turner *et al.*, 1994; Primack, 1998). First, as species migrate across international borders protection efforts in one country will be ineffective if their habitats are destroyed in a second country to which an animal migrates. Second, due to the international trade in biological products, a strong demand for a product in one country may result in the overexploitation of particular species by another country, so as to supply this demand. Third, as will be discussed in more detail later, the benefits of nature are of international importance. The international community makes use of species in agriculture, medicine and industry. In addition, nature helps to regulate climate, supports the stability of biochemical cycles and has an international scientific and recreational value. Finally, several types of environmental pollution such as acid rain, which threatens ecosystems, are international in scope and require international co-ordination of environmental policies.

During the 1970s, four global treaties that address specific aspects of biodiversity were established (Miller *et al.*, 1995):

- The Ramsar Convention on the Conservation of Wetlands of International Importance Especially as Waterfowl Habitat (1971);
- The Convention Concerning the Protection of the World Cultural and Natural Heritage (1972), or the World Heritage Convention;
- The Convention on the Conservation of Migratory Species of Wild Animals (1973);
- The Convention on International Trade in Endangered Species of Wild Fauna and Flora (1979), or CITES.

Other international treaties have been included to protect species and ecosystems, but they all are voluntary. Countries can withdraw from the convention to pursue their own goals when the conditions of compliance are too complicated or act contrary to national interests (Primack, 1998).

One of the most significant attempts in adopting a global approach to sound environmental management was the United Nations Conference on Environment and Development in June 1992 in Rio de Janeiro, Brazil. The purpose of the conference was to discuss ways of combining increased protection of the environment with more effective economic development in less wealthy countries (Primack, 1998). One of the five major documents that the conference participants discussed and most eventually signed, is the Convention on Biodiversity that came into force at the end of 1993. This Convention was the first agreement that recognises the sovereign rights that states have over their own resources, but also the responsibilities they have to ensure that activities within their borders do not cause damage to other nations or international waters (Miller *et al.*, 1995). It has been ratified by almost 170 nations so far (Primack, 1998). Currently, the Convention on Biodiversity is the most important global framework for biodiversity conservation and for the incorporation of nature conservation goals into other policy areas (Delbaere, 1998).

According to the Convention, governments, with the co-operation of the United Nations, non-governmental organisations, the private sector and financial institutions, must conduct national assessments on the state of biodiversity and are required to prepare environmental assessments of proposed projects likely to have significant adverse impacts on biodiversity. Parties must promote the conservation of ecosystems, natural habitats and the maintenance of viable populations of species in their natural surroundings. Communities, including women, must be involved in conserving and managing ecosystems. Parties are also required to conduct long-term research into the importance of biodiversity for ecosystems that produce goods and environmental benefits. They must adopt national strategies to conserve and sustainably use biological diversity and make these parts of overall national development strategies. Furthermore, the Convention requires parties to encourage traditional methods of agriculture, agroforestry, forestry, and range and wildlife management, that use, maintain or increase biodiversity. It also requires the development of sustainable uses of biotechnology, and ways of safely and equitably transferring it, particularly to developing countries. The parties must implement fair and equitable sharing of benefits from the use of biological and genetic resources between the sources and users of these resources. The Convention recognises that indigenous people and their communities should share in the economic and commercial benefits, particularly when they have contributed their own local knowledge of species (Keating, 1993).

The success of the Convention will partially depend on the willingness of developed countries to provide financial support and technological transfer to poorer countries that are rich in biodiversity. Since protecting nature and conserving biodiversity yield significant global benefits, the developed countries have an important role to play (Munasinghe, 1992). Paragraph 4 of Article 20 of the Convention explicitly links effective implementation by developing countries to the fulfilment of these obligations by developed countries. It enables developing countries to meet the costs arising out of their obligation under the Convention (Barrett, 1995; Miller *et al.*, 1995).

In short, the Convention on Biological Diversity has three objectives: protecting biological diversity, using it sustainably and sharing the benefits of new products made with wild and domestic species. In addition, it poses a challenge to ecologists, economists and scholars from other disciplines to provide further insight into the fundamental economic and ecological dimensions of biodiversity (Barbier *et al.*, 1994).

A major source of funds for environmental protection related to the Convention on Biological Diversity is the Global Environmental Facility (GEF). The GEF was established by the United Nations and the World Bank in 1991 to transfer funds from rich to poor countries in return for conservation of biodiversity, reduced pollution in international waters, control of carbon dioxide emissions, and adoption of measures to combat deforestation and desertification (Turner *et al.*, 1994). In developing countries and Central and Eastern Europe, the GEF is the prime source of investment funds for biodiversity and nature protection (Delbaere, 1998). However, the lack of participation by community groups and government leaders has been identified as a major problem (Primack, 1998). Incentives to increase the participation in the programme of the GEF are therefore necessary. In addition to the GEF, countries can take advantage of the opportunities offered by Joint Implementation (JI). The concept of Joint Implementation is used to describe a wide range of possible arrangements between interests in two or more countries, leading to the implementation of co-operative development projects that seek to reduce or sequester greenhouse gas emissions.

The pace of tropical deforestation has accelerated since the late 1970s. This acceleration has raised global concerns because the vast majority of terrestrial species is found in rain forests, providing both regional and global ecological and economic benefits. Reforestation efforts may safeguard against the loss of economic benefits of natural tropical forests, such as supply of timber, watershed protection, carbon store and climate regulation. In addition, reforestation can reduce the costs of biodiversity loss resulting from deforestation (Burgess, 1995). At the same time, improving the economic conditions that tend to promote deforestation, such as market failures and macroeconomic conditions, has the potential to reduce clearing and degradation of tropical forests. Improving these conditions requires efforts at both national and international levels. With regard to the latter, the provision of debt relief in the form of debt for nature swaps and the provision of information are essential. Furthermore, environmental investment provided by programs such as the Tropical Forestry Action Plan can yield important results in terms of both improving rural standards of living and reducing deforestation. The Tropical Forestry Action Plan was initiated in 1985 by the Food and Agricultural Organization, the World Resource Institute, the World Bank and the United Nations Development Program to conserve and sustainably develop tropical forest resources on a long-term basis (Kahn, 1998).

International development agencies, such as the World Bank, now looks for new approaches to management of protected areas that incorporate local stakeholders into protection, benefit sharing and planning. The World Bank is in a good position to identify where the various cross-sectoral linkages and weaknesses are as well as to suggest financing mechanisms to bridge them. The World Bank emphasises that the environment must be considered to be a basic function and not, as in the past, a function of development (Drucker, 1998).

Agriculture is a major beneficiary from the protection of biological diversity and its life-support functions. It has also preserved some species which would otherwise have been driven to extinction by hunting or gathering (Tisdell, 1991). However, agriculture has been responsible for a direct negative impact on biodiversity at all levels and on both natural and domesticated diversity. Commercial agriculture has led to homogenization of the landscape and it can be considered as one of the most important causes of pollution through its use of pesticides and generation of chemical wastes (McNeely *et al.*, 1995). Stimulating sustainable agriculture production will tend to be supportive of biodiversity. Production subsidies to achieve domestic policy goals carry implications for the environment. Subsidy policies can distort the response of agricultural producers to market signals and prevent the efficient use and allocation of resources. In addition, subsidies related to production can contribute to more intensive (and polluting) agricultural production. Import barriers distort trade among countries. Closing off market access to developing countries capable of lower cost and lower intensity production prevents the efficient and environmentally-beneficial allocation and use of resources. So, in general, trade liberalisation and environmental protection are both necessary in governments' efforts to move to global sustainable development (OECD, 1994). Direct international transfers from North to South, which may reduce the dependency of a developing country on the exploitation of natural ecosystems for export earnings, such as forests, seem to be effective in promoting protection of the ecosystems. A large transfer to assist with sustainable agricultural management has the effect of 'freeing up' domestic financial resources for other purposes (Barbier and Rauscher, 1994).

Another facet of trade which is responsible for the decline of many species, is the often highly lucrative legal and illegal trade in wildlife (McNeely *et al.*, 1995; Primack, 1998). Such trade is characterised by its global status. Whereas major exporters are primarily found in the developing world, especially in the tropics, most major importers are developed countries and East Asia (Primack, 1998). The global status of current trade has driven certain rare or endangered species near to extinction through overexploitation. International treaties such as the Convention on International Trade in Endangered Species (CITES) have encouraged the development of illegal networks with low investment costs and potentially high profits (McNeely, 1995). A black market links poor local people, corrupt customs officials, dealers and wealthy buyers who fail to value the environment and its resources. Therefore, dealing with these illegal activities has become a task for international law enforcement agencies (Primack, 1998).

