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ECOLOGICAL ECONOMICS: THE STUDY OF INTERDEPENDENT ECONOMIC AND ECOLOGICAL SYSTEMS

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ECOLOGICAL ECONOMICS: THE STUDY OF INTERDEPENDENT ECONOMIC AND ECOLOGICAL SYSTEMS

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1 Introduction

What is ecological economics? In institutional terms, the International Society for Ecological Economics was formed and launched its journal, Ecological Economics, in 1989. Since then membership of the society has expanded dramatically in all continents, and the journal has increased its output from four to twelve issues a year. Two ecological economics research institutes have been established: the International Institute for Ecological Economics at the University of Maryland, and the Beijer International Institute for Ecological Economics at the Royal Swedish Academy of Sciences, Stockholm. Both governmental and non-governmental organisations have begun to make appointments in the field, and environmental authorities are increasingly asking for an ecological economics perspective. Ecological economics is clearly something of a phenomenon. And looks as though it is here to stay — at least for a while. Yet the intellectual content of this development remains unclear to most economists. Indeed, most do not get beyond an identification of who is involved. The question 'what is ecological economics' is pre-empted by the question 'who does ecological economics'. The editors of the journal comprise two economists, Herman Daly and David Pearce, but also two ecologists, Robert Costanza (the editor) and Ann-Mari Jansson. Of the economists, David Pearce is widely recognised for his work on environmental policy but Herman Daly, despite his contributions to the economics of the steady state, is regarded by many economists as highly idiosyncratic. Given that many contributors to the journal have no background in economics, most economists have been persuaded that they need to know no more.

[1] We are grateful to M.S. Common for helpful comments on an earlier draft of this paper.

This paper offers a guide to the intellectual motivation, concepts and methods of ecological economics as these appear to us at the moment. As in any new development, ecological economics has been characterised by numerous false starts. It is also subject to conflicting claims by those who would harness it for their own purposes. Both things muddy the water considerably. Nevertheless, we think that it is becoming possible to identify the distinctive intellectual substance of ecological economics. While the evolution of the approach continues to involve a variety of irrelevant and ephemeral contributions that add very little to our understanding, we argue that there are matters of real substance involved. It is possible to identify both where an ecological economics approach differs from other approaches, and why. Nor are the points at issue trivial. Indeed, the most striking evidence that ecological economics is raising questions of fundamental importance is to be found in the fact that both the questions and the approach are increasingly being taken up by economists in related fields who remain dismissive of the journal. Papers in ecological economics are now appearing not only in the specialist environmental and resource economics journals, but also in the mainstream generalist journals.

The following sections address three separate questions. In section 2 we consider the foundations of ecological economics. This covers questions posed about the joint evolution of ecological and economic systems, and the reasons why these questions were not adressed either by standard models of natural resource utilisation or by standard models of ecosystem dynamics. The section puts the emergence of ecological economics in historical context, indicating its relation to earlier developments in economics (bioeconomic, mass-balance and entropy models) mathematics (non-linear system dynamics) and physics (far-from-equilibrium thermodynamics). Sections 3 and 4 then consider the two areas where ecological economics appears to be provoking a change in perceptions of the environmental problem. The first concerns the behaviour of jointly determined ecological and economic systems. The second concerns the valuation of non-marketed ecological resources. We indicate how these issues have been addressed in the literature, and how they are altering the approaches taken in related literatures. Sections 5 and 6 then discuss the policy implications of the change in perception about the time behaviour of joint ecological economic systems, indicating where an ecological economics approach indicates differences of either emphasis or substantive policies.

2 Foundations of ecological economics

The evolution of theory in the sciences is generally driven by the existence of a problem or set of problems which existing science is ill adapted to address. This is the case with ecological economics. The set of problems that stimulated the development of ecological economics are explored below, but they are all linked to two perceptions. The first is that the dynamics of economic systems are not independent of the dynamics of the ecological systems that constitute their environment. While there are different degrees of interdependence between economic and ecological systems, it is increasingly hard to find either ecological processes that are not impacted by economic activities, or economic activities that are constrained not just mediated by the natural environment. And the dynamics of both ecological and economic systems reflect this interdependence. The second common perception is that as economies grow relative to their environment, this affects the dynamics of both. More particularly, the dynamics of the jointly determined system become increasingly discontinuous, the closer economic systems get to the limits of the 'assimilative' and 'carrying' capacity of the environment.

Neither joint system dynamics nor threshold effects have been adequately addressed by existing economic and ecological theory. Yet the interdependence of ecological and economic systems has never been more apparent. Ecologists interested in the evolution of stressed ecosystems and economists interested in the development of economies operating close to the limits of assimilative capacity of their environment have not been able to appeal to a body of research in their respective disciplines that satisfactorily addresses this problem. In part this is because the problem has just not been posed before. But we conjecture that it is also because both disciplines have, over the last one hundred years at least, developed a very strong focus on the equilibrium properties of the systems under study, and that this has precluded many questions about the behaviour of those systems away from equilibrium. The Hicksian traverse, to take one of the better known counter examples, was dropped as an interesting topic in economics almost as soon as it was posed.

It is worth remarking that essentially the same general problem about the time behaviour of disequilibrium systems is stimulating analogous developments in both the natural and social sciences other than ecology and economics. Nor is ecological economics the only response within economics. Developments in the theory of non-linear economic dynamics, endogenous growth, technical change and preferences share a similar stimulus. The emergence of ecological economics can be seen to be part of a widespread reappraisal of the theory of complex dynamical systems. It is directly concerned with the implications of such system dynamics for the economic process, the role of the price system, and the allocation of non-marketed environmental resources. We shall argue that these implications are far reaching: affecting not only the construction of the economic problem, but also the valuation of environmental resources, and the identification of policy options and instruments

Thermodynamic models of economy-environment interactions

There are two closely related themes in the intellectual development of ecological economics. The first is reflected in work on the structure of joint ecological-economic systems and addresses changes in both the description of the physical dimensions of the joint system, and in the treatment of externality in the economic representation of that system. The second is reflected in work on the evolution of ecological-economic interactions and the role of scale in the evolutionary process. Historically, they have been carried in two distinct literatures: one on the thermodynamics, the other on population dynamics. Both literatures have called on the mathematics of non-linear systems.

In the first literature are a number of contributions, going back to Soddy [1912], whose primary concern has been that the physical relations underlying most economic models are incompatible with the laws of thermodynamics [Boulding, 1966; Odum, 1971; Georgescu-Roegen, 1971; Daly, 1973; Ayres, 1978; Ayres and Nair, 1984]. This inconsistency was argued to stand in the way of the development of models that could adequately capture the interdependence of economic and ecological systems. Its most significant contribution to the development of ecological economics is that it stimulated the mass-balance models which not only formalised one important property of economy-environment relations, but established a fundamental link between growth in material output and an increase in stress on the ecological systems of the environment. The assumption that a closed physical system must satisfy the conservation of mass condition, and hence that material growth in the economic system necessarily increases both the extraction of environmental resources and the volume of waste deposited in the environment was first made explicit in the context of a general equilibrium model by Ayres and Kneese [1969] and subsequently by Mäler [1974], but it is also a feature of the series of linear models developed after the mid 1960s [Cumberland, 1966; Victor, 1972; Lipnowski, 1976], and of later more generalised thermodynamic models [Amir, 1989,1994; Van den Bergh, 1993; O'Connor, 1991; Ruth, 1993; Ayres, 1994].

Two conclusions of relevance to the development of ecological economics were drawn from the early thermodynamic models. First, since perfect recycling of resources is precluded on thermodynamic grounds the potential growth of physical output is finite. Second, since the waste generated in the process of production is seldom inert, higher rates of physical growth imply higher rates of change in the processes of the environment [Geogescu-Roegen, 1971, 1973]. Similar conclusions had earlier been drawn by energy analysts based on their description of the interactions in ecological economic systems in terms of energy flows [Cottrell, 1955; Odum, 1971; Odum, 1975; Zuchetto and Jansson, 1985; Martinez-Alier, 1987; Cleveland, 1987;

Hall, Cleveland and Kaufmann, 1989]. However, some energy analysts went further in arguing that energy might better reflect the relative scarcity of resources than economic measures. Given that the price system does not adequately reflect the biophysical characteristics of the joint system (and hence the real effects of economic activity), the tracing of flows of energy and matter was initially argued by energy analysts to be an alternative to economic analysis. Indeed, there were some who initially saw ecological economics as the application of energy analysis to ecological economic systems. Energy analysis is now more commonly argued to be a useful complement to economic analysis where externalities are significant, and is as frequently linked with environmental economics as it is with ecological economics [see, for example, Pethig, 1994].

The most important and enduring of the implications of the thermodynamic approach for the development of ecological economics concern the evolution of the joint system. Change in one component implies change in the other, and the more highly 'connected' are the two, the more pronounced are the feedback effects between them: the more they co-evolve in Norgaard's [1984] terms [Boulding, 1978]. There is, however, both a spatial and temporal structure to the connectedness of jointly determined systems which was not discussed until more recently. The structure of the jointly determined system may be such that components that are unconnected over one time horizon may be highly connected over another, and a level of connectedness that is insignificant at one scale of activity may be highly significant at another [Perrings, 1986, 1987]. Indeed, the spatial and temporal structure of the joint system, and the relation between the scale of economic activity and the nature of change in ecological systems may be thought of as the *differentia specifica* of ecological economics. Daly has persistentently argued both that economic growth beyond the 'carrying capacity' of the biosphere will lead to dramatic environmental change (environmental collapse), and that there exists no feedback mechanism to ensure that an unregulated market economy will respect the carrying capacity of its environment [Daly, 1968, 1973, 1991]. Moreover, the strong support he has received from biologists may be identified as one of the origins of ecological economics [Vitousek et al, 1986; Costanza, 1989].

