Sustainability Assessment: A Review of Values, Concepts, and Methodological Approaches

Issues in Agriculture 10

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CONSULTATIVE GROUP ON INTERNATIONAL AGRICULTURAL RESEARCH

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About the CGIAR

The Consultative Group on International Agricultural Research (CGIAR) is an informal association of 53 public and private sector members that supports a network of 16 international agricultural research centers. The Group was established in 1971.

The World Bank, the Food and Agriculture Organization of the United Nations (FAO), the United Nations Development Programme (UNDP), and the United Nations Environment Programme (UNEP) are cosponsors of the CGIAR. The Chairman of the Group is a senior official of the World Bank, which provides the CGIAR system with a Secretariat in Washington, DC. The CGIAR is assisted by a Technical Advisory Committee, with a Secretariat at FAO in Rome.

The mission of the CGIAR is to contribute, through its research, to promoting sustainable agriculture for food security in the developing countries. International centers supported by the CGIAR are part of a global agricultural research system. The CGIAR conducts strategic and applied research, with its products being international public goods, and focuses its research agenda on problem solving through interdisciplinary programs implemented by one or more of its international centers in collaboration with a full range of partners. Such programs concentrate on increasing productivity, protecting the environment, saving biodiversity, improving policies, and contributing to strengthening agricultural research in developing countries.

Food productivity in developing countries has increased through the combined efforts of CGIAR centers and their partners in developing countries. The same efforts have helped to bring about a range of other benefits, such as reduced prices of food, better nutrition, more rational policies, and stronger institutions. CGIAR centers have trained more than 50,000 agricultural scientists from developing countries over the past 25 years. Many of them form the nucleus of and provide leadership to national agricultural research systems in their own countries.

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Sustainability Assessment: A Review of Values, Concepts, and Methodological Approaches

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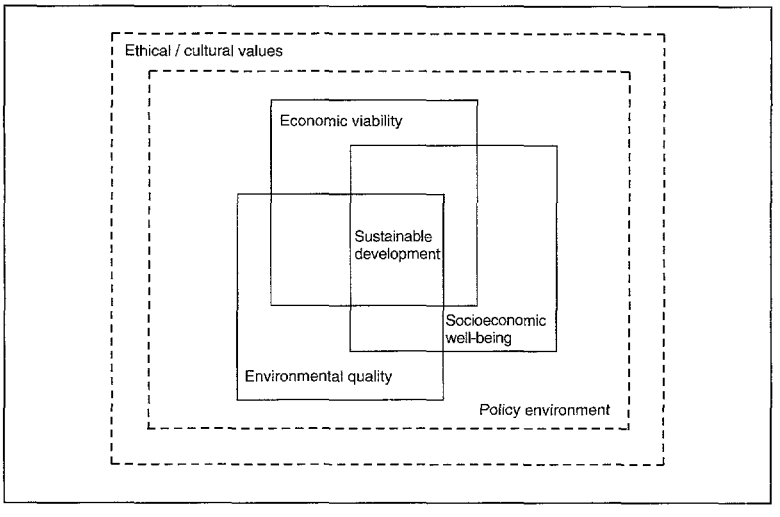
Introduction

Since *sustainable development* became the catchword in international discussions, several approaches to sustainability assessment have been developed. In order to measure or predict the sustainability of a land use system or a society, one must consider the inherent problems of *ex ante* analysis of complex systems. Appropriate scales and time horizons must be chosen; the preconditions and requirements for operationalization and quantification of sustainability must be defined; and the philosophy and value system behind this concept and its translation into policies must be made explicit. On the other hand, the ethical and political convictions behind the multitude of policy recommendations made under the umbrella of *sustainable development* often remain obscure. There is a need to develop criteria that can be used to indicate to what degree strategies and policies contribute to sustainable development.