2.2 Dutch nature policies

Due to the intensive cultivation of the Dutch landscape, the amount of 'real' nature in the Netherlands is very limited. Mammals have been driven to extinction, primaeval forests have all been cleared and water has been drained and converted to farmland. Only about 10 percent of the area has hardly undergone any changes since 1840. Nevertheless, mankind and its activities did not only impoverish nature since an enrichment of biological diversity has occurred over time as a result of human-induced species invasions.[2] In addition, as agriculture has preserved some species that would otherwise have been driven to extinction by hunting or gathering, agricultural areas exhibited major ecological qualities. Nowadays, however, demographic, technological and economic developments have a negative influence on the quality of nature and landscape. Due to developments in methods of agriculture and pollution of the environment the ecological quality of agricultural areas deteriorates. Illustrative of the impoverishment of the Dutch nature is the fact that since 1950 500 out of 1400 species of higher plants have declined in number and more than 70 species have become extinct. In the same period, the number of summer birds has declined a third (Ministry of LNV, 1990).

Until the late 1980's, Dutch nature policies gave priority to the protection of a landscape with small-scale agriculture and high ecological values realised by extensive farming. In 1990, however, a more offensive policy was launched when the Dutch Nature Policy Plan (Ministry of LNV, 1990) was published. This plan emphasised the withdrawal of areas from agriculture and

[2] Despite some positive effects on biological diversity, the effect of invasive species in isolated ecosystems is often recognised as one of the most severe disruptions in the world's fauna and flora. Some nature reserves established to conserve native species have been heavily influenced by exotic species (McNeely *et al.*, 1995). The latter often prey on endemic species. In addition, endemic species suffer a reduction in fecundity and chance of survival or growth as a result of resource exploitation or interference by exotic species.

converting these former agricultural areas into natural areas. From that year onwards, Dutch nature policy goals were not only defined at the level of species and communities, but also at the level of populations and landscape. The Nature Policy Plan includes a description of the Ecological Network, a coherent spatial network of existing natural areas and natural areas that have yet to be developed. The need to maintain large natural areas, facilitate migration and restore habitats has prompted the government to develop such a network. It consists of core areas, nature development areas and linking zones. Core areas contain ecological values which are of national or international importance. They are at least 250 hectares in area. The maintenance and development of existing ecological qualities have priority in these areas. Influences that are negative from an ecological point of view will be kept out. Nature development areas have the potential to be transformed into new core areas or to contribute to the enlargement of existing core areas. Much emphasis is placed on redesigning the landscape together with an alteration of land use. Linking zones make the dispersion, migration and exchange of species between core areas and natural development areas possible. They are considered as an essential part of the Ecological Network and can have the shape of both (narrow) corridors and stepping stones.

In the Nature Policy Plan, large-scale natural processes are recognised as the basic principle of Dutch nature policies. Diversity and 'naturalness' – as few human influences as possible – are the most important criteria for a qualitative valuation of nature and landscape. While diversity as a criterion is translated into the intention of the government to prevent species losses, 'naturalness' is measured by the size and the undisturbedness of natural areas. In practice, however, especially undisturbedness is difficult and perhaps even impossible to quantify. To overcome this difficulty, the applied management of a natural area can be used as an indication of naturalness. With regard to diversity, special attention is paid to the so-called *itz*-species. These are species of international importance (i-species) that show a downward trend in their existence since 1950 (t-species) and are rare (z-species). Species that meet two or three of these criteria are called 'goal species'. The Nature Policy Plan gives no ecological arguments for the protection of these goal species (Eijsackers, 1996). Together with a description of the applied management, goal species determine 'types of nature goals'. There are 132 types of nature goals and each of them is associated with a particular set of environmental conditions (e.g., the availability of nutrients, the pH-values of the soil, the average level of groundwater in the spring).

The environment in general and environmental policies in particular creates specific circumstances for nature. The influence of substances on nature, especially the negative impact of toxic materials, has received much attention during the last decades. Nature, in turn, influences the environment. During the process of nature development, new environments are brought into existence. Natural processes create and maintain environmental conditions, not only in natural systems but also in systems that are influenced by mankind. In Dutch policy making, nature is, on the whole, subordinate to the goals of environmental policies, water management policies and physical planning (spatial policies). However, nature is the source of these three areas of policy making. Many materials – i.e. energy and biological resources – emanate from nature. In a report from 1995 by the Ministry of Agriculture, Nature conservation and Fisheries, the relation of nature policies to environmental policies is recognised (Ministry of LNV, 1995). This report states that the sustainable use of biological resources should steer both nature policies and environmental policies. The report demands attention to (i) an evaluation of the resistance of nature to polluting circumstances; (ii) actions with regard to postponed and indirect effects of toxic materials; and (iii) a specification of the species at which actions of recovery should be aimed (Eijsackers, 1996).

In 1998, nature protection received a new impulse as the Ministry of Agriculture, Nature conservation and Fisheries launched a new approach of managing nature and landscape in the Netherlands. In this approach, titled 'Programme Management', the quality of nature and landscape occupies centre stage. The outlines of nature policies – as described in the Nature Policy Plan – remain the same, but by changing the implementation of the policies, the government tries to increase the participation of citizens in maintaining and developing nature. For example, the government extends the possibilities covering eligibility for subsidies to nature development, both within and outside the Ecological Network. The amount of subsidy depends on the activities needed to achieve certain nature goals and is thus linked to the results of nature development. These nature goals are clearly defined and predetermined. They usually do not consist only of an enumeration of desired plants and animals, but also of required water level and relief of the soil. The 'Programme Management' was introduced in 1998. Nevertheless, it will take several years to be completely implemented.

3. ECOLOGICAL THEORY

3.1 Ecological theory

Ecology deals with the relationship between organisms and their biotic (living) and abiotic (non-living) environment (van der Ploeg, 1982; Begon *et al.*, 1990). The term 'environment' refers to all entities – whether living or not – which surround a living entity (van der Ploeg, 1982). Ecology has been developed from the eighteenth century onwards; however, it was not until the beginning of the twentieth century that formal tools for the measurement and modelling of the relationships between organisms and their environment were developed.

Within the branch of ecology, there are three levels of concern, namely the individual organism, the population (including all the individuals of the same species) and the community (the species that exist together in space and time). However, no ecological system, whether individual, population or community, can be studied in isolation from the environment in which it exists. Therefore, another category of ecological study has been set apart: the ecosystem. An ecosystem encompasses the biological community together with its physical environment (Begon *et al.*, 1990). Tansley (1935) introduced the term 'ecosystem' to refer to the basic units of nature on the earth's surface. So, everything in nature happens within ecosystems (Aber and Melillo, 1991). An 'ecosystem' can be defined as all the individuals, species and populations in a spatially explicit unit of the Earth, the interaction among them, and the interaction between the organisms and the abiotic environment (Tansley, 1935; Odum, 1983; Udo de Haes and Klijn, 1994). Ecosystems exist in many different forms and sizes and represent both abstract units (ecosystem types) and concrete recognisable objects (Udo de Haes and Klijn, 1994; Haber, 1994).[3] The world encompasses an immense range of terrestrial and aquatic ecosystems, from polar ice caps to forests. In general, attempts to classify ecosystems are based on biomes, that is the largest ecological unit recognised by the structure of its flora and fauna. Terrestrial biomes include arctic and alpine tundra, boreal coniferous forests, temperate deciduous forests, temperate grasslands, tropical savannah and grassland, Mediterranean vegetation or chaparal, deserts, semi-evergreen

[3] In order to avoid confusion it is suggested in the literature that a distinction should be made between concrete ecosystems and abstract ecosystems. Therefore, it is convenient to use the word ecosystem only in a functional way as the processes which connect the components. The concrete and tangible ecosystem is often called an ecotope (van der Ploeg, 1982; Haber, 1994). Ecotopes are defined as "... a homogeneous ecological unit, the spatial expression of ecosystems predominantly determined by their structural characteristics." (Udo de Haes and Klijn, 1994, p. 12). Thus, an ecotope is an ecosystem, but an ecosystem of a certain size and with homogeneity that is basically defined by abiotic criteria. Ecotope is often used to refer to the basic spatial unit for ecosystem classification and as the smallest part of the landscape (Runhaar and Udo de Haes, 1994).

tropical forests and evergreen tropical rainforests. The biomes include on the aquatic side open oceans, deep seas, continental shelves including coral reefs, upwelling areas and estuaries, and the freshwater ecosystems of lakes and ponds, rivers and streams, and wetlands (Folke, 1999). Nevertheless, there are several other ways of classifying ecosystems (see, for example, Bisby, 1995; Stiling, 1999).