A separate development that runs parallel to ecological economics, that shares some of its motivations, but that ultimately addresses a different set of questions are the neo-Austrian models associated primarily with Faber and Proops [Faber and Proops, 1992, Proops, Faber and Wagenhalls, 1993; Speck, 1994]. These linear models are similarly based on thermodynamic considerations, but are motivated more by questions raised in the literature in capital theory than ecology. They are also concerned with the equilibrium properties of the constructed system, whereas ecological economics is most readily interpreted as a reaction against the equilibrium focus of both traditional economics and traditional ecology. Indeed, ecological economics is increasingly characterised by a focus on the behaviour of systems away from equilibrium. In fact, it appeals directly to the work of physicists on far-from-equilibrium systems [see for example Prigogine and Stengers, 1977; 1984; Nicolis and Prigogine, 1977], together with earlier developments in the theory of complex non-linear system dynamics based on work by Poincaré and Lyapunov [Li and York, 1975]. This work has begun to have a profound effect on the perception of the long run dynamics of ecological-economic systems [Rosser, 1991; Kay, 1991].

Bioeconomic models of economy-environment interactions

Aside from the thermodynamic models, another important foundation of ecological-economics is to be found in bioeconomics — coincidentally one of the cornerstones of orthodox renewable resource economics. This may be described as an approach to the exploitation of biological resources which depends on the specification of the control problem in terms of the dynamics of the population or populations being exploited [see especially Clark, Clarke and Munro, 1979; Clark, 1976; Conrad and Clark, 1987]. The exploited populations are the state variables of the problem and their dynamics are described by the equations of motion of the system. Bioeconomics is grounded in population dynamics of Lotka-Volterra form. The difficulty with bioeconomic models from an ecological economics perspective is that while such models do incorporate the dynamics of the resources under exploitation, they take a partial rather than a general equilibrium approach [Van der Ploeg et al, 1987]. Multi-species models with interactive terms improve the generality of bioeconomics, but from the systems perspective favoured by ecological-economics such models necessarily omit the most interesting feedbacks, particularly with respect to the interaction between organisms and biogeochemical and hydrological cycles.

Part of this problem is that proper specification of any system requires identification of all the key populations and communities. In terrestrial systems this may not be as daunting a task as it seems. The dynamics of those systems depend on the interactions between a relatively small number of processes, each of which is mediated by particular species. However, since the functions of such species is seldom invariant with respect to the relative size of all populations in the system, it cannot be assumed that the depletion of any given population leaves all other things the same. The ecological services which are exploited when any one population is harvested may be sensitive to the size of the harvested population. Ecological services include maintenance of the composition of the atmosphere, amelioration of climate, operation of the hydrological cycle including flood controls and drinking water supply, waste assimilation, recycling of nutrients, generation of soils, pollination of crops, provision of food, as well as the maintenance of particular species and landscapes [Ehrlich and

Mooney 1983; Ehrlich and Ehrlich 1992; de Groot 1992]. It is obvious, for example, that timber harvest can affect the hydrology on which timber production depends and this is sometimes taken into account in forestry models, but there are many other less obvious feedbacks that have not been taken into account. This is also the case with marine systems where the sensitivity of fish populations to the level of harvest due to interactions between components of the food web has historically been ignored.

Another aspect of the problem is that bioeconomic models generally assume that the systems concerned are Hamiltonian rather than dissipative, and even though it remains the case that ecological economics has yet to develop adequate models of discontinuous change in dissipative ecological-economic systems this remains high on the research agenda. What ecological economics seeks to add to bioeconomics are the insights to be had from recent developments in community and systems ecology. Here, recent research on scale, complexity, stability and resilience is beginning to influence the theoretical treatment of the coevolution of species and systems. The results that are most important to the development of ecological economics concern the evolutionary link between the spatial and temporal structures of hierarchical systems. It has, for example, been shown that the dynamics of fitness 'landscapes' reflects their interdependence: adaptive moves by one organism deforms the landscapes of those organisms with which it interacts. Moreover, small adaptive moves may trigger 'avalanches' of evolutionary response [Kauffman and Johnsen, 1991]. It is not yet clear how the transition between orderly and chaotic states in ecosystems is related to changes in scale as it is, for example, in turbulent fluid flows. There is, however, reason to believe that the spread of the effects of perturbation depends on spatial structure of ecosystems and that for terrestrial systems, at least, ecosystem dynamics are scale dependent. Landscapes may be conceptualised as hierarchies, each level of which involves a specific temporal and spatial scale [Wiens, 1989; Levin, 1992; Holling, 1994]. The dynamics of each level of the structure are predictable so long the biotic potential of the level is consistent with bounds imposed by the remaining levels in the hierarchy [Allen and Starr, 1982; Norton, 1990]. Change in either the structure of environmental constraints or the biotic potential of the level may induce threshold effects that lead to complete alteration in the state of the system [O'Neil, Johnson and King, 1989]. However, there remains considerable uncertainty about the way in which information is transferred accross scales [Levin, 1992].

Interestingly, the mathematics of non-linear dynamical systems were applied in biology (and especially in ecology) well before they were applied in economics. Examples of mathematical and Riemann-Hugonoit catastrophe had been observed in spruce budworm outbreaks in boreal forests in the 1970s [Jones, 1975; Ludwig et al, 1978]. Later work on dryland systems explored the role of catastrophe in the dynamics and management of the system [Walker and Noy -Meir, 1982; Walker, 1986; Westoby et al, 1989]. It also demonstrated the propensity for stressed systems to flip from one thermodynamic branch to another: grazing pressure beyond some critical threshold, for example, induced a non-reversionary switch in vegetation type [Perrings and Walker, 1995]. The framework within which such dynamics have been analysed is the 'four box' model developed by Holling [1973, 1986, 1987] in which the dynamics of ecosystems is described in terms of the sequential interaction between four system functions. These are exploitation (processes responsible for rapid colonization of disturbed ecosystems); conservation (the accumulation of energy and biomass); creative destruction (abrupt change caused by external disturbance which releases energy and matter); and reorganization (mobilisation of matter for the next exploitive phase). Reorganisation may be associated with a new cycle involving the same structure, or a switch to a completely different structure.

By now, the parallels between these theoretical developments in ecology and economics are striking. Economists have recently become interested in the dynamics of complex non-linear systems [Anderson, Arrow and Pines, 1988; Arthur, 1992; Brock and Malliaris, 1989; Goodwin, 1990; Puu, 1989; Benhabib, 1992]. Indeed, there are now numerous applications of nonlinear dynamics in economics, particularly to problems in finance where there is an interest in endogenising fluctuations [see for example Hommes, 1991; Granger and Terasvirta, 1992; Scheinkman and LeBaron, 1989]. What is particularly interesting is that the approach is rationalised in terms of the recognition that complex non-linearity is now generally accepted as a useful way of approaching the description of real phenomena in the natural sciences, and especially in epidemiology, biology and ecology [Brock, 1992]. These economists have paid less attention to spatial scale and its significance at or near system thresholds [though see Puu, 1981; 1989; Rosser, 1991], but there is now a growing body of literature with roots in geography which seeks to inject a spatial dimension into nonlinear economic models [see for example White, 1990; Hannon, 1994]. There is also an economic analogue to the biologist's interest in evolution and the significance of codependence between gene landscapes. The steady accumulation of evidence that economic development is not a stationary process, that human understanding, preferences and technology all change with development and that such change is generally non-linear and discontinuous [Wilkinson, 1973; Common, 1988], has prompted economists to seek to endogenise technological change [Romer 1990a, 1990b; Lucas, 1988; Barro, 1990; Rebelo, 1990]. Moreover, this has begun to be applied to 'environmental' technology [Huang and Cai, 1994]. To date such research is not linked with parallel work in the nonlinear biological sciences, but the scope obviously exists for such a link.

To summarise,there appear to be three distinct but related strands in the development of ecological economic models. The first is to be found in the realisation not just that the economy and its environment are jointly determined systems, but that the scale of economic activity is now such that this matters. There are environmental feedback effects that have potentially important implications for the welfare of both present and future generations. The second is to be found in the perception that the dynamics of the jointly determined system are characterised by discontinuous change around critical threshold values both for biotic and abiotic resources, and for ecosystem functions. Ecological economics is concerned with the evolution of non-linear ecological economic systems in which path-dependence means that system history is relevant to current and future opportunities. The third lies in the recognition that the stability of the jointly determined system depends less on the stability of individual resources, than on the resilience of the system — or the ability of the system to sustain its self-organisation in the face of stress and shock. Each of these strands affects the valuation of environmental resources. Each also affects the nature of the policy response, both in terms of the target of that response and the instruments required to meet them. Before we consider the implications of an ecological-economics approach for environmental valuation and policy, however, we show how scale effects, discontinuities and ecosystem resilience are integral components of the ecological-economic problem.

3 Scale, resilience and the dynamics of joint ecological-economic systems

There is a widespread perception in the ecological-economics literature, that the implications of ignoring the ecological impacts of economic activity would be less significant in a world with a smaller human population and consequential level of demand for environmental resources. This perception is, of course, shared by many others dealing with the same phenomena. Most environmental economists would, if pressed, concede that population growth and the population externality are at the core of the environmental problem. What is interesting, though, is why the level of demand for environmental resources is thought to be important.