This paper helps clarify the conceptual background and the implications of the prevalent sustainability paradigm; and the terminology is analyzed to reveal underlying normative philosophical and political perceptions and intentions. To present the interdisciplinary nature of sustainability assessment, a conceptual framework (Figure 1) is proposed that is covered by the disciplines of ecology, economics, and social sciences. All these disciplines are embedded in the policy environment of a society and reflect its underlying ethical and cultural values.

Relating the different approaches to sustainability assessment across disciplines and against the background of the conceptual framework allows us to appraise their relative potentials and limitations. A space and time matrix presents the scale and scope of the different methodologies used for sustainability assessment. The constraints to scientific operationalization of sustainability and to its translation into policy measures, which are revealed by this reference system, highlight the necessity for continued integrated systems research.

Figure 1. Conceptual Framework for Sustainability Assessment



Definitions and Concepts

The importance that the term *sustainability* has gained in international debate can be attributed to its use in the Brundtland Commission's report, *Our Common Future* (WCED 1987), which linked the term to *development.* This report emphasized the economic aspects of sustainability by defining *sustainable development* as "economic development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs." This combination of sustainability and develop-

ment tries to reconcile economic growth in the neoclassical tradition with a new concern for environmental protection, recognizing the biophysical "limits to growth" (Meadows *et al.* 1972) as a constraint to economic development. The term *sustainability* also was used in the CGIAR's mission statement in 1989 to mean "successful management of resources for agriculture to satisfy changing human needs while maintaining or enhancing the quality of the environment and conserving natural resources" (TAC/CGIAR 1989).

Earlier use of the term *sustainability* in ecological and agricultural literature had hardly been noted outside the scientific community directly involved. The term *sustainability* was used in the context of productivity, either as a descriptive feature of ecosystems, "sustainability is the ability of a system to maintain productivity in spite of a major disturbance (intensive stress)" (Conway 1983), or as "sustainable yield" of agricultural crops (Plucknett and Smith 1986).

These definitions have since been expanded to a comprehensive (yet hardly quantifiable) holistic concept (e.g., by the non-governmental organization [NGO] treaty [1993]) in an unpublished draft report "Agriculture is sustainable when it is ecologically sound, economically viable, socially just, culturally appropriate and based on a holistic scientific approach." Although this type of definition has been rejected as too vague by some scientists, it reflects the concern of many environmentalists and development agents to not separate society and environment, economy and ethics (Spendjian 1991). These three types of definitions represent the most common approaches; that is, economic, ecological, and holistic sustainability concepts, which are equivalent to the categories: Sustainable Growth, Agroecology, and Stewardship, as suggested by Harrington and others (1993) and Ruttan (1994).

The concept of sustainability has its roots in forestry, fisheries, and range management. The most commonly agreed upon German equivalent term, *Nachhaltigkeit* (though not identical in meaning and etymology), was first introduced in forestry by the miner von Carlowitz in the eighteenth century (Peters and Wiebecke 1983; Wiersum 1995;

BML 1995) to describe the maintenance of long-term productivity of timber plantations to continuously provide construction poles for the mining industry. This use of the term was driven by the same political interest in economic growth as the World Commission on Environment and Development (WCED) report 200 years later.