All ecosystems undergo various kinds of changes (van der Ploeg, 1982). This process of change is called succession, which is defined as "... the non-seasonal, directional and continuous pattern of colonization and extinction on a site by species population." (Begon *et al.*, 1990, p. 628). For example, wetlands are filled up with silt, become a swamp and subsequently a grassland or even a forest. The presumed end point of a successional sequence is called a 'climax'. It is the final stage in succession. In practice, however, it is very difficult to identify a community that is at maximum stability under the present environmental conditions. Due to high probabilities of certain disturbances, such as fire, a process of succession may never go to completion. For example, some forest communities in northern temperate regions are still recovering from the last glacial period (Begon *et al.*, 1990).

Intimately linked to the concept of climax is the concept of the 'steady state' of systems, which can be defined as a situation in which a constant pattern of flows, cycles, storages and structures prevails (Odum, 1971; van der Ploeg, 1982). The term is frequently used in ecological theory, especially with regard to structure and functions of ecosystems. Although particularly climax ecosystems show stability over long periods, the concept of climax is not identical to the concept of steady state. Due to rigorous or unfavourable environmental factors, the climax community may be very unstable (van der Ploeg, 1982). In an unfavourable environment, evolutionary changes within systems become important for the survival, growth and reproduction of species. Evolution by natural selection is said to have occurred if some individuals produce more offspring than others, leading to a change in the heritable characteristics of a population from generation to generation (Begon *et al.*, 1990).

Organisms are thus affected by the conditions in which they live and by the resources which they obtain. As environmental conditions on Earth vary, organisms do not live scattered all over the world. A group of individuals of one species in a certain area is called a population (van der Ploeg, 1982; Begon *et al.*, 1990). Each organism belongs, for at least part of its life, to a population composed of individuals of its own species. Population sizes often fluctuate with time because of natality, mortality and the dispersal of organisms. Each individual in the population will contribute to this fluctuation. A distinction is made between at least two basic models of population increase. The first model is the J-shaped growth form, in which the population increases exponentially and then is stopped abruptly because one or more factors have become limited. The second model is the so-called sigmoid or S-shaped growth form. In this model the initial increase is slow because the population is small. But with the lapse of time, the population is getting larger which results in a more rapidly increase in size. Then, as environmental factors become more and more limiting, population growth decreases. Finally, the population reaches its carrying capacity at which the birth and immigration rates equal the death and emigration rates. This density is called carrying capacity because it relates to the maximum population size which the environment can sustain (or carry) without a tendency to either increase or decrease. In many natural and experimental situations, a sigmoid shaped population growth can be detected (Begon *et al.*, 1990).

3.2 Characteristics of ecosystems

Holling (1986; 1987) has described ecosystem behaviour in terms of the dynamic sequential interaction between four basic system functions: exploitation, conservation, release and reorganisation. Many ecosystems are subjected to regular or irregular disturbances that are severe enough to kill communities. During exploitation, a rapid colonisation of recently disturbed areas finds place. Conservation in this context refers to the resource accumulation that builds and stores increasingly complex structures, such as energy and material. Release or 'creative destruction' – a term borrowed from the economist Schumpeter, as noted by Elliot (1980) – occurs when an abrupt change caused by external disturbance, such as fire, disease or grazing pressure, releases energy and material that have accumulated during the conservation phase. In other words, the stored structures are then suddenly released and the tight organisation is lost. Creative destruction generates opportunities for reorganisation, which can be defined as the mobilisation of released materials that allow the next phase of exploitation (Perrings *et al.*, 1995; Costanza *et al.*, 1997a).

Solar energy is the driving force of any ecosystem as it enables the cyclic use of materials and compounds required for the self-organisation and self-maintenance of the system (Costanza *et al.*, 1997a). An ecosystem under stress seemingly keeps much of its function even though the composition of materials, i.e. species, changes. However, it is the self-organising ability of an ecosystem, which determines its capacity to respond to external stress. Since mankind is disturbing nature at an increasing rate, it is essential to know how ecosystems respond to such an external stress and how they are likely to respond in future. The stability of an ecosystem measures its sensitivity to disturbance (Begon *et al.* 1990). In this regard, a distinction should be made between stability and resilience (see Figure 1). Whereas stability and productivity of an ecosystem are determined by the exploitation rate and conservation sequence, resilience depends on the system's capability to reorganise after creative destruction and its capacity to maintain its functions and structure (Perrings *et al.,* 1995).

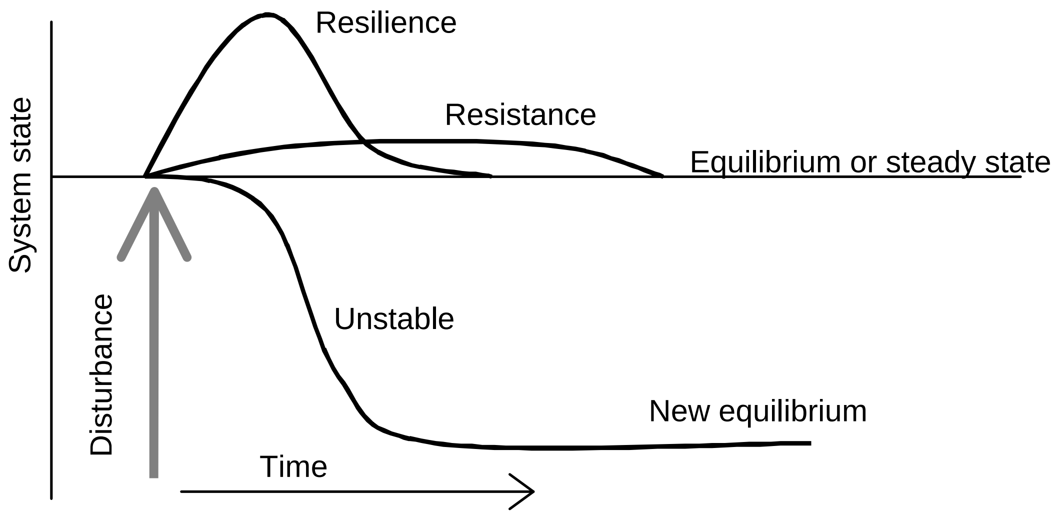

Figure 1. Source: *Aber and Melillo (1991), p. 67 Figure 5.3. Ecosystem responses to disturbance: resistance, resilience and unstable.*

The concept of resilience has two main variants (Perrings, 1998). One approach – and according to Holling *et al.* (1995), the more traditional – concentrates on resistance to disturbance and speed of return to (globally) stable equilibria (Pimm, 1984; Perrings, 1998). A second approach refers to the magnitude of disturbance that can be absorbed before an ecosystem is displaced from one state to another (Holling, 1973; Holling *et al.*, 1995; Perrings *et al.*, 1995). This approach assumes that ecosystems are characterised by multiple locally stable equilibria. If a certain disturbance occurs and – despite this disturbance – an ecosystem does not change from one stability domain to another, the system may be said to be resilient with respect to that perturbation (Perrings, 1998). An ecosystem which has a stable equilibrium within a narrow range of environmental conditions, or in other words, an ecosystem that only can absorb small perturbations without changing from one equilibrium to another, is said to be fragile. Figure 2 provides an illustration of fragility and resilience.