There are two aspects to the argument in ecological economics. The first relates current levels of demand to the traditional concept of carrying capacity. It holds that current population and consumption levels are beyond the long run carrying capacity of the biosphere, and that this implies both degradation of the resource base and the necessity for some sort of Malthusian adjustment [Daily and Ehrlich, 1992; Ehrlich and Ehrlich, 1970, 1990; Ehrlich, Ehrlich and Daily, 1993; Kendall and Pimental, 1994]. This line of argument supposes a traditional view of ecosystem dynamics. The second does not. The central point in the second argument is that ecological economic systems are characterised by multiple locally stable states, the boundaries of each local stability domain marking points of discontinuity or thresholds between such states [Rosser et al, in press]. They are the unstable manifolds of the general system. In any one state, the distance from the boundary is a function of the stress to which the ecological component of the system is subjected. The greater the level of stress, the smaller the perturbation needed to dislodge the system from one stability domain to another. So long as the depletion of ecological resources and the generation of ecologically significant waste lies inside the limits of the carrying or assimilative capacity of the system in some state, local effects tend to remain local and the dynamics of the system tend to remain predictable. The regime to which human and other species are adapted remains intact. However, given the existing size of the human population, current rates of population growth and consequential rates of growth in the demand for ecological services are bringing in an era of novel evolution of ecological economic systems that involve an irreversible and rapid change in the state of the system analogous to the density-dependent threshold effects or the 'avalanches' of evolutionary response noted in ecology [May and Anderson, 1979; Kauffman and Johnsen, 1991]. This is partly due to the increased interconnectedness of ecological and economic systems in time and space [Costanza et al. 1993]. This is argued to have moved societies and natural ecosystems into such novel and unfamiliar territority that the future evolution of both has become much more unpredictable than it was for earlier generations [Holling 1994; Folke, Holling and Perrings, 1994].

Critical ecological thresholds — the unstable manifolds of ecosystem dynamics — are defined in terms of the level or density of ecosystem components. For example, thresholds in predator-prey systems are defined in terms of the relative density of each. If the relative density of one exceeds the critical threshold, the system will frequently experience discontinuous and unpredictable change. The perception that the dynamics of jointly determined systems may be discontinuous around ecological thresholds is a characteristic feature of ecological economics. It differs from the Marshallian view that characterises much research in economics (including environmental economics) though it is consistent with the growing literature in economics on nonlinear system dynamics referred to earlier.

The main point, as we have already remarked, is that the closer the system is to a threshold, the smaller the perturbation needed to dislodge it. There already exist numerous examples of discontinuous ecological change as a result of a gradual build-up of economic pressure. In many such cases large-scale modifications of ecosystems are the result of many local and disconnected activities (the tyranny of small decisions). The widespread destruction of mangrove ecosystems in South East Asia and South America for shrimp farming is an example. In this case, the incremental destruction of mangrove systems has had a non-incremental affect on the ability of these systems to provide spawning and nursery grounds for fish and shellfish [Barbier, Burgess and

Folke, 1994]. In the Honduras, the incremental transformation of the landscape has induced the evolution of new diseases by shifting the pattern and abundance of insects [Almendares et al. 1993]. In many coastal waters, where an incremental build-up of pollutants has changed the structure of plankton communities causing an increase in toxic algal blooms, an incidental effect has been the transmission of cholera from one country to another through the migration of infected marine algae populations [Epstein, Ford and Colwell, 1993].

The connection is often very indirect indeed. Consider, for example, the link between migratory insectivorous bird populations and changes in insect (budworm) outbreaks in Boreal regions of Canada. A set of thirty-five species of insectivorous birds is one of the controlling factors of the forest renewal patterns produced by budworm population cycles. Simulations based on long-term studies of budworm/forest systems dynamics indicate that if the bird populations were reduced by around 75% the whole pattern of boreal forest renewal would be fundamentally altered, and the whole of the forest-based economy disrupted [Holling 1988]. A large proportion of these bird species spend the winter in Central America and parts of South America, where they are adversely affected by a range of incremental changes in land use (involving habitat destruction) and agricultural technology (involving the use of pesticides and herbicides). Radar images of flights of migratory birds across the Gulf of Mexico over a twenty year period reveal that the frequency of trans-Gulf flights has already declined by almost 50%, approaching the range of uncertainty in Holling's simulations. Hence, Canadian Boreal forests and the economic activities based on those forests, appear to be threatened in a very non-incremental way by human population growth in Central and South America and the land-use pressures to which it has given rise [Holling 1994].

The biophysical carrying and assimilative capacity of ecological systems is not, of course, static. It varies markedly with the culture, preferences and technology of the user [Berkes and Folke 1992, 1994; Daily and Ehrlich 1992]. But one implication of the change in the scale of human activity relative to the existing carrying and assimilative capacity of the environment is that produced capital may not be the binding constraint on economic performance: it may be the capacity of ecosystems in a given state to support continued expansion of economic activity. This has been referred to in the ecological-economics literature as ecological scarcity [Barbier, Burgess and Folke 1994], and lies behind the emphasis in ecological economics on the importance of investment in natural capital [Jansson et al, 1994]. The general point made by ecological economists is familiar: that the scope for substitution between produced and natural resources in both production and consumption is limited by the evolutionary capacity of the system. The problem lies not in the human ability to develop new technologies, but in the ability of natural systems to adapt to change in demand as a result of new technologies.

Resilience and sustainability

In non-equilibrium, non-linear dynamical ecological-economic systems the problem is to understand the behaviour of the joint system away from any system equilibria. More particularly, the problem is to understand the behaviour of the joint system when its component parts are subject to stress induced by change in 'environmental' conditions, where the term environmental here refers to the given conditions in which a system of interest operates. The natural focus of ecological economics is not the properties of the equilibrium state or states of the system, but the persistence or sustainability of system function under varying environmental conditions.

Ecology is notoriously imprecise in its analysis of the stability of dynamical systems. There is disagreement as to which components of the system should be evaluated. Moreover, some concepts of ecological stability accord with standard Lyapunov criteria, others do not. The shift in focus from population to system dynamics does, however, suggest the value of measures that capture the sensitivity of the system structure to external disturbance. Holling's notion of system resilience does this. Resilience is defined to mean the propensity of a system to retain its organisational structure following perturbation, and so refers to the stability of structure, process and function rather than the component populations of an ecological system [Holling, 1973, 1986; Common and Perrings, 1992]. It reflects the sensitivity to disturbance of the 'integrity' or 'health' of the system [Kay, 1991; Costanza, Norton and Haskell, 1993].

There is a natural link between sustainability and resilience. Sustainability, as a concept, has been given a bewildering variety of definitions which it is beyond the scope of this paper to review [though see Pearce, Markandya and Barbier, 1989; and Turner, 1993]. But as far as we are aware there is no definition of sustainability that does not imply maintenance of the productive potential of the asset base. System resilience is a measure of the robustness of that potential in the face of the stress induced by economic activity. In terms of the theory of dissipative systems, maintenance of system resilience implies maintenance of the system either along a given thermodynamic branch [Kay, 1991] or within a basin of attraction, the structure of which defines the self-organisation of the system [Isomäki, 1993]. The problem for economic policy lies in the fact that market prices do not indicate whether a system is approaching the limits of system resilience. This is partly due to the structure of property rights and other institutions, partly to our lack of understanding of ecosystem dynamics, and partly to the public good nature of many environmental resources. It turns out that maintenance of system resilience in these conditions has some very clear policy implications which are explored later in this paper

4 Ecological economics and the valuation of environmental resources

The problem of valuation is central to any economic approach. We now consider whether the distinctive perception of system dynamics implies a distinctive approach to the valuation of ecological resources. Environmental economists have developed a terminology of valuation which distinguishes between use values (direct and indirect use value, option value and quasi-option value), and non-use values (bequest and existence values) [recently reviewed in this journal by Cropper and Oates, 1992; but also by Kolstad and Braden, 1991; and Kopp and Smith, 1993]. Debate continues over the precise boundaries between these different components of economic value, but the conventionally accepted approach to the valuation of environmental resources is based on the assumptions that households maximise utility deriving from these different sources of value subject to an income constraint; and that their private willingness-to-pay is a function of prices, income and household tastes (including environmental attitudes), together with conditioning variables such as household size and so on. The social value of environmental resources committed to some use is then simply the aggregation of private values.

Ecological economics has raised questions both about the nature of the conventional distinction between use and non-use value, about the most appropriate method of uncovering the distinction, and about the relation between private and social value. These include the following: is there such a thing as existence value and can it be measured? Can the current approach to valuation adequately capture the 'true' social value of ecosystems and their interrelationships? What is the most appropriate method for uncovering such social value? While these are the same questions that are posed by others concerned with the valuation of environmental resources, the answers proposed by ecological economics turn out to be quite distinctive.