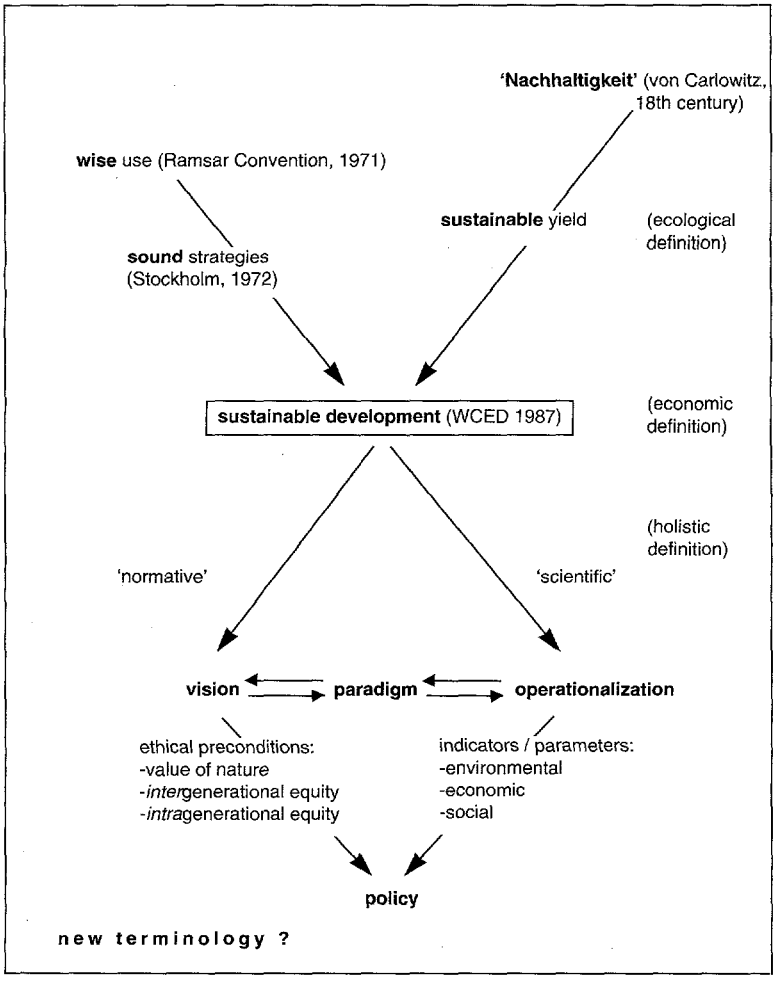
The etymological roots of *sustainability* as a derivation from the Latin verb *sustenere* (= uphold) are discussed by Redclift (1994). This etymology is also reflected in the debate among Spanish-speaking scientists; that is, whether *sostenibilidad* (from *sostener*) or *sustentabilidad* (from *sustentar*) is the more accurate translation. The first term is closer to the passive connotation of "being upheld," while the latter reflects more the active aspect of "to uphold."

These considerations of terminology indicate that there is a strong normative component in the concept of *sustainable development.* This value-driven normative aspect makes sustainable development attractive for policymakers because it permits a direct translation of political objectives into a broadly agreed upon overall concept. However, the normative approach has two severe disadvantages. First, it can be misused for ideological objectives and economic interests that are far from the original ideas of sustainability (e.g., an advertisement campaign of a chemical company). Second, the normative aspect impedes an "objective" or "neutral" scientific analysis of the concept, which is the basic difficulty for scientifically sound sustainability assessment. Thus, a critical analysis of the normative concept of sustainability is required.

Figure 2 shows the relationship between the normative and scientific aspects of sustainable development and the development of the terminology and definitions. On the normative side, two early political documents of the international environmental debate are cited as predecessors of the WCED report: (1) the Ramsar Convention of 1971 on the protection of wetlands and (2) the documents produced following the first United Nations (UN) conference on the environment in 1972 in Stockholm (cf. Wolters 1995). These documents spoke of *wise use* of natural resources and of environmen-

tally *sound strategies*, respectively, terms that were much more obviously normative than *sustainable development* as the overall paradigm for the second UN conference on the environment, held in Rio de Janeiro in 1992.

Figure 2. Normative and Scientific Aspects of Sustainability



The combined term *sustainable development* was coined in the "World Conservation Strategy" of the IUCN in 1980 (Haber 1995), but it never gained paradigmatic appeal before its use and interpretation in the WCED report. Since then, in addition to its political impact, the term rapidly became a new research paradigm in a wide range of disciplines, from the social sciences to biology (cf. Kuhn 1962 and 1969; Norgaard 1989; Vedeld 1994). To be scientifically sound, however, the new paradigm must be operational.