Fragile *Resilient*

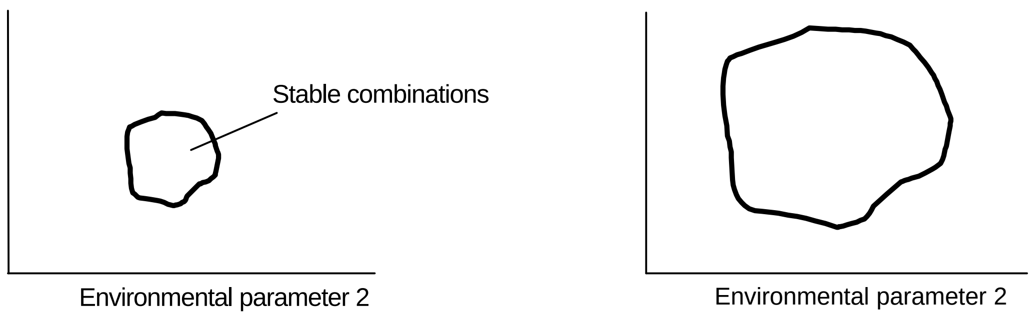

Figure 2. Source: *Begon* et al. *(1990), p. 793 Figure 23.1. The difference between a fragile and resilient ecosystem, illustrated here in a figurative way.*

Changes in stability domains are often induced by human activity and may occur in many ecosystems, such as savannah ecosystems, coral reef systems and shallow lakes (Costanza *et al.*, 1997a). Human activities affect the ecosystem's characteristics, and vice versa. Although human activities influence different ecosystems in different ways, for every system an increase in stress or reduction in the resilience of a system makes it more susceptible to exogenous shocks or changes in environmental conditions (Udo de Haes and Klijn, 1994; Perrings, 1998). There are many examples of managed ecosystems that share this same feature of loss of resilience, followed by a shift into an irreversible state. Loss of resilience in a human-dominated ecosystem is caused by a reduction of the natural variability of variables and processes that control behaviour. This results in an ecosystem that becomes more spatially uniform, less functionally diverse and more sensitive to perturbations that otherwise could have been absorbed (Holling *et al.*, 1995).

Resilience is one of the operational definitions of ecosystem health. Healthy ecosystems are considered to have the ability to absorb disturbances and to recover from stress. Various other concepts and dimensions of ecosystem health have been proposed (Costanza, 1992): (i) health as homeostasis: any and all changes in an ecosystem represent a decrease in health; (ii) health as the absence of disease; (iii) health as diversity or complexity; (iv) health as vigour or scope for growth; and (v) health as balance between system components. Health is most easily defined as the absence of disease. But applying this definition of health implies defining disease. From a medicinal point of view, Costanza (1992, p. 245) defines disease as follows: "... a particular destructive process in the body, with a specific cause and characteristic symptoms; a specific illness, ailment, or malady." In the context of an ecosystem, disease can be thought of as a stress to the system with certain negative effects. However, whereas medical practitioners have a defined process for assessing health by using reference data and their knowledge of known diseases, ecologists lack this experience of known diseases or stresses with associated symptoms and indicators. In addition, doctors usually concentrate their efforts on the health of a given patient and all their decisions will be made from the perspective of that patient. Ecosystems and their health on the other hand, exist on many levels and can be described and analysed on many scales (Haskell *et al.*, 1992). Or, as stated by Norton (1991), what unit should be the 'patient' in ecosystem management? Due to the complex structure of ecosystems, a cut-and-dried answer to this question can not be given. Despite some discrepancies between medicine and ecosystem treatment, the health analogy between the two disciplines are in several ways illustrating. First, the object of ecosystem management is a dynamic and changing system rather than a static unchanging machine. Second, both ecosystems and human bodies are complex systems so that affecting parts of the 'body' leads to a certain influence on the larger whole of which those parts are functioning elements. Third, as ecosystems are more like organisms than they are like machines, environmental management acts to achieve a worthy goal, namely to protect nature from a disease (Norton, 1991).

The idea of environmental managers and conservation biologists is to protect and restore health to ecological processes at all levels (Haskell *et al.*, 1992). Protecting the health of ecological systems is needed because nature is more profoundly a set of related processes than a collection of species. Populations of species are interdependent; they have co-evolved in ecosystems on which they depend. More specifically, a species may depend on just one other species for food, or it may depend on an entire complex of interrelated species (Norton, 1988). However, the concept of health is here understood as a normative concept: every time when the concept is used, it implies that it is known what that system should look like (Norton, 1992). The concept reveals a desired endpoint of environmental management. Without an adequate identification of the relevant indicators of health – such as a species or a group of species – and an adequate identification of important objectives of health – such as stability – effective environmental management is unlikely (Costanza, 1992).

Change and adaptability play not only a major role in the resilience of an ecosystem, but they are also the essence of ecological sustainability. Biologically, sustainability means avoiding extinction and living to survive and reproduce (Costanza *et al.*, 1997a). Important to the sustainability of an ecological system is its ability to co-evolve with its environment. Whether an ecosystem will co-evolve with its environment depends on the resilience of the system, because resilience determines the range of options (to evolve in response to disturbances) available to *future* generations of organisms within the system. However, the range of evolutionary choices available to *current* generations of organisms influences the resilience of the ecosystem (Pearce and Perrings, 1995). After all, the interaction between species and their natural environment contributes to the generation of new biological characteristics and functions. These may provide a means of minimising the risks of fluctuating environmental conditions and disturbances, which eventually may result in a higher resilience of the ecosystem.

A minimum level of biological diversity is a necessary condition for the resilience of ecosystems (Pearce and Perrings, 1995). Biological diversity, or biodiversity for short, is an umbrella term used to describe the extent of diversity of nature at all levels of organisation, ranging from the genetic, population and species levels to the community and ecosystem levels (Daily and Ehrlich, 1995). Biodiversity protects the system from irreversible and unpredictable changes. It should be noted however, that although system resilience depends on biodiversity, more species do not always imply greater resilience. In other words, there is no well-defined general relationship between species diversity and ecosystem resilience (Perrings *et al.*, 1995). To prevent a catastrophic change or a fundamental reorganisation in an ecosystem, losses of biodiversity need to be minimised. Biodiversity should be protected from further erosion or, at least, the rate of erosion of biodiversity should be slowed down (Pearce and Moran, 1994). It needs to be stated, however, that protection is not merely an objective for idealist preservationists, as it serves also purposes of value to society. The value of nature is one of the topics that we discuss in the next section.

4. ECONOMIC ASPECTS OF NATURE POLICY

4.1 Interactions between the economy and nature

Mankind has substantially changed the natural environment and people will most likely alter it further in future times. Influences on nature can have major implications for economic welfare and development, society and the quality of life since mankind depends on nature for continuing economic productivity, consumption, welfare and existence (Tisdell, 1991). At this point, economics and ecology become intertwined.