It is useful to distinguish between issues that are general to all economic research on the environment, and those which are specific to the ecological economics. Amongst the former is the very general problem of the nature and formation of preferences. Ecological economics, like environmental economics, has had to confront the accumulation of evidence that some of the most basic axioms of the theory of demand are systematically violated by humans in both controlled and uncontrolled conditions. The transitivity axiom (the preference reversal phenomenon) is prominent in this repect [Lichtenstein and Slovic, 1971; Grether and Plott, 1979]. Evidence of nontransitivity of preferences has been explained in a number of different ways: in terms of regret theory [Loomes and Sugden, 1983; Loomes, Starmer and Sugden, 1989]; differences between individual responses to choice and valuation problems [Slovic and Lichtenstein, 1983]; differences in the devices used by experimenters to elicit valuations; and the 'random lottery selection' incentives system [Holt, 1986; Karni and Safra, 1987 and Segal, 1988]. However, recent experimental work that has controlled for shortcomings in experimental design has confirmed the existence of systematic violations of transitivity [Loomes, Starmer and Sugden, 1989]. The problem has to be addressed directly. One fruitful line of analysis suggests that utility may not be a function of states of wealth, but may be assigned to gains or losses relative to some reference point, and that it may reflect loss aversion. Whether one is seeking to value resources in a jointly determined ecological-economic system or not, the problem of preference formation is important for non-market economic valuation. The reference operating condition 'familiarity with the good' needs considerable formal clarification in the light of evidence that individuals engage in 'incomplete optimisation' and are subject to 'preference learning'. Most recently Kahneman [1994] has argued that individuals find it difficult to predict their future tastes. They rely on past experience, but in a very imperfect and selective way, and it is this that violates the economic rationality paradigm. Clearly, economic valuation of resources needs to address the questions raised by the fact that rationality is bounded no matter what the field of study.

Where ecological economics raises questions that are distinctive is in its analysis of the distinction between use and non-use values, and in its treatment of the link between non-use value and environmental public goods. The systems approach has implications for both things. First, it implies an understanding of the role of environmental resources in the 'production' of the full range of ecological services of value to humanity [Dasgupta, Folke and Mäler, 1994]. What is being valued is the way in which individual biotic or abiotic resources mediate ecological functions. This requires specification of a production function that captures the indirect use value of individual resources. Although the links between the output of ecological goods and services and particular environmental resources may be very complex, it has been claimed that for many non-marketed environmental resources the additional data requirements of the approach are not onerous [Turner 1988a and 1991; Farber and Costanza, 1987; Barbier, 1993 and 1994].

When we consider the valuation of the functionality of complete systems as distinct from individual components of those systems, however, we are unable to apply the same methods. The structural and functional value of 'healthy' evolving ecosystems cannot generally be assessed in the same way. Fundamental life support services underwritten by the resilience or 'integrity' of the system are impure public goods that have a value which in some sense determines the value of other ecological goods and services, but that is ignored by individual users of the resource. A rough approximation of the significance of this 'primary' ecosystem value may be given by deployment of 'damage avoidance', 'substitute service' or 'replacement cost' methods [Gren et al 1994; Turner and Pearce, 1993]. In this sense, the primary value of the system is the value of the insurance it provides against such losses. At present there are no good means of evaluating this, but we conjecture that much of what is currently being treated as existence value reflects the subjective evaluation of primary (use) value.

The use and non-use value of non-marketed environmental resources is generally distinguished in terms of standard welfare measures [Larson, 1993]. If a household production function is additive separable, with one term describing the utility deriving from the environmental resource alone and a second term describing the utility deriving from the same environmental resource and all other marketed resources, then the first term reflects the non-use value of that resource, while the second term reflects its use value [Mäler, 1994]. Since non-use value involves neither personal consumption of derived products nor *in situ* contact, one has to assume some separability between the nonmarket good that is the subject of non-use value and all marketed goods. The questions raised in ecological economics are, first, whether the separability of utility functions is supportable in interdependent ecological economic systems, and second, how the category of non-use value may be related to the perception of environmental public goods.

Environmental public goods

Consider, first, the relation between non-use value and environmental public goods. To the extent that non-use value signifies that the individual places value on maintaining resources that are of current or potential value to other individuals, societies or species, it can be interpreted as evidence of altruism. In the environmental economics literature, this has recently been argued to distort estimates of social value derived from stated preference methods. Kahneman and Knetsh [1992a,1992b] argue that altruism is really the purchase of moral satisfaction, the 'warm glow effect' and deep down is a self-regarding motivation. This calls into question the degree to which contingent valuation bids, for example, reflect the inherent economic value of public environmental goods [Smith,1992; Harrison, 1992]. Andreoni [1990] has shown that in the public goods context, individual willingness-to-pay may be related both to the amount contributed towards the provision of the public good and the degree of social approval an individual perceives to be associated with making such a contribution (probably relative to some average contribution), so distorting any estimate of social value derived.

The notion of social approval can be extended to the concept of the 'social interest' under which individuals are argued to have social preferences quite separate from their self-interested private preferences. Margolis [1982], for example, posits two separate concerns — self interest and social or group interest — that underline human motivation. The origin of social interest may be explained by theories of reciprocal altruism, mutual coercion or by socio-biological factors [Sen, 1979; Elster, 1989]. In the Margolis model individuals allocate income between social and private preferences so as to be seen to be doing their 'fair share' for the provision of collective social welfare. In this case the willingness-to-pay estimate derived from a contingent valuation may be more a personal judgement about socially acceptable fair shares than the inherent economic value of the environmental resource in question.

The Margolis analysis strains the bounds of enlightened self-interest. Even though individuals make trade-offs between social and private interests, their behaviour may still be motivated by ethical preferences for justice and fairness. Nevertheless, it still seems that 'pure altruism' is not captured, and if existence value is interpreted as an anthropocentric measure of the intrinsic value of nature (the value to non-human users of natural resources) we do require such a motive. If a person chooses an act of conservation that they believe will involve personal costs but will satisfy their moral commitment to conservation, then the resource being conserved may be said to have 'moral resource' value. Such individuals are unlikely to accept compensation for the loss of the resource, especially if it is unique. They tend to exhibit lexicographic preferences when faced with the loss of, for example, a particular species (violating the axiom of utility function continuity). Thus some zero bids in contingent valuation surveys may be interpreted as protest bids related to ethical commitments, or a refusal to link nature conservation and money expenditure [Stevens et al, 1994].

Sagoff [1988] has distinguished between the individual's role as a consumer and as a citizen. As a citizen the individual considers the benefits or disbenefits of a policy, project or course of action to society (nation usually) as a whole. The citizen is guided in his or her deliberations by 'ethical rationality' which is underpinned by sentiment, historical, cultural, ideological and other motivations and values. This is analogous to the perception of the 'intrinsic' or 'moral resource' value of nature noted by Sen [1977]. The valuation of most environmental public goods engages the citizen part of the 'dual self', not the consumer. By this view, aggregate willingness-to-pay estimates derived from stated preferences is an inappropriate measure of the social value of environmental public goods.

The problem with estimation of the non-use value of environmental public goods are one reason for the polarisation of views on the contingent valuation method [Mitchell and Carson, 1989]. Advocates of stated preference approaches argue that contingent valuation is capable of yielding both use and non-use values for a range of environmental resources, and contingent valuation is clearly developing into a major focus of environmental economics. The reliability and validity testing protocol [Arrow et al, 1993] adopted in response to earlier criticisms of contingent valuation is argued to be sufficient to show that contingent valuation results are not random answers, and that they provide theoretically consistent and plausible measures of value for many types of environmental resources [Smith, 1993]. The next stage in the evolution of contingent valuation will presumably be one in which a 'valuation protocol' is developed in order to get a standardised set of definitions of environmental 'commodities'. According to Smith [1993] the definitional structure must be consistent with people's environmental perceptions, compatible with ecological constraints and interrelationships and responsive to policymakers requirements for valuation data.

From an ecological economics this trend has two major implications. The first derives from the fact that the interdependence of ecological and economic systems reveals a set of public goods that is both more deeply embedded and less understood than the 'habitats' or 'recreational amenities' valued by stated preference methods to this point. Non-marketed services deriving from ecological systems are routinely stressed by economic activities, the value of which depends on the resilience of those systems in ways that are little understood by economic agents. While agents may have preferences for the health of ecological systems on which they depend as consumers and as citizens, those preferences are generally based on incomplete information. The public good nature of such resources and the existence of fundamental uncertainty about their role in production and consumption mean that any estimates of their social value based on aggregate willingness-to-pay bear little relationship to the opportunity cost of their use.

The problem is compounded when such resources are not used by the agents themselves, but by others whose welfare is of concern to the agents. Indeed, some have argued that application of stated preference methods has failed to yield any plausible non-use values of environmental public goods. 'Embedding or mental account' bias problems and citizen preference revelation problems are argued to have compromised the results obtained in almost all cases [Common, Blamey and Norton, 1993]. Amongst the most pessimistic evaluations is the view that the method is only applicable in a severely restricted range of contexts, i.e. ones in which individuals can directly perceive the environmental resource and the change in the state of that resource [Bowers, 1993]. It has been suggested that difficulties of the kind referred to by Common et al may be averted if estimates of the values relating to 'unfamiliar' environmental resource contexts are compared to referenda results [Mitchell and Carson, 1989]. But the resulting value estimates still cannot qualify as legitimate 'economic' data for, for example, incorporation into standard cost benefit analysis or development of Pigovian taxes. The problem for ecological economics is to develop an understanding of the role of environmental public goods in production and consumption so as to open up the prospect of alternative approaches to the valuation of such resources, as well as to improve the information on which stated preferences may be elicited.

The second set of research questions raised by the trend towards valuation by stated preference methods concerns the problem of preference formation (and especially social preference formation) in a jointly determined ecological economic system. An ecological economics approach would suggest that it is unhelpful to regard preferences as fixed in anything but the very short term, and that preferences (like technology) evolve with the system. Contingent ranking or contingent choice may certainly play a useful role in revealing current citizen or social preferences [Adamowicz, Louviere and Williams, 1994], but the way in which social preferences change with changes in the characteristics of the system is an open research question.