Because the term *sustainability* has recently become somewhat discredited by the obscuring ambiguity of normative and scientific aspects, there is a move to replace it. At first, it seems appealing to return to the term *wise use* for the normative component. However, this term has been usurped by a broad, conservative, anti-environmentalist movement in the United States (Brick 1995). Although a new term may be justified, there is still too little consensus on an alternative. Thus, for the time being, *sustainability* is still the most powerful concept for agricultural research and development, despite its limitations and the potential for misinterpretation.

Based on the three representative definitions, three aspects will be discussed in order to translate normative concepts into scientific categories or, *vice versa*, to detect the ethical values and political concepts behind the (apparently) objective and neutral scientific assessment of sustainability. The first aspect when dealing with the value of the environment is the conceptualization of nature; the second is the temporal dimension of *inter*generational equity; and the third is the spatial or social aspect of *intra*generational equity. These aspects relate to the scientific operationalization of sustainability from ecological, economic, and social points of view, respectively.

Philosophical-Ethical and Sociocultural Considerations

Value of Nature

The current debate on the value of nature as a basic precondition of sustainability assessment can be characterized by two extreme posi-

tions. On one side the environment is regarded as a pool of resources that can be exploited by man for maximum economic prosperity, hence the demand for sustainability to maintain the environment and its productivity. On the other hand, nature is considered a value for its own sake, but is threatened by the increasing human population and the destruction and consumption of natural resources. Both positions need careful analysis to determine their impact on current approaches to sustainability assessment and policies.

Hampicke (1993, 1994) revised the current literature on ecological ethics in view of the economic valuation of nature conservation. Five philosophical concepts are commonly distinguished: (1) theological arguments, (2) a pathocentric position (animal rights movement), (3) biocentric individualism, (4) biocentric holism, and (5) anthropocentric arguments (cf. Birnbacher 1989; Müller 1993).

Different approaches to ecological ethics in the Western world have been developed during the last few decades, as religion lost its hitherto unquestioned uniting power as an ethical principle and as the ecological crisis became apparent (Fraser-Darling 1969). Although, following Hampicke (1993) and Birnbacher (1980), in a liberal society theological arguments can no longer be used as a commonly agreed upon basis for deriving secular laws on environmental protection, they still contribute important aspects to environmental ethics. In contrast to other approaches to ecological ethics, only the theological position regards nature as creation. Regarding nature as creation implies a creator, and thus man's ultimate responsibility is toward the creator instead of toward other creatures as moral subjects (Birnbacher 1980; Hampicke 1993). With respect to sustainability, the belief in a creator implies that only this creator in his sovereignty can sustain his creation. Although this belief does not release man from a responsible behavior toward the creation, it relieves him from the unbearable burden of maintaining the life on earth.

Important concepts in the discussion on sustainable development originated in theological ethics, such as the *concept* of *stewardship* (Fraser-Darling 1969). Similarly, *sufficiency concepts* (as opposed to *effi-*

ciency concepts) as a normative approach for sustainable development have a strong affinity with religious world views (e.g., Sachs 1993). Nelson (1995) pointed out that efficiency, as the foremost economic paradigm of the modern world, has replaced religious principles. In such a neoclassical world view, according to Nelson, "the possibility that consumption should be reduced because the act of consumption is not good for the soul, or is not what actually makes people happy, has no place within the economic value system."