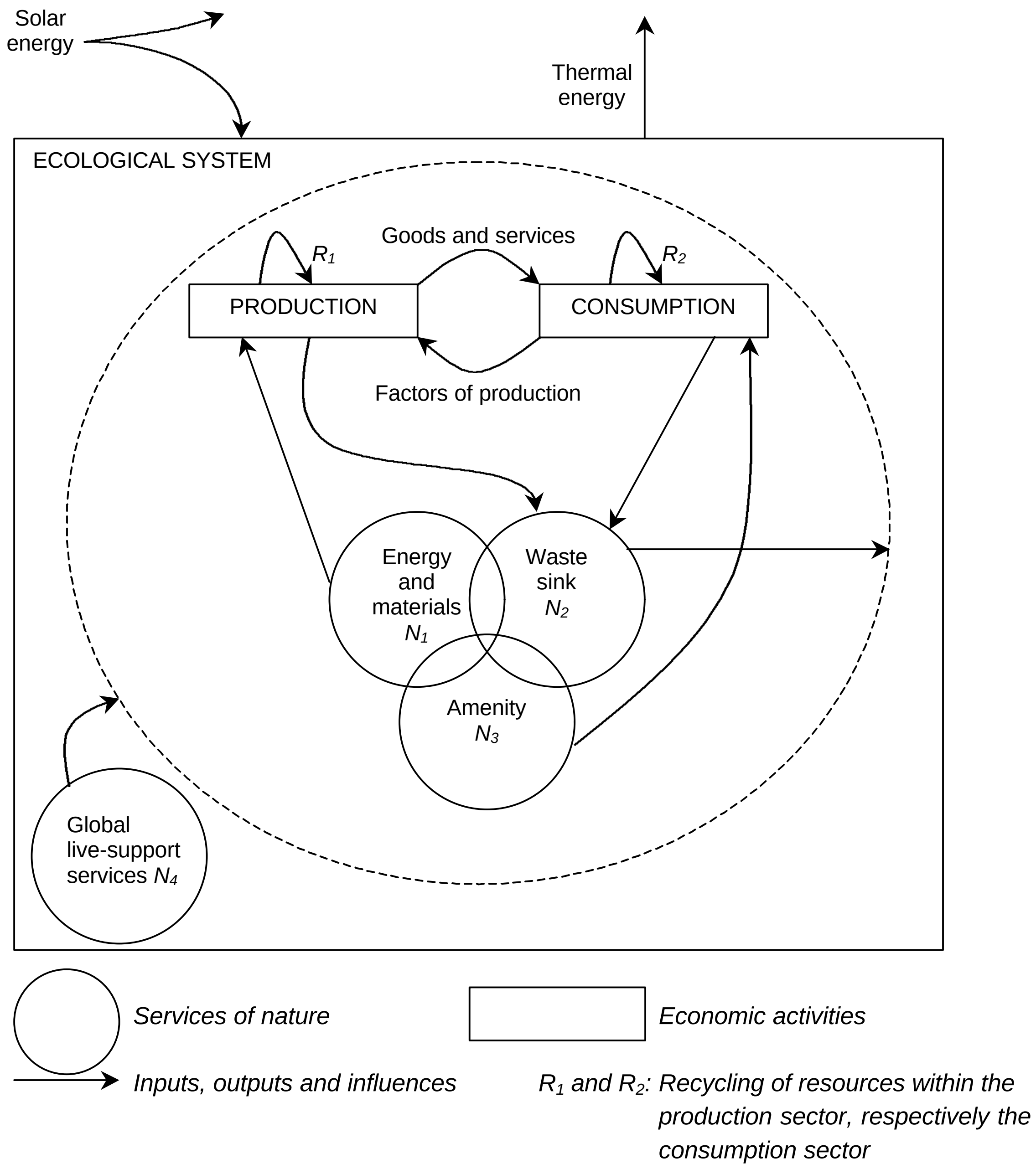

Figure 2. Source: *Adapted from Perrings* et al. *(1992), p. 201 Figure 1 and Hanley* et al. *(1997), p. 3 Figure 1.1. Economy-nature linkages.*

Ecosystems and biodiversity provide four types of services to people. Nature's first service is as a supplier of resources and energy inputs. Living organisms have specific properties that are useful to economic activities and therefore they generate direct welfare benefits. The second service of nature is as a receptor for waste products; nature recycles human wastes from production and consumption. A third service is as a supplier of amenity, educational, religious and cultural values to individuals and society. Lastly, nature serves life-support functions to the economic process, such as the maintenance of the quality of atmosphere, the maintenance of climate and temperature and recycling of water and nutrients. The interactions between the economy and nature are continually changing, i.e. they are dynamic (Hanley *et al.*, 1997).

The interactions between nature and the economy are visualised in Figure 2. The economy is shown as a simplified circular flow of goods and resources. It consists of only two activities, namely production and consumption. Producers supply goods and services to households, which in turn supply producers with labour and other resources – such as capital and land – required for production. The four types of natural services are shown as the three overlapping circles N_1, N_2 and N_3 and the all-encompassing boundary labelled N_4. Because there are conflicts in the use of nature, the three interlinked circles are shown as overlapping. So, the different types of natural services are not independent of one another. More of one natural service can mean less of others. For producing outputs, the production sector extracts energy resources (such as oil) and material resources (such as iron ore) from the environment. These natural and energy resources are used as inputs for the production of marketed goods.

As already mentioned, besides the tangible benefits nature generates also many intangible benefits to human kind. Societies and individuals place different values on these intangible benefits based on their specific social, cultural and religious background. For example, the attitude of Buddhists towards the biophysical environment is nurtured and determined by the Buddhist philosophy of compassion with and respect for all forms of life. This attitude induces environment and biodiversity protection and has economic or utilisation implications (van Ierland, 1998). The ethical, non-use values of nature are in Figure 2 indicated as the arrow from amenity (N_3) to the sector of consumption.

The inputs that are used in the production sector are transformed into two kinds of outputs. First, some outputs are supplied to consumers; they are summarised as goods and services. Second, at each stage of the production process, wastes arise. However, wastes do not result from production only, because also consumption creates wastes by generating sewage, litter and municipal refuse. Within both the consumption and the production process a recycling of resources takes place, which is shown by the two loops R_1 and R_2. Finally, the line between N_2 and N_4 indicates that emissions can affect global life-support services, the fourth role of nature in the economy. (Hanley *et al.*, 1997; Wills, 1997). The whole system is driven by solar radiation. The concentrated solar energy is used by natural systems and by humans.

4.2 The economic valuation of nature

Natural goods and services often have no price tag because they are not fully captured in markets. Nevertheless, it can be useful to estimate the value of natural goods and services. The justification for a valuation of nature lies in the fact that it offers a method of measuring the ecological consequences of economic activities. To allow for a proper determination of the value of nature, it is necessary to express the benefits of nature in such a way that all possible benefits that might derive from nature are taken into account (Turner *et al.*, 1998). Valuation, then, provides a tool to assist in environmental decision-making. It can make a worthwhile contribution to environmental policy discussions and decisions.

The often-used concept of total economic value is an aggregate measure of nature valuation (Turner *et al.*, 1994). It consists of two main elements. One element contains the services provided in the course of the actual use of the natural environment in consumption and production activities. This is referred to as use value. In addition to the use value, there is a non-use value of natural goods and services. Non-use values involve no tangible interaction between the natural resource and the people who benefit from it. That is, people who do not intend to make use of a natural resource would nevertheless consider it a loss if this resource was to disappear (Perrings, 1995).

For use values a separation is made between direct use values and indirect use values. Many (but not all) direct uses can be valued in markets. For example, the direct use values of tropical forests are based, among others, on their function as supplier of plant genetic material that can be used to produce medicines. Ecotourism is another example of a direct use value. However, whereas medicinal plants can be marketed, resulting in market prices that signal the (true) scarcity of the asset, the direct use of ecotourism is unpriced (Turner *et al.*, 1994). The indirect use values indicate the indirect support to economic activity by natural goods and services. Returning to the example of the tropical forests, the indirect use values – or life support benefits – include carbon fixation, water purification, watershed protection, soil formation and the decomposition and assimilation of wastes (Wills, 1997).

Depending on exact definitions, non-use values may include all of the following: option values, quasi option values, bequest values, philanthropic and existence values.[4] Turner *et al.* (1994; 1998), however, classify option values as use values, referring to the possibility of future use. In the present, option value is a non-use value (Wills, 1997). In the economic literature it has been suggested that option value represents a difference between *ex ante* and *ex post* valuation, where the terms *ex ante* and *ex post* refer to the amount of information that is available. *Ex ante* relates to the situation where the state of the world is still unknown, while *ex post* refers to the situation after the state has been revealed (Bishop, 1982; Smith, 1983; Freeman III, 1984; Ready, 1995). Quasi-option value, a concept originally introduced by Arrow and Fisher (1974), is the expected benefit of awaiting improved information derived from delaying exploitation. It suggests a value attached to protection given the expectation of the growth of knowledge (Henry, 1974). Bequest value is a willingness to pay to keep a natural asset intact for the benefit of one's descendants, or more generally, future generations. Existence value is a value people attach to a natural asset unrelated to their actual or potential use. Existence value involves a subjective valuation as it is based on the satisfaction which individuals experience from knowing that a certain natural asset exist, for themselves and for others, without being used now or in the future (Barbier, 1995; Wills, 1997).

Despite the use of the concept of total economic value, it is often argued that economists have up to now not been very successful in capturing all ecosystem functions in economic valuation. The total economic value fails to reflect the life support service of an ecosystem, which is essentially the existence, functional operation and maintenance of the entire ecosystem that are behind the assigned values of natural assets (Barbier, 1995). Natural goods and services depend on the life support service provided by an ecosystem. In other words, the existence, operation and maintenance of an ecosystem are necessary before natural assets can be utilised and valued by humans. The total economic value therefore underestimates the true value of an ecosystem. On the other hand, going beyond the total economic value to measure the extra value is extremely difficult or even impossible.

[4] A thorough discussion of these different values can be found in Turner *et al.* (1994, pp. 108-128).