Use and non-use value in the ecological economics approach

The significance of the interdependence of ecological and economic systems is that economic production functions will, in general, include ecological arguments. More importantly, while these arguments may include specific environmental resources — particular species, for example — they will also include a range of ecological services and the ecosystem functions which support those services. The sense of much of the ecological-economics literature is that at current levels of demand for environmental resources few ecological services can still be treated as non-scarce. Moreover, while almost all have at least some of the properties of public goods, most are congestible.

The natural approach to valuation in ecological economics is accordingly the production function approach (and associated revealed preference methods) [Perrings et al, 1995; Mäler, 1994]. However, it is clear that the functions relating welfare to the employment of environmental resources are not the static household production functions assumed in much of the early work on this problem. The adoption of a systems perspective serves to re-emphasise the point that economic systems are underpinned by ecological systems, and that there is a dynamic interdependency between the two. Biophysical systems may be part of the constraint set that bounds economic activity, but their internal dynamics are sensitive to the level of economic activities (extraction, harvesting, waste disposal, non-consumptive uses). The dynamics of the joint system need to be specified in the production function if it is to reflect the implications of current economic activities [Common and Perrings, 1992]. Moreover, it needs to be recognised that in hierarchical systems smaller subsystems change according to a faster dynamic than do larger encompassing systems, in the same way, for example, as the dynamics of the individual firm or household are faster than the dynamics of the economy to which they belong [Norton and Ulanowicz, 1992].

This said, even the best of the production function-based estimates of the value of non-marketed environmental resources do not capture their full contribution to the life-support and other enabling functions provided by ecosystems [Gren et al, 1994]. In part this is because the underlying structure and functions of ecological systems are public goods whose role in individual production and consumption processes has yet to be adequately specified. The value of ecosystem structure in enabling production or consumption to take place has been referred to as 'primary' or 'glue' value. It consists of the system characteristics (environmental public goods) upon which all ecological functions depend [Turner and Pearce, 1993; Gren et al, 1994]. Primary value arises from the enabling function of systems which support processes having direct or indirect use value (referred to by the same authors as 'secondary value'). These processes depend on the integrity — existence, operation and maintenance — of the ecosystem as a whole. But though such primary value is clearly a component of the economic value of environmental resources (it is an indirect use value) it is not directly measurable in terms of consumer preferences for the reasons indicated in the last section. That is, it is a public good whose role is subject to fundamental uncertainty. Not only do economic agents seldom understand the role of functional ecosystems in the provision of goods and services, they also have a strong incentive to dissimulate about their preferences. For these reasons, the social value of the ecosystem is likely to exceed the estimated sum of the values of its individual functions [Turner and Pearce, 1993; Costanza, Farber and Maxwell, 1989].

An additional difficulty in the aggregation of economic derives from the fact that the dynamic interdependence of economic and environmental components of the system ensures that there will be feedbacks to any decision which leads to a change in the level of ecological services, and that because of both the evolutionary nature of an ecosystem's responses to change in the level of economic stress and the existence of threshold effects, these feedbacks may be largely unpredictable except over ranges of stock sizes in which the ecosystem exhibits local stability (stays within the thresholds). Put another way, there will be dynamic general equilibrium effects which (a) will tend to be ignored in the process of aggregating values derived from partial observations of expenditure patterns given some change in the level of ecological services, and (b) will be unpredictable except over the range of biodiversity in which ecosystems are stable.

Consider the problem of biodiversity loss. Part of the difficulty in estimating the value of ecosystem integrity lies in the fact that under given environmental conditions, a healthy ecosystem contains a degree of ecological redundancy. There are species or populations which may not be important under those environmental conditions but which may become important if the environmental conditions change.

There is thus an 'insurance' value to the maintenance of the general capacity of the system that is closely related to the diversity within it [Swanson, 1995].

In terrestrial ecological systems, the use value of currently redundant species lies in capacity they give the system to absorb shocks without loss of resilience [Holling, 1990; Schindler, 1990; Walker and Noy-Meir, 1982]. Systems ecologists now take the view that the dynamics of most terrestrial ecosystems are dominated by a small set of structuring processes [Holling 1992], mediated by different species under different environmental conditions [Schindler 1990; Vitousek 1990]. In other words, stressed ecosystems may maintain many of their functions even though the composition of the species changes. In fact the ability of the key structuring processes of a system to operate over a range of environmental conditions depends on the number of alternative species that can take over functions when perturbation of an ecosystem causes the disappearance of the species currently supporting those functions [Schindler 1988]. In short, it is the 'functional' diversity of ecosystems that determines their resilience. Other things being equal, the greater the mix of species in terrestrial systems, the greater the resilience of those systems implying the greater the perturbation they can withstand without losing their self-organisation.

Biodiversity underpins the ability of far-from-equilibrium ecological systems to function under stress, and in so doing it underpins both the predictability and the productivity of those systems. It follows that the use value of biodiversity conservation lies in the value of that protection: the insurance it offers against catastrophic change. More importantly, it follows that redundant species have use value even if they are not currently used.

Much of the 'existence' value (by definition, a non-use value) assigned to natural organisms turns out to be the value attaching to precisely this property of those systems. That is, it is the value attaching to the capacity of the system to continue to function effectively over a range of environmental conditions. While it is surely a use-value — specifically an option value — the individual need not perceive the chain linking it to any given economic activity. As a result this value has frequently been confused with non-use altruistic or vicarious values. However, the general point is that the ecological economics approach encourages a much wider perception of the use value of biological resources than has been common in the literature.

To summarise, the social value of environmental resources committed to some use may not be equivalent to the aggregate private value of the same resources in any given system, because of the following factors:

(i) Incomplete information. The complexity and coverage of the underpinning functions of resilient ecosystems is not known. What is known is that the system is characterised by discontinuous and often irreversible change around thresholds of resilience, and hence that the range and probability distribution of the consequences of a loss of resilience may be unpredictable.

(ii) Environmental public goods. Because the value that can be instrumentally derived from an ecosystem is contingent on the evolution of that system the health of the system represents an environmental public good from which all economic activity derives value. This 'primary' value is not currently measurable.

(iii) The aggregation problem. This refers both to the problem of aggregating estimates of private willingness to pay in the face of externality, and to the fact that a healthy ecosystem is more than the sum of its individual components.

(iv) The redundancy problem. A resilient ecosystem also contains a redundancy reserve, a pool of latent 'keystone' species which are required for system maintenance in the face of stress and shock. Since such species may only occupy 'keystone' positions under environmental conditions not previously observed their identification under current environmental conditions is problematic.

These sources of discrepancy between private and social value establish both an important part of the ecological economics research agenda and the major policy implications of the approach. If one accepts the arguments on the dynamics of the joint system at the core of ecological economics approach, it is clear that the valuation problem centres on the role and significance of a set of environmental public goods that are currently very imperfectly understood. Many of the difficulties in the valuation of these resources are generic to the class of congestible public goods. Others derive from the problem of specifying their role in the dynamics of the joint system. Still others stem from the evolutionary nature of social preferences in the joint system. To this point ecological economics has asked more questions in these areas than it has provided answers. But in asking such questions it has also suggested a particular approach to policy.

5 Policy implications: sustainability criteria

One of the main results in ecological economics is that intertemporal efficiency in the allocation of resources is neither a necessary nor a sufficent condition for the sustainability of resource use [Common and Perrings, 1992]. Indeed, sustainability is widely viewed as an equity rather than an efficiency category [Pearce, 1987; Norgaard, 1991]. We consider the policy implications of this perception below. At the same time, our discussion of the valuation of environmental resources implies that much environmental degradation is a consequence of the discrepancy between private willingness-to-pay for and social opportunity cost of environmental goods and services, and especially of environmental public goods and services. This too has policy implications. To see how and why the policy implications of an ecological econmics approach are distinctive, however, we need to consider the way in which equity and efficiency considerations are addressed in the policy problem.

Sustainability, resilience and the 'stabilisation' of ecological-economic systems

To begin, we recall that the non-linearity of joint ecological economic systems shows up in the pervasiveness of path dependence, threshold effects and the irreversibility of change. The conversion and modification of ecosystems by the current generation means less diversity and may mean lower ambient environmental quality for all future generations. The effect of activities already undertaken on, for example, the concentration of greenhouse gases will involve very significant change in the biogeosphere irrespective of whether or not corrective measures are taken now. By changing the environmental conditions within which the ecological economic system operates, past emissions of greenhouse gases may, for instance, have changed the set of species required to assure the functioning of a number of ecosystems in the future. The result is that the environmental 'risks' associated with economic activity have become endogenous (the 'risks' are affected by that activity), and correlative (they are not statistically independent). Given that they are also potentially catastrophic, it is not surprising that the menu of policy choices should focus on measures that preserve options whilst encouraging learning [Chilchilnisky and Heal, 1993].

The specific problems to be addressed by policy derive from the fact that there currently exist no signals as to the long-term consequences of a wide range of environmental effects including the loss of ecological resilience. This is partly because ecosystems typically continue to function in the short term even as resilience declines. Indeed, they often signal loss of resilience only at the point at which external shocks at previously sustainable levels flip those systems into some other basin of attraction and so some other regime of behaviour. The principal policy challenge is not just to correct the institutional biases in the price signals that constitute the main measures of system performance, but to design institutions to accommodate the opacity of ecosystems, and the uncertainty it engenders.