Of the other ecological ethics approaches, biocentric holism is the most relevant philosophy in the sustainability debate, as compared with the animal rights movement or with biocentric individualism. All these philosophies contrast with anthropocentrism in that they consider non–human beings as moral subjects with an intrinsic value. The principle of biocentric holism is summarized by Leopold (1949): "A thing is right if it trends to preserve the integrity, stability, and beauty of the biotic community. It is wrong if it trends otherwise."[1]

Shearman (1990) discussed the dichotomy of anthropocentrism versus non-anthropocentrism applied to the sustainability debate. He concluded that non-anthropocentrism in the sense of Leopold is not a valid condition for sustainability because it is based on an intuitive appeal and lacks rational support. Furthermore, Hampicke (1993) showed clearly that—similarly to theological arguments—biocentric holism cannot be accepted as a common basis for today's liberal society because in its final consequence, it would lead to nondemocratic authoritarian policies for its implementation; in the extreme case it would lead to some form of ecofascism. However, biocentric holism is widely accepted by environmentalists as, for example, shown by Flitner (1995), who analyzed the contributions to the well-known book on

[1] In a drastic form, the principle of biocentric holism and its consequence was expressed by Nietzsche: "*Es sind schon viele Tierarten verschwunden; gesetzt daß auch der Mensch verschwände, so würde nichts in der Welt fehlen*" (quoted by Birnbacher 1989, p. 404). Author's translation: "Many animal species have disappeared in the past; taken the case that man disappeared, too, nothing in the world would be missing."

biodiversity edited by Wilson (1988), with the result that about a third are based on biocentric holism.

Because neither theological arguments nor biocentric holism provide a basis for consensus in today's society, anthropocentrism must be evaluated. This position has dominated occidental philosophy from its beginning (Müller 1993). In particular, Kant's "categoric imperative"[2] is, in agreement with Hampicke (1993), the only common basis for democratic societies. According to Hampicke (1994), it is not necessary to decide if nature can be valued as a good in itself or as an instrument for the benefit of mankind in order to develop environmental policies. Human-centered arguments are sufficient as a point of departure for action (cf. Turner and Pearce 1993; RSU 1994).

The same conclusion—that anthropocentrism is sufficient and is the only common ground for the interpretation of the environment in the context of sustainable development—was clearly expressed in the first paragraph of the United Nations Conference on Environment and Development (UNCED) Rio Declaration:

> *The CONFERENCE ON ENVIRONMENT AND DEVELOPMENT...proclaims that:*
>
> *Principle 1: Human beings are at the center of concern for sustainable development. They are entitled to a healthy and productive life in harmony with nature.*

This statement was strongly opposed by biocentric holistic environmentalist groups during the preparatory process.

Although anthropocentric arguments are the only agreed upon basis for consensus in society, and although the duty toward humanity—as compared with the duty toward a creator or toward nature—is

[2] "Act only on that maxim whereby thou canst at the same time will that it should become a universal law" (Kant 1785).

sufficient basis for policy development, there are still different positions being held with regard to the value of nature as a pool of resources (exploitable) as opposed to the recognition of immaterial values. This is reflected by the scope of monetarization of natural resources and amenities as a basis for economic sustainability assessment.

The concept of limited natural resources (Meadows *et al.* 1972) was taken up in economic theory, which recognized that scarcity or limited availability applies not only to human labor and capital but also to natural resources, including the sink capacity of the environment (Daly 1991; Haber 1994). This recognition gave rise to new economic approaches by "ecological economists" (Costanza *et al.* 1991). However, this approach still is based on the value system of the neoclassical economic tradition; that is, it "rejects the idea that some things are literally priceless" (Nelson 1995).

Although this economic perspective is central to the operationalization of sustainability as a valuation of the environment for present and future use, there are still differences among economists (and ecologists) about how far and in what way the monetarization of natural resources is possible, meaningful, and legitimate. In the extreme case, this may lead to "knowing the price of everything but the value of nothing," as Oscar Wilde stated (quoted by Redclift 1994). This issue was raised recently in CGIAR discussions on water management, when identifying the development of water accounting standards as a priority of the Inter-Center Initiative on Water Management (IIMI 1995).