4.3 *Externalities, public goods and property rights*

As already stated, whereas some direct use of a natural asset – such as timber and medicinal plants – can be provided for and valued in a market, the indirect use benefits and the non-use benefits of an asset have no market price at all. Thus many ecosystem functions result in goods and services which are not traded in markets. In economic terms, many natural assets may be considered to be a public or collective good, or may possess some features associated with such goods. Pure public goods have the characteristics of joint consumption and non-exclusion. So, the good can be consumed or enjoyed by one person without its available supply being diminished. Non-exclusion indicates that one person (or a public authority) can not prevent or exclude another person from consuming or enjoying the good once it is supplied (Tisdell, 1991; Turner *et al*., 1994). Due to the non-exclusion attribute, a pure collective good cannot be marketed. Because it is impossible or at least very costly to deny access to an environmental asset, markets fail to allocate resources with public good characteristics efficiently. Prices do not signal the true scarcity of the asset (Hanley *et al.*, 1997). It should be noted, however, that most of what we think of as public goods are not pure public goods, as natural resources have some degree of exhaustibility and excludability. An example of an environmental resource that has pure public good characteristics is climate. After all, all people in a geographic location experience the same climate and no one can be excluded from experiencing it (Kahn, 1998).

The majority of natural assets fall into a category in which markets values are not available. The crucial feature of an externality is that there are goods people care about that are not sold in markets (Turner *et al.*, 1994). Externalities are effects on others that are not properly accounted for by the decision-maker. Or more precisely, as stated by Baumol and Oates (1988, p. 17), an externality is present whenever some individual's utility or production relationships include non-monetary variables, whose values are chosen by other persons, firms or governments without paying particular attention to the effects on the respective individual's welfare. Externalities may be either positive or negative. Although it is possible that externalities take the form of private goods, most externalities associated with the use of environmental assets are public in nature. Externalities with public good features are called non-depletable externalities and are characterised by non-rivalry in consumption. For example, one person's consumption of polluted drinking water does not reduce the amount of the water pollution to which other people are exposed (Perman *et al.*, 1996; Kahn, 1998).

Many natural assets are not privately owned but are characterised by either common property rights or open access. Hence, ownership is not clear and well defined or does not exist. Consequently, communal property regimes and open access may risk conservation of the asset, but do not necessarily lead to overuse (Turner *et al.*, 1994). Schmid (1995, p. 46) defines property rights as "... sets of ordered relationships among people that define their opportunities, their exposure to the acts of others, their privileges, and their responsibilities..." A private property right exists if a right is exclusive to one person or corporation. In these circumstances, markets will tend to develop naturally. Biodiversity and ecosystems, however, are public goods and as mentioned earlier, cannot be optimally provided through markets. This non-existence of markets reflects the failure or inability of institutions to establish well-defined property rights (Perrings, 1995).

The lack of property rights is also one of the important reasons for the existence of externalities. For instance, as no one has property rights to clean air, a person who suffers from air pollution has no clear legal rights to prevent someone from polluting the air or to claim compensation from someone who does pollute the air. But even if property rights exist, there may be a lack of ability to enforce them, which may result in an inefficient and uncontrolled use of the natural asset (Kahn, 1998).

4.4 Economic development versus nature protection

Nature conservation entails the creation of opportunity costs, defined as foregone benefits from possible alternative uses of nature. Choosing one of these alternatives foregoes benefits that would otherwise be derived from nature conservation. Cost-benefit analysis seeks to estimate conservation benefits in a way that makes them comparable with the returns derived from economic development. It offers a method to aid decision-makers in quantifying and evaluating projects from the point of view of the community. The technique incorporates clear principles for valuation of benefits and costs in monetary terms. The cost-benefit criterion may need to be modified as policy makers introduce, or respond to, concerns other than economic efficiency, such as equity, employment and nature conservation (Turner *et al.*, 1998).

Traditionally, evaluating the costs and benefits of a proposed project that affects nature and environment is based on the assumption that the preservation alternative entails neither a cost nor a benefit. From this traditional point of view, the evaluation of a project is all about economic development and the question whether the flow of benefits derived from development exceeds the flow of development costs. Krutilla and Fisher (1975), however, have argued that development benefits will fall over time. They proposed a new approach to wilderness preservation through cost-benefit analysis. The fundamental point of this new approach is its rejection of the view that the profitability of a project is an adequate criterion for the acceptability of the project when it destroys ecological values. The Krutilla-Fisher approach explicitly recognises asymmetric growth rates in development and preservation benefits, based on which forgone preservation benefits are treated as part of the costs of economic development (Porter, 1982; Hanley and Spash, 1993).

Because society is deemed to be uncertain about both the biological consequences of current actions and the future costs of current environmental degradation, cost-benefit analysis may result in a spurious precision of its calculations. In addition, cost-benefit calculations are considered to be unfair to future generations because most of the benefits of nature use are received by the present generation, while many of the costs will fall on future generations. Future generations have no voice in the matter and are not necessarily compensated for reductions in their endowment of nature (Wills, 1997). Therefore, there are good reasons to impose a safe minimum standard of conservation unless the costs of so doing are intolerably high (Randall and Farmer, 1995). A 'safe minimum standards' approach to environmental protection represents a supplement to cost-benefit analysis, which places greater emphasis on the environmental protection wherever thresholds of irreversible damage are threatened (Crowards, 1998). The safe minimum standard defines the level of preservation that ensures survival (Randall, 1988). Advocates of safe minimum standard approaches to nature conservation believe that uncertainties about the risks of environmental damage, and the irreversible nature of some environmental damage, are so great that cost-benefit analysis is inadequate as a technique for determining the conservation of species. The safe minimum standard approach implies a conservative approach to risk bearing. In effect, deciding to conserve today is shown to be the risk-minimising way to proceed given the presence of uncertainty about the consequence of nature loss. In other words, nature conservation can minimise the maximum loss to society (Tisdell, 1991; Hanley *et al.*, 1997). Consequently, in comparison with a cost-benefit analysis the use of a safe minimum standard tends to favour the conservation of living organisms. In fact, due to the absence of scientific certainty about the consequences of using biological resources, a safe minimum approach shifts the burden of proof from those who wish to conserve to those who wish to develop (Randall and Farmer, 1995). Going below the safe minimum standard should only be allowed if society believes the opportunity costs of preserving minimum standard to be unacceptably high. However, how should 'unacceptably high' opportunity costs of maintaining the standard be identified? Another limitation of the safe minimum standard approach is that it treats one gene, species or ecosystem as being just as good as another, so that the priorities for nature conservation depend solely on the costs of conservation. The safe minimum standard approach thus ignores the fact that there may be some scientific and value information available about the benefits (Wills, 1997).

4.5 Economics of biodiversity[5]

From an economic point of view, biodiversity is important for several reasons. First, as already mentioned, biodiversity promotes ecosystem stability and thus also promotes all the services derived from ecosystems. Furthermore, individual plants and animals have a value because they can be used to produce economic goods, both directly (e.g. fruits or nuts) and indirectly (species as a direct source of natural chemicals and compounds). Finally, the genes of species may be a source of genetic information. This information can, among other things, be used to create new (varieties of) plants or animals through genetic engineering (Kahn, 1998).

A variety of arguments is used to explain why depletion of biodiversity occurs. The most important argument relates to intertemporal preferences and discounting. Exploiting biodiversity involves a trade-off between present benefits and future costs that depends on how the latter are discounted relative to the former (Hanemann, 1988). Since biodiversity cannot be recreated in all of their essential features by humankind, its losses being incurred are irreversible. Once gone, we cannot reproduce species (Turner *et al.*, 1994). Van Kooten (1993) distinguishes two dimensions of irreversibility, namely a biophysical and an economic one. The first implies that some environments can never be restored to their original state once economic development has occurred. Economic irreversibility occurs when the costs of restoring an environment are higher than the benefits of restoration. The costs of restoring an environment will increase if economic development of this environment goes ahead. This also provides clues for finding policy solutions to prevent biodiversity loss.

Assigning a monetary value to the benefits of, or avoided damages, from protecting biodiversity can be done by a number of measurement techniques, based on either observed market behaviour (or revealed preferences) or on stated preferences. Approaches based on revealed preferences seek to recover an explicit relationship between individuals' willingness to pay for environmental quality and the demand for a market good. This category of methods estimates a value for a non-market-good by observing individuals' behaviour with regard to market goods and services, which are related to the environmental good or service of interest (Shechter, 1999). Direct methods involve monetary valuation of utility. These approaches consider an improvement of the environment and, through appropriately constructed questionnaires, they seek directly to measure the value of these improvements (Pearce and Turner, 1990; Perrings, 1995).