There are, in general, two sources of uncertainty that have to be addressed in designing a sustainable management strategy. First, in any feedback control problem optimisation of the problem requires the continuous measurement of state variables. The available measures may be subject to error. Second, the system dynamics may be themselves be known imperfectly, implying that feedback control will necessarily be misdirected. The main source of uncertainty of the first kind, at least in so far as the economic system is concerned, is identical to the source of market failure. Measurement error in the ecological-economic system is synonymous with the failure of prices to act as accurate system observers [Perrings, 1991]. The main source of uncertainty of the second kind, again in so far as the economic system is concerned, appears to be the evolutionary nature of the system. Not all evolution creates the same amount of difficulty. Genotypic evolution is in principle unpredictable, but phenotypic evolution is not [Faber and Proops, 1992]. Failure to predict phenotypic evolutionary trends may be due to the product of ignorance about the functional structure of ecosystems, but failure to predict genotypic evolutionary trends is inherent in the nature of the changes involved. Genotypic evolution is accordingly the least tractable source of system uncertainty. This is not to say that knowledge of the system, or at least of parts of the system, cannot be improved over time. Estimates of the distribution of possible environmental outcomes of economic activity may well be improved through, for example, a passive Bayesian learning process. But it does imply that there is likely to remain a very large measure of fundamental ignorance about the future effects of current actions.

The problem for policy lies in the fact that the ecological-economic system is neither observable (through the set of prices) nor controllable (through any set of incentives based on those prices). If the system is not observable, the available information set does not include a sufficient profile of the statistical properties of the unavailable information set to predict its conditioning effect on the future behaviour of the system [Perrings, 1991]. This may be because of the existence of novel developments whose implications for the time-behaviour of the system are unclear. In a far-from-equilibrium evolutionary system, the distribution of outcomes associated with such developments cannot be inferred from the history of the system both because of the paucity of relevant observations, and because of the effect of novelty on the system parameters [O'Connor, 1993].

Formally, if the ecological-economic system is not controllable because of uncertainty due both to measurement error and our lack of understanding of the system dynamics, it may still be 'stabilisable'. That is, it may still be possible to protect the resilience of the joint system providing that the uncontrolled (and unobserved) processes of the environment are themselves resilient [in the sense that a bounded input generates a bounded response Holling, 1973, 1986]. System resilience (in this sense) and system sustainability are accordingly closely linked concepts. If it is possible to achieve system stability, it will in general be possible to achieve system sustainability.

The important point here is that ecosystems may be resilient over certain parameter ranges only. Even if such ecosystems are not themselves controllable, therefore, sustainability of the joint ecological-economic system may be assured they are not driven beyond the parameter ranges over which they are resilient. Providing that the ecological system is contained within the threshholds of resilience, it may be 'stabilisable' and its resilience protected. Ecological as opposed to economic sustainability requires that economic activity be constrained within limits given by the local stability of what ecologists term 'essential' ecosystems [Common and Perrings, 1992]. The implications of this for choice of policy instrument are discussed below.

Sustainability and intergenerational equity

The Brundtland Report [WCED, 1987] saw sustainable development as the protection of the productivity of the global system with a view to maintaining the options open to future generations. In other words it saw sustainable development as a condition imposed on the present generation in order to satisfy some notion of intergenerational equity. Much the same notion is explicit in ecological economic discussions of the policy implications of sustainable development [see for example discussions by Pearce et al 1989; Daly and Cobb, 1990; Turner, 1993 all building on the foundational contributions of Solow, 1974, and Hartwick, 1977, 1978]. It is argued that the current generation should compensate future generations for the effects of current behaviour through the transfer of capital bequests — including bequests of biodiversity and other natural capital. More particularly, if inter-generational equity is accepted as an objective (and therefore bequest motivations and value are taken to important) the present generation can assure the welfare of future generations by underwriting the health, diversity, resilience and productivity of natural systems.

The inter-generational equity goal can be achieved by strategies aimed at fostering an ecologically sustainable economy by maintaining natural capital stocks, and in particular by maintaining 'critical' non-substitutable natural capital, the loss of which would be irreversible. While this implies many of the same policy instruments that come out of alternative approaches, it places a premium on precautionary instruments such as safe minimum standards [Barbier, Burgess and Folke, 1994; Bishop, 1978]. It should be noted that this does not signify especially 'risk' averse behaviour. But it does reflect the type of 'risks' being faced — endogenous, correlated 'risks' not easily accommodated in a standard expected utility maximisation approach. Precautionary instruments seek to ensure that irrespective of the actual outcome of current activity, the next generation is left with an equivalent resource endowment (allowing for some trading between different forms of capital) and opportunities for economic activity [Young, 1992, 1993].

There is a sense in which a systems perspective privileges the requirements of the system above its individual components. This is, in fact, a widely held view in ecological economics. But it is not a value free view. It flatly contradicts the principle of consumer sovereignty which privileges the rights of the individual not only with respect to the collectivity, but also with respect to future generations. It affects, for example, the treatment of private discounting. It has long been recognised that society may not choose to discount future costs at the same rate as private decision-makers [Goodin, 1982], but the principle of consumer sovereignty implies that it is the private rate that matters. As Marglin had earlier pointed out, the sovereignty of the present generation of consumers denies any role for the state in securing the welfare of future generations [Marglin, 1963]. If the principle of consumer sovereignty is maintained, all that is open to policy-makers is to persuade individual resource users to take a different view of their own responsibility with respect to future generations. The collectivity has no natural mandate to restrict the consumption choices of individuals. It is the private valuation of resources that matters, not the social valuation. It is the private rate of time preference that matters, not the social rate.

At issue is the perception that if the existence of the component parts of a system is contingent on the health of the whole, as is the case in an ecological system, then it is meaningless to analyse the 'sovereignty' of any one component of the system without identification of the bounds within which that sovereignty may be exercised, and the responsibilities that sovereignty brings. The bounded nature of consumer sovereignty and the responsibility that accompanies it is what lies behind arguments for a new ethic or morality to govern the relation between the individual and the public good [Przewozny, 1991; Regan, 1986; Wilson, 1988]. While many economists remain sceptical of the 'pious sentiments concerning our moral duty' [Dasgupta, 1991], it does seem that society has a choice: either it upholds a principle that compromises the interests of wider society, or it compromises on the principle. Historically, it is the principle that has been compromised. Consumer sovereignty is hedged about with restrictions designed to protect society from the effects of ill-informed, irrational or malevolent individual behaviour.

Ecological economics cannot claim a privileged position on environmental ethics. But the ethics discussed by those who would consider themselves to be ecological economists do tend to differ from those discussed by environmental economists [see, for example, Turner, 1988b; Pearce, 1992; Daly and Cobb, 1990]. Ethics of stewardship, from weak anthropocentrism [*sensu* Norton, 1987] to Gaianism and 'deep ecology' [Turner, 1988a] tend to feature more strongly. The ethical argument at the core of the inter-generational equity proposition is that future generations have a right to expect an inheritance (capital bequest) sufficient to allow them a level of wellbeing no less than that enjoyed by the current generation. It implies an intergenerational social contract which guarantees the future the same 'opportunities' that were open to the past. Such a contractarian approach finds support in the 'Lockean Standard' (each generation should leave 'enough and as good for others'), and also in the neo-Rawlsian 'justice as opportunity' view. The latter holds that the present generation does not have a right to deplete the economic and other opportunities afforded by the resource base. Since the resource base cannot literally be passed on 'intact' what is proposed is that future generations are owed compensation for any reduction (caused by the current generation) in their access to easily extracted and conveniently located natural resources. For justice to prevail future loss of productive potential must be fully compensated [Rawls, 1972; Pasek, 1992; Page, 1982]. In ecological economic systems based on unknown and unknowable technology this is approximated by the resilience of the system [Perrings, 1994].

The relevant value of natural assets in this case is bequest value. Descriptively, bequest value is an intuitively appropriate concept for populations local to an ecosystem or natural environment who enjoy many of its benefits, and want to see their way of life or association with the ecosystem ('a sense of place') passed on to their heirs. The bequest value of the resource is a measure of the satisfaction derived by the present population from passing it on intact. But bequest value may also be derived from the 'contractarian' arguments of Rawls [1972]. The contractarian approach derives principles of justice from the behaviour of rational and risk-averse individual representatives from contemporary society operating from behind a 'veil of ignorance' covering, in an intergenerational context, membership of any particular generation. Mutually agreeable principles of conduct, binding upon all parties, result from hypothetical negotiations between generations. The best known of these principles, the 'difference principle' or maximin criterion, guarantees future generations at least the living standards of contemporary society. The value of the contract may then be thought of as the bequest value of the supporting assets. Application of the principle has been taken to imply that the value of the natural resource base passed on to future generations should be preserved [Page, 1982; Norton, 1989].[2] More recently, it has been shown that a simple intergenerational maximin criterion does not in fact deliver maximum sustainable consumption for future generations. There exist investment strategies that satisfy non-declining utility and that yield a net national product of higher present value than that yielded by the maximin criterion [Pezzey, 1994]. However, it is not yet certain how this relates to the 'difference principle'.

6 Policy implications: environmentally adjusted performance indicators

Concern over the environmental impacts related to the increasing scale of economic activity and support for the sustainable economic development objective have generated considerable interest (in both developed and developing economies) in environmentally adjusted macroeconomic performance indicators. Chapter 8 of Agenda 21 (approved at the Rio Earth Summit in June 1992) calls on governments to "expand existing systems of national economic accounts in order to integrate environment and social dimensions in the accounting framework, including at least satellite systems of natural resources in all member States" (8.42).

Two general approaches toward the development of environment-economic performance indicators have emerged: environmental satellite accounts (in natural resource accounts, pollution emissions accounts and environmental protection expenditure accounts) which are annually produced in additional to conventional GNP accounts; and adjusted GNP or extended monetised accounts [Peskin, 1991].

A range of motivations seem to lie beneath this research work to which mainstream, environmental and ecological economists have all contributed. There is the concern (first highlighted in the early 1970's by the work of Nordhaus and Tobin [1972 with the links between measures of income and measures of welfare [Repetto, 1989; Bartelmus et. al., 1991; Adger & Grohs, 1994]. This led to the development of adjusted GNP measures of economic welfare [Daly and Cobb, 1990]. Hueting [1991] has gone further and attempted to estimate the costs of meeting a range of predetermined sustainability norms.

Other economists have used a capital theoretic approach, combined with Hicksian measures of income, to formulate sustainability indicators and estimates of sustainable national income [Pearce and Atkinson, 1993; El Serafy, 1991; Maler, 1991; Hamilton et al, 1994]. A number of unresolved theoretical and one methodological issues have been debated in this literature. Ecological economists have sought to include the 'critical natural capital' concept and 'constant capital' rule in the indicators debate. Sustainability threshold levels for critical natural capital use, for example,

[2] The same principle is, of course, implicit in the much older concept of Hicks/Lindahl income.

would in effect require a range of supplementary indicators for sustainability. Critiques of this approach have either focused on the inapplicability of a "capital intact" rule (e.g. technical progress can guarantee sustainable welfare in the presence of declining total capital stocks; or the significance of natural capital and its lack of substitutability has been exaggerated [Nordhaus, 1992]; or in the unacceptable welfare effects imposed on the current generation in order to fulfil obligation to future generations [Beckerman, 1992]. The ecological economics riposte is as we pointed out earlier, not to deny the human capacity for invention and innovation but to stress the potential problems that natural systems may face when forced to adapt to changes in demand induced by the technological switches.

More fundamentally, ecological economists argue that none of these environmental-economic indicators are 'proper' measures or indicators of sustainable national income, or sustainable economic development. All of these environmentally adjusted economic indicators only serve to indicate the 'costs of achieving sustainability vis-a-vis the current development path with its existing configuration of capital stocks and assumptions about substitution possibilities. The corrected national income measure is therefore based on measures of income in current prices, and of opportunity costs associated with preserving capital stocks also valued in current prices or shadow prices linked to the status quo situation. What they do give us an indication of is the 'distance' from sustainability that current economic-ecological systems are characterised by, in terms of costs of adjustment. They do not, however, tell us much about the magnitude of the sustainable national income associated with potential sustainable development paths (based on appropriate capital values and prices) if policy makers actually choose such a path [Adger and Grohs, 1994,: Faucheux et al., 1994; Common, 1993].

So the question becomes what utility if any do such indicators possess. Common [1993] concludes that such numbers could obscure rather than clarify issues relevant to the pursuit of sustainability. Rather what is required are more extensive physical accounts and indicators based on existing data sources which link human welfare and ecosystem changes in a relatively straightforward manner given the availability of scientific knowledge. Hamilton et al., [1994] come to more positive conclusions on the basis of a survey of a number of developed and developing countries' practical experiences. They conclude that satellite accounting techniques, especially construction of pollution emissions accounts and, to a lesser extent, resource flow and environmental expenditure accounts, do have real policy relevance. But, on the other hand, the direct policy use of adjusted national accounts aggregates are more limited, the more so as long as the methodological disputes remain unresolved.

7 Policy implications: sustainability instruments

In so far as ecological economics identifies the same underlying forces of environmental change as environmental economics, it supports the same set of economic instruments. Indeed, the set of economic and regulatory instruments discussed by ecological economics overlaps very substantially with instruments discussed by environmental economics, particularly in the treatment of unidirectional externality. There are, however, some differences both in emphasis and in kind. In what follows we consider only those policies and instruments that appear to be treated in a distinctive manner by ecological economics, and that derive from an ecological economics perspective on the underlying physical system.

There are three characteristics of ecological economic systems that call for distinctive policies: the existence of threshold effects, the existence of fundamental uncertainty (including Chichilnisky's endogenous and correlated 'risk'); and the congestible but global public good nature of many ecological functions. The policies indicated by an ecological economics approach to these three characteristics are more cautious than those deriving from a standard approach. The first indicates policies that safeguard the range of options open to future generations by protecting thresholds of resilience. The second indicates policies that minimise the fundamental uncertainty associated with economic activity either by restricting the level of activity to preserve a degree of system predictability or by ensuring that the 'risks' associated with innovative activities/experiements that test the resilience of the system are bounded. Both these two involve microeconomic instruments at the national level. The third indicates policies designed to ensure that environmental effects are taken into account in the international trade regime. It involves institutions and instruments that operate at the international level.

Safe minimum standards

Policies that safeguard the range of future options by protecting thresholds of resilience are generally conceptualised as sustainability constraints [Pearce, 1987; Conway, 1987; Pezzey, 1989; Perrings, Mäler and Folke, 1992; Perrings and Opschoor 1994]. The aim of such constraints is to assure that self-organising ecological systems maintain sufficient stability to enable human societies to adapt to any changes that do take place in environmental conditions. Sustainability requires each generation to maintain the self-organising systems that provide the context and the opportunities for human activitity [Costanza et al., 1992]. The best example of the instruments associated with sustainability constraints are safe minimum standards/quota/limits and their associated penalties. The rationale for safe minimum standards (or other precautionary instruments) in ecological economic systems is clear. The existence of threshold effects involving irreversible loss of potential productivity, and the failure of markets to signal the nearness of such thresholds, both imply the need for instruments that maintain economic activity within appropriate bounds [Costanza et al., 1992; Turner, 1988a]. This is because the component of value least likely to be picked up in market transactions involving threshold effects is user cost — an unsystematic source of error in market indicators.[3] The tendency for specific ecosystems to experience catastrophic and irreversible change when stressed beyond some threshold level is a problem which becomes more acute the more distant the effects, which is why forward markets for environmental resource-based products are so poorly developed.

Safe minimum standards are generally conceptualised as quantitative restrictions. Indeed, the class of instruments to which safe minimum standards belong — including harvesting quota and limits, hunting 'seasons', emission permits, as well as ambient standards — is generally conceptualised in the same way. Since such 'restrictions' have force only to the extent that they are backed up by penalties, the instrument of the policy are the penalties corresponding to the standards. Such instruments are market based, and like others of their kind (taxes, subsidies, user fees and so on) standards and their corresponding penalities work by changing the private cost of resource use. What makes safe minimum standards appropriate in ecological economics is not that they involve quantitative restrictions, but that they involve discontinuous private cost functions that more closely mirror the discontinuities in social costs associated with ecological threshold effects [Perrings and Pearce, 1994].

Sustainability constraints have also been justified on uncertainty grounds. Uncertainty about system boundaries and the effects of scale and thresholds indicates a precautionary approach, and many sustainability instruments have the property that they are precautionary. The degree of scientific uncertainty is such that it is not, for example, possible to specify minimum viable populations and minimum habitat sizes for the survival of many species [Hohl and Tisdell, 1993]. A precautionary approach would protect such species by setting standards involving a significant (though uncertain) margin for error. The implication is that for many ecosystems conservation decisions will necessarily be based on ethical, cultural or political considerations. Indeed, it has been concluded that 'society may choose to adopt the safe minimum standard not because it results from a rigorous model of social choice, but simply because individuals in the society feel that the safe minimum standard is the right thing to do' [Bishop and Ready, 1991].

[3] User costs in this context include losses due to the depletion of biomass and inorganic matter from ecosystems or the disposal of wastes in ecosystems; the loss of ecological services deriving from biogeochemical cycles; and loss of evolutionary potential.

Environmental assurance bonds

A second set of instruments prompted by the pervasive uncertainty of innovative economic activity involves a variant of the long established deposit-refund systems: environmental assurance bonds. The application of deposit-refund systems to environmental protection was initially suggested by Mill [1972] and Solow [1972]. The development of environmental assurance bonds in ecological economics has focused on the private incentives they can offer to research the environmental effects of economic activity in a way that both bounds the potential harm inflicted on society and insures society against such harm [Perrings, 1989; Costanza and Perrings, 1990; Farber, 1991]. Agents undertaking activities for which there exist no precedents are required to post a bond with the environmental authority equal to the 'expected' worst case losses. This indicates the value placed by the environmental authority on allowing the activity to proceed given the current state of knowledge about its wider and longer-term effects. To accommodate the results of research, the bond may be revised in line with experimental or historical data available on the user or external costs of the activity.

A third set of instruments worth mention are those designed to limit access to congestible public goodsWherever the range and probability distribution of the future environmental effects of present activity is known, it is sufficient to require resource users to take out commercial insurance against environmental costs. In other words, bonds should be required of resource users only where the future environmental costs of present activities are commercially uninsurable because the actuarial risks cannot be calculated from historical data. However, since innovative use of environmental resources in a dynamic and evolving system means that fundamental uncertainty is endemic, one would expect that the class of activities for which bonds might be required in a growing economy would be very large.