The travel cost method (TCM), which is a revealed preference method, is one of the earliest valuation methods employed by environmental economists (Clawson and Knetsch, 1966). The method is especially valid for assessing recreational values and to that end, it has been widely used in both the U.S.A. and the U.K. (Perrings, 1995). The underlying assumption of the TCM is that the incurred costs of visiting a national park, nature reserve, open space or any other recreational site are directly related to the benefits individuals derive from the amenities within the area, such as hiking, camping, fishing, swimming et cetera. Basically, the method involves using the value of time spent in travelling, the cost of travel (e.g., petrol costs) and entrance and other site fees as a proxy for computing the demand price, or value, of the environmental resource. So, if travelling to and entering a particular site becomes so expensive that no one decides to go there,

[5] Introducing analytical frameworks that represent a useful way of thinking about the economics of biodiversity through the medium of abstract mathematical models can, among others, be found in Munro (1997) and Weitzman (1998).

the value of the site, or the price the public would have been willing to pay to secure this form of land use, is zero (Perrings, 1995; Shechter, 1999).

The hedonic pricing method (HPM) derives the value of environmental amenities, such as pollution and noise level, from actual market prices of some private goods. Just like the travel cost method, the HPM is based on observed behaviour (van Ierland *et al.*, 1998). By far the most common application of HPM is to the real estate market. House prices are affected by many factors, such as number of rooms, the size of the garden, but also the environmental quality of the surroundings. If the non-environmental factors can be controlled for, then the remaining differences in real estate prices are expected to be the result of environmental differences (Turner *et al.*, 1994).

In the absence of appropriate data or interdependent market goods, applying an indirect method to value the benefits of biodiversity protection is either not possible or will lead to spurious results. Direct methods bypass the need to refer to market prices by asking people directly what their willingness to pay for a change in environmental quality (e.g., forest conservation or the presence of certain species in nature) is. The contingent valuation method (CVM) uses such a direct approach. It basically invokes a framework of a contingent (or hypothetical) market, used to indicate what individuals are willing to pay for a benefit or what they are willing to accept by way of compensation to tolerate a cost (Pearce and Turner, 1990; Shechter, 1999).

We have mentioned above several methods that attempt to measure the value of non-market environmental goods and services, such as biodiversity, in money units. Despite the common yardstick in the benefit valuation methods presented, none of them are panaceas; rather, each has its advantages and disadvantages. However, since nature policies entail costs, decision making by policymakers and the public at large regarding nature is supported by information on the expected benefits as the rationale for spending on such policies.

5. FURTHER OPTIONS FOR ECONOMICS-ECOLOGY INTEGRATION

In the previous sections, the initial impetus was given to the integration of economics and ecology. In this section, however, more creativity is practised in looking for new opportunities for the integration of both disciplines. Emphasising the importance of a multidisciplinary approach to study nature policies means that integrated ecological-economic frameworks should include not only the measurement of total economic costs and benefits from protected areas, but also the ecological criteria for nature protection. For instance, with regard to the Krutilla-Fisher approach to cost-benefit analysis, further adjustments can be undertaken by adding ecological constraints to the framework, reflecting ecological thresholds and the relationship between ecological stability of our natural environment and biodiversity. In particular, cost-benefit analysis need to include comparative studies of similar economic development projects completed elsewhere and the probabilities and costs of possible worst-case scenarios.

Another category of analysis can be based on employing spatial models of land use, which implies a fully integrated approach to study nature policy issues. In general terms, two approaches of land use models are distinguished. One deals with the allocation of land between alternative uses. The second describes endogenous land use and vegetation cover modelling, aiming at habitat patch dynamics. In order to be able to assess the effects of economic activities on ecological systems in landscapes, it is essential to understand the (ecological) processes determining the landscape. There is a need for an integrated approach that focuses on the development of the countryside – possibly combining economic growth, employment and protection of nature and landscape. Such a framework can incorporate economic and ecological dynamics based on regulations and human decision-making processes so that something can be learned about the consequences for ecosystem and land use configurations (Bockstael *et al.*, 1995). For instance, integration of models can provide a framework for regulatory analysis in the context of risk assessment of nature use.

The ecological part of an integrated land use model can be represented by a model of succession of ecological systems, by a minimum viable population of a biological resource representing a measure of biodiversity, or by biological features that are essential to species conservation, such as space for populations and cover or shelter (see Batabyal, 1998). Many economic phenomena are fundamental to land use changes. Economic models will characterise production decisions, and capture the allocation of land. The production decisions are subject to constraints imposed by prevailing technology, resources and policies. The relations between inputs and outputs are formulated by a production or transformation function. Inputs include items such as land, labour and capital, whereas outputs include (marketable) products.

Modelling the use of biodiversity is more than non-market valuation of endangered species and the economic incentives to preserve them from extinction but also the formulation of dynamic interactions between human activities and the ecological system. Such a dynamic interaction requires that the dynamics of species richness are taken into account explicitly. The richness of species is used as a general measure of biodiversity. Biodiversity provides four types of services to people (see Section 4.1). Using the utility function and the dynamics of species richness, conditions for existence, uniqueness and stability of steady state solutions can be established. Incorporating a threshold level of the richness of species can extend the analysis (Li and Löfgren, 1998).

Policy plans for nature protection can be studied with integrated ecological-economic models. Scenario-studies offer an alternative approach to the benefit valuation methods described in Section 4.5. Scenario-studies for biodiversity describe and analyse the past and the existing situation and sketch a future situation and how to arrive at it. Scenarios may indicate the threats to biodiversity of specific countries. They describe and analyse options for biodiversity protection, including the economic costs of carrying them out. Policy measures to protect biodiversity can then be ranked according to cost effectiveness and priorities in biodiversity protection can be established. The analysis of the scenarios can be carried out by means of partial cost-benefit analysis, assessment of indicators and by means of multi-criteria analysis, which makes it possible to rank alternative policy plans on the basis of explicit weights that are attached to the various criteria. These criteria should include economic aspects, such as costs and impacts on employment, as well as ecological values, such as the number of species and the quality of the landscape. A multi-criteria analysis facilitates trade-offs among different objectives. Once the scenarios have been developed it becomes possible to analyse their impacts. In addition, opportunities for financing biodiversity protection from domestic or international sources can be analysed together with the distribution of the benefits over the various stakeholders. So, in the process of scenarios making it is essential to communicate the various scenarios with the stakeholders at the levels of decision-making, from international to local (van Ierland *et al.*, 1998).

The analysis of habitat fragmentation, mainly resulting from physical (transport) infrastructure, provides another area where integration of ecology and economics is required. Habitat fragmentation can be defined as a process that reduces a large continuous nature area or habitat into two or more fragments. The fragments are isolated from one another, which leads to a degraded landscape. Ecological consequences of fragmentation can be captured by the island model of biogeography, which states that the number of species on an island (or an isolated habitat) is determined by a balance between immigration and extinction (see MacArthur and Wilson, 1967). This balance is dynamic, with species going extinct and being replaced (through immigration) by the same or by other species. Economic aspects can also be taken into account. For instance, new transport infrastructure offers access to products from previously ‘remote’ areas and decreases the cost of migration and travelling.

6. CONCLUSIONS, RECOMMENDATIONS AND A RESEARCH AGENDA

6.1 Conclusions

Economics and ecology can only contribute to the identification and solution of biodiversity loss and environmental degradation if adequate communication and co-operation between the disciplines will be established. The Convention on Biological Diversity provides a useful unifying framework for the development of policy, but securing international agreement on nature priorities is going to be extremely difficult. Subscription of the protocols on biodiversity and global climate change in Rio de Janeiro in 1992 does not mean that the aims regarding nature protection will be realised. Current and future economic activities will continue to result in negative effects on biodiversity and environment, except when additional activities to existing protocols on the protection of nature and environment are undertaken by national governments (van der Straaten, 1996). For example, individual citizens need to learn about local environmental issues, communicate that information and take action when necessary. For that purpose, governments should encourage and support local conservation organisations (Primack, 1998).

Although the continued conversion of natural areas can benefit certain individuals, it may threaten the flow of global services from nature. The loss of nature is therefore an international problem (Swanson, 1992). Only international co-operation and intervention can halt the loss of global services provided by nature. Investment in international institutions is necessary to make sure that global benefits of nature and biodiversity are well taken into account in the decision-making frameworks of people living in or affecting areas with diverse biological resources. International institutions must provide incentives for states to invest in the protection of the resources. The nature of these incentives will be a country's financial compensation for the global benefits generated by its biological resources. In other words, international institutions should provide for mechanism to channel payments to host states that invest in the protection of the global services from nature (Swanson, 1992; 1995).