It is argued that environmental assurance bonds with these characteristics both indemnify society against the potential environmental costs of unprecedented activities, and provide an incentive to both the environmental authority and the resource user to commit additional resources to research activity in proportion to the authority's best estimate of the worst case losses arising out of the use of the resource. The bond is a precautionary instrument in the sense that it imposes the cost of anticipated environmental damage on the resource user in advance. Doubts have been raised about the effectiveness of the indemnity component of environmental assurance bonds based on the experience of performance bonds in the labour market [Shogren, Herriges and Govindasamy, 1993]. A closer common analogue of the indemnity component of the bond would be the housing rental market, but this has not so far been investigated.

While use of the bond for the protection of environmental assets continues to grow, the research incentives it provides have not yet been tested in practical applications. The point about environmental assurance bonds is that they change the private cost of resource use in a way that may or may not align it with the social cost, but which protects the social interest. The reference point in setting the penalty is not social opportunity cost but the marginal net private benefit of resource use [Pearce and Perrings, 1994].

International trade

The argument that the liberalisation of agricultural product markets would have beneficial environmental effects was made very forcefully in the mid 1980s. More recently, the focus of attention has switched to the environmental implications of the liberalisation of trade in general, and the more exaggerated expectations of the effects of liberalisation have been tempered in the process. This is partly in anticipation of the fact that environmental issues are expected to be a significant element in the next round of negotiations over world trade. The main point at issue concerns the potentially contradictory implications of environmental and trade policy.

Considered as an allocation problem, the degradation of environmental resources is inefficient due to externalities. Where the externality cannot be internalised by the appropriate allocation of property rights, the solution lies in the correction of market prices through the use of taxes, charges, fees, regulations together with supporting penalities, and so on. That is, the solution lies in intervention in the price system. The liberalisation of trade policy, on the other hand, implies the removal of barriers to the effective working of markets, including the elimination of taxes, subsidies and protective measures. Driven by the potential efficiency gains from free exchange in competitive markets, the liberalisation of national and international markets alike is, in a sense, 'blind' to externalities.

The potential contradiction between the two shows up in the implications that trade policy is thought to have for the environment, and that environmental policy is thought to have for trade. There are two levels to the debate. At the theoretical level, its is possible to identify a number of potentially contradictory effects of the liberalisation of trade and the internalisation of environmental externalities, giving rise to competing (testable) hypotheses. At the institutional level, the context within which national and intyernational environmental policy is being developed is a pre-existing set of arrangements governing international trade - especially the GATT. Part of the current debate concerns the environmental implications of the GATT as it is currently structured, and the extent to which the internalisation of environmental externalities depends on reform of the GATT.

There are two issues raised by the fact that trade liberalisation is independent of the internalisation of environmental externality, and that the impact of trade liberalisation on incomes is somewhat ambiguous. First, as trade policy directly affects both the volume and location of productive activities, if liberalisation of trade stimulates demand for the products of environmentally damaging activities, then it follows that it will increase environmental damage. Moreover, if the (external) environmental costs of these activities increase by more than the gains to be had from liberalisation, there will be a net welfare loss [Anderson, 1992]. Even if there is no net welfare loss, the resultant pattern of trade will be distorted, for the reason that trade liberalisation does not address the inefficiency due to environmental externalities. The arguments of ecological economists on this point are that the specialisation induced by trade liberalisation has been wholly inappropriate judged on the basis of the social of the use of environmental resources rather than world market prices [Daly and Goodland, 1994; Young, 1994]. In addition, it is claimed that any increase in the overall volume of trade involves an increase in the transportation of commodities. To the extent that transportation is associated with environmental externalities, trade liberalisation will increase environmental demage irrespective of the pattern of specialisation [Røpke, 1994].

A second point concerns the positive impact of trade on welfare (under the compensation principle) providing that the gainers are able to compensate the losers to the point that the latter are better off than they would be under autarky. Indeed, trade liberalisation is generally held to be environmentally beneficial precisely because it raises incomes and, given that environmental protection is in the nature of a luxury good, environmental protection expenditures [Anderson and Blackhurst, 1992]. On this point, ecological economists argue that many countries have become locked in to a pattern of specialisation in export oriented activities, and that the compensation that might make such a pattern of specialisation worthwhile has not been forthcoming. In these cases, it is argued that the income effects of trade on the environment may be perverse [cf Røpke, 1994]. This is the 'Brundtland hypothesis'. Where countries specialise in products for which the terms of trade decline then, in order to maintain foreign exchange earnings, they tend to increase exports through expansion at the extensive margin — bringing increasingly economically marginal, environmentally sensitive resources into exploitation [Pearce and Warford, 1993].

To the extent that the GATT exaggerates such effects, ecological economists have argued for its reform on environmental grounds. While it is clear that environmental and safety regulations have been used to protect local industries against foreign competition where more conventional trade restrictions are outlawed by agreement, there is also evidence that governments tend to impose an excessively lax regime of pollution control in order to enhance the market competitiveness of local industries. Such 'ecological dumping' can induce competitors to impose countervailing tariffs equal to the difference in emission abatement or environmental protection costs between the two countries [Ulph, 1993; 1994]. The position taken by ecological economists on this is that such countervailing tariffs may be optimal and hence allowable under the GATT. There are two areas in which the GATT as currently structured is argued to compromise attempts to protect the environment:

First, the GATT is argued to institutionalise the presumption that the environmental costs of trade liberalisation will be outweighed by the benefits of increased trade. In part, this is argued to be because it does not recognise an important class of externalities. The GATT artificially distinguishes between externality in production (welfare loss caused during the production of a good) and externality in consumption (welfare loss caused by the consumption of that good), and allows only the latter as an exception [Pearce and Warford, 1993].

Second, the GATT prohibits subsidies that make export prices lower than domestic prices, and where a subsidy does exist in contravention to the terms of the Subsidies Code, it allows countervailing duties to be imposed. The implication of this is that the GATT does not recognise the right of countries to impose countervailing duties on countries which implicitly subsidise their exports by overexploiting the environment. The effect of this is that a country is permitted under the GATT rules to protect its own environment, but is denied the right to protect its producers against countries which choose not to protect their environment [Ekins et al, 1994; Daly and Goodland, 1994].

There appears to be a consensus that in order to realise the potential gains from trade without incurring environmental costs, environmental externalities should be addressed directly. That is, there is a consensus that trade restrictions are not the best way of addressing environmental externalities. There is, however, no consensus as to the value of proceeding with trade liberalisation independently of environmental policy reform. The GATT position is that environmental concerns should not be a reason to slow the reform of trade policy, and that trade policy should eschew environmental objectives [GATT, 1992; Anderson and Blackhurst, 1992]. However, given that trade liberalisation may lead to the exacerbation of environmental damage, it is unhelpful to take such position.

The point made by ecological economists is that the degradation of environmental resources reflects market failure — whether due to economic policy, institutional rigidities, or uncertainty — and is socially inefficient. Where it threatens life support systems such inefficiency is also dangerous. The promotion of trade at current market prices irrespective of the external effects of such trade ignores this problem. If environmental effects cannot be addressed directly, a more rational procedure would be to evaluate the welfare gains and losses from trade liberalisation, taking environmental and other external costs into count, and to liberalise only if there are indeed net gains in welfare. The treatment of environmental effects as an entirely separate issue from trade liberalisation is not only theoretically unsustainable, it is also potentially harmful to the interests of producers and consumers alike. The question of whether any environmental costs of trade liberalisation will be outweighed by the gains is an empirical one, and should not be assumed away.

7 Concluding remarks

The treatment of both technology and consumption preferences as endogenous to the economic growth process is a fundamental change that brings economics much closer to ecology. Economics can no longer be seen as the science of the allocation of an arbitrary set of resources amongst the competing uses given by a fixed institutional structure (the subject matter of politics, anthropology and sociology), technology (the subject matter of engineering, physics and chemistry) and preferences (the subject matter of psychology), within an unchanging environment (the subject matter of ecology, geology, hydrology, climatology etc). The constraint set within which economists have traditionally analysed the allocation of resources has become part of the system dynamics. It should be said that a very similar set of observations might be made about each of the disciplines just mentioned. Within organisational theory, for example, it has been argued that insights into the time-behaviour of complex systems deriving from physics and biology fundamentally changes our understanding of how organisational systems evolve over time. Indeed, it has been claimed that the perception of system dynamics as an alternation between long periods of relative stability and short periods of upheaval — called 'punctuated equilibrium'— involves nothing less than a paradigm shift from the existing, gradualist view of organisational change [Gersick, 1991]. It is not clear to us that ecological economics involves a paradigm shift, but it certainly involves a substantive change in the method of analysis, and it has generated new results.

The ecological economics research agenda continues to be less focussed than research agenda in more established fields. It is, however, dominated by questions raised by properties of the underlying physical system. These include the implications of the non-linearity of that system: path dependence, discontinuity, far-from-equilibrium behaviour, and uncertainty. There are two classes of research question prompted by these properties. The first concerns the development of a coherent theory of the dynamical behaviour of ecological-economic systems based on an axiomatic structure that respects the properties of ecological as well as economic systems. The second, centres on problems of the valuation of ecosystem services, and the development of enabling institutions and instruments. The latter agenda is not, of course, exclusive to ecological economics, but the perception of the underlying physical system is. However, as more environmental economists are persuaded of the relevance of this perception, there are signs that the research agenda is being adopted by economists and ecologists with no link to either of the main institutional supports of ecological economics. The intellectual content of the approach may be gaining ground independently of the institutional structures that are ecological economic's most visible sign.

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