In economic terms, the benefits from nature protection – although difficult to measure and varying from area to area – are limited to a local scale. In practice, people living in or near a protected ecosystem often capture little economic benefit from preservation or sustainable resource use. The benefits of protection increase with the scale from local to regional to national to global. In contrast, the economic costs incurred as a result of protection measures follow an opposite trend and tend to be felt most severely at local levels. The heaviest burden tends to be borne by people situated in rural areas in the vicinity of protected areas (Wells, 1992; 1995; Miller *et al.*, 1995). This creates an incentive problem. Due to the unfavourable cost-benefit from nature protection at local, national and global levels, the co-operation and support of local people has emerged as a major priority for the implementation of the nature policy (Miller *et al.*, 1995).

A multidisciplinary approach can contribute the formulation of policy measures to preserve or sustainable use natural resources (Turner *et al.*. 1998). To this end, a number of frameworks and models are proposed, in which ecology is incorporated in economic analysis by offering information about the hierarchy of dynamic ecological processes, such as ecosystem succession and long run trends of selection and evolution. A further adjustment of cost-benefit analysis is proposed by adding ecological constraints to the analysis, reflecting ecological thresholds and the relationship between ecological stability of our natural environment and biodiversity. Land use models, dealing either with the allocation of land between alternative uses, or with vegetation-cover modelling, can be extended by a model of succession of ecological systems. This can include a minimum viable population of a biological resource, or biological features that are essential to nature protection, such as habitats. Modelling the use of biodiversity requires that the dynamics of species richness is taken into account explicitly, for example, by incorporating conditions for existence, uniqueness and stability of steady state solutions. In addition, integrated modelling in the context of ecologically sustainable economic development of a region can be used to study policy scenariosthat address biodiversity issues in the region. External trends and events can play a substantial role in the economic performance and ecological quality of a region. Therefore, the scenarios should deal with policy, protection and development options (and their consequences) not only at a regional level, but also at a national and global level (see van den Bergh, 1991). Finally, habitat fragmentation, mainly resulting from physical (transport) infrastructure, has ecological and economic consequences. Ecological consequences can be captured by the island model of biogeography, which explains the influence of distance and size of an island or an isolated habitat on ecosystem succession. As physical infrastructure offers access to products from remote areas and decreases the cost of travelling, economic consequences can also be taken into account.

An understanding of social institutions, and ecological and economic processes, is necessary for analysing and understanding the forces that determine environmental change as well as for choosing a set of nature policies that move us toward a sustainable future. A sustainable use of natural resources calls for an institutional reform to establish property rights over the resource. Distortionary policies, such as bans, tariffs, subsidies and other trade measures which actually encourage unsustainable use of a natural resource, should be removed. These economic distortions harm not only the environment but also the national economy. Removal of these policies applies to both rich and poor countries. Indeed, since policies that lay down removal of economic distortions will encounter obstacles, the benefits of such policies must be made clear. Therefore, a full valuation of all goods and services that natural resources provide is necessary. The resulting values should provide a basis for policies aimed at internalising externalities in the use and exploitation of natural resources. Integrated models can help the analysis of complex interactions and implementation of policy packages in ecological-economic systems (Panayotou and Ashton, 1995).

6.2 Recommendations

For the protection of world’s nature a set of protected areas (natural reserves or parks) is not enough. Because of global ecological interdependence, the establishment of a world ecological network, in which the protected areas areconnected and buffered, is essential. Corridors between nature reserves facilitate movements of species, whereas buffer zones or nature development areas stand as a buffer between the core area and the surrounding area, and have the potential to be transformed into new core areas. The idea of a world ecological network should be promoted and supported by non-government organisations such as the European Centre for Nature Conservation (ECNC), the International Union for the Conservation of Nature (IUCN) and the World Wide Fund for Nature (WWF), and by international organisations such as the World Bank and the United Nations Environmental Program (UNEP). From the perspective of these international organisations, working with local conservation organisations and citizen groups is an effective strategy for dealing with the development and establishment of an ecological network. To prevent habitat fragmentation, individual citizens must become aware that they bear a direct responsibility for protecting the environment and that an incomplete ecological network has an influence far beyond their immediate community.

Policymakers must now rise to the challenge of developing and implementing a nature policy that takes into account particular aspects of economy-ecosystem interactions, such as land use and physical planning, protection of species and biodiversity, and sustainable and optimal use of renewable resources. Since much of the benefits of nature protection are likely to accrue to the people in developed countries, these countries have an important role to play. Policymakers should seek to extract financial support for nature protection from those who are able to pay. Large-scale North-South transfers of capital are essential to enable the developing countries to preserve habitats and to improve natural resource management. The Global Environment Facility (GEF) provides a vehicle for the international financial assistance to developing countries for global nature protection efforts. However, the lack of participation in the GEF by governments and the mismatch of large-scale funding over short periods with the long-term needs of developing countries are identified as major problems (Miller *et al.*, 1995; Primack, 1998). Existing international financial support is insufficient for effective nature protection in the poor countries and therefore the development of institutions for additional financial resources is essential. There is a pressing need to increase the amount of money directly available for protection activities. Where existing budgets cannot cover financial requirements to meet nature policy goals, countries should be able to borrow or to seek international financial co-operation. For the receipt, management and disbursement of financial support, formal and unequivocal administrative procedures are required.

6.3 Research agenda

The following are specific aspects that merit more detailed attention.

- The disciplines of ecology and economy both play an important role in identifying efficient, effective and equitable policy options for nature protection. However, designing and formulating a nature policy from the perspective of one or other discipline alone results in an informational disadvantage as the problem of nature loss requires the incorporation of ecological concerns into socio-economic decisions. In addition, ecologists and economists should take account of the insights various other sciences. Ecologists, economists and researchers from other disciplines must take up the challenge to provide further insight in the fundamental economic and ecological dimensions of nature and biodiversity. Getting the right mix of models, new theories, innovative approaches and practical examples will be the key to a successful identification and solution of species losses.
- Valuation methods can support the policy-making process, especially at a local level, regarding land use planning, agriculture and infrastructure activities. Assigning a monetary value to the services of nature indicates the importance of these services and the potential impact on mankind's welfare of continuing to destroy nature. Nevertheless, there are many conceptual and empirical problems inherent in monetary valuation methods (see Costanza *et al.*, 1997b). Additional research is necessary and further experience should be gained in order to determine for which cases monetary valuation is applicable. Capture of values to change incentives, efficiency and distribution of costs and benefits, can only occur if nature related values are methodologically correctly estimated.
- Because the spatial arrangement of habitats and of land cover have a substantial effect on virtually all ecological processes, modelling spatial dynamics of ecosystems is essential to ecologists, and has given rise to the development of a explicitly spatial approach to ecological structure and processes in landscape ecology. Many economic phenomena are fundamentally spatial in nature. However, often when economists address spatial distributionthey regard it as a constraint instead of explaining it as a dimension of an economic decision (Bockstael, 1996). Integrated modelling of economic and ecological systems and interactions is hampered by differences in spatial approaches. Especially in the area of economics the role of spatial perspectives needs to be upgraded.
- Various studies suggest that species become extinct at an escalating pace. However, it is impossible to say what the actual rate of extinction is, because much uncertainty exists about the numbers of species originally present. The difficulties inherent in developing nature policies are exacerbated by this lack of knowledge. Building capacity and

infrastructure for data generation and information dissemination tailored to the needs of policymakers and conservation organisations is necessary in order to make precise assessments and recommendations. For the protection of species it is essential to know which species are present as well as their geographical ranges, biological properties and possible vulnerability to environmental change (Wilson, 1988). The next step will be the development and establishment of a global early warning system, which indicates in an early stage an alarming decline of species.

- Decision makers rely on indicators, which inform them about existing or new environmental problems. Indicators describe a state or condition that is considered relevant for a more general phenomenon of interest. Because of several reasons, such as lack of data, this phenomenon cannot be measured directly (Brouwer *et al.*, 1998). Operational indicators should be guided by a number of specific criteria (see van den Bergh and Verbruggen, 1999). Changing environmental circumstances and increasing knowledge require a revision of existing indicators and the development of new indicators.

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