*Inter*generational Equity

The next ethical issue in the analysis of sustainability assessment, after valuation of the environment, is the demand for intergenerational equity, an entirely new aspect in international debate. This demand, as a duty toward humanity, goes beyond the traditionally accepted kinship care for the next generation (cf. Heinen 1994). Although intergenerational equity lies within the anthropocentric approach, its philosophical justification as a commonly agreed upon basis for society to derive laws and policies may be debated.

Two philosophical principles are used to justify the duty toward future generations. The first principle is again the categoric imperative by Kant. The second is the theory of justice by Rawls (1971) as an extrapolation of Kant's philosophy, which was not developed initially to address intergenerational justice (Hampicke 1994). Rawls presented a hypothetical case in which a group of people design a future society and distribute the available resources with the expectation that they themselves, in their second lives, will have to live in that society without knowing what their social position will be. He concluded from that scenario that such a society will provide, comparatively, the most favorable conditions for their least-well-off members.

Although, at first glance, these two principles appear convincing as policy principles for intergenerational equity, there are two severe shortcomings. First, their implementation cannot be forced by fear of revenge or outbreak of anarchy if the principle is not adhered to because the human beings of future generations cannot retaliate for any injustice done to them. Thus, these principles are more an appeal for moral duty than policy principles because they lack the element of egotism in the sense of Hume (Hampicke 1994). Second, Rawls's model does not take into account the dynamics of society, when in fact social conditions and environmental goods change with time so that the extrapolation of current values to future generations may lead, in retrospect, to undesirable imbalances (Redclift 1994). Thus, both principles, in line with the normative "imperative of responsibility" (Jonas 1984; cf. Christen 1996), serve as moral appeals to the present understanding of "just" resource use, but they cannot be demanded as policy consensus from society if its members do not agree to such principles.

***Intra*generational Equity**

Intragenerational equity as an ethical demand is not a new issue specific to sustainable development; there are numerous publications on this topic in the literature of the social and political sciences. In the context of sustainable development, however, at least three issues need to be discussed:

1. Although the WCED report viewed intragenera-

tional and intergenerational equity as equally important, the spatial and social dimensions (intragenerational equity) tend to be neglected for the sake of the time dimension (intergenerational equity) (Dietz *et al.* 1992; Haber 1995; Hailu and Runge-Metzger 1993).

2. The sustainability debate currently is dominated by the industrial countries, on the normative-political side in the neoclassical tradition to ensure global economic development and on the scientific-theoretical side in the occidental tradition of value judgment and the perception of nature. In a non-western culture, where each species is viewed as a spiritual being and future generations are regarded as spiritual contemporaries, the value of nature and the moral responsibility toward future generations are considered differently than in the occidental tradition. If, in that society, these views are commonly agreed upon, laws and social rules for sustainability policies may be based on this vision. However, given today's political conditions, such local sustainability policies are likely to remain minority exceptions. Redclift and others (1994) showed how local sustainability strategies are suppressed by national and international pressures and policies. Redclift also claimed that re-empowering local populations would result in their becoming stewards of their environment again rather than to remain poachers. These aspects relate to the underlying concept of science as pointed out by Feyerabend (1987), who makes the distinction between "scientific" and "traditional" knowledge (Redclift 1994a).

   If intragenerational equity is taken seriously, the distribution of power needs to be revealed and questioned. Foucault (1981) showed the relationship between knowledge and power for modern sciences

(Redclift 1994a; Flitner 1995), which is clearly reflected by the sustainability debate. *Sustainable development* is defined on the basis of occidental science and value systems. Agarwal (1993) expressed the view that it is "a Western design to keep India backward" and that "in their desperate attempt to survive today, people are forced to forsake their tomorrow and overuse their environment."

3. The issue of power and participation is closely linked to the third relevant issue of intragenerational equity in the sustainability debate: unequal spatial endowment of natural resources, unequal access to these resources, and unequal profit gained from their use (Flitner 1995; Heinen 1994). Haber (1994) used the importation of minerals into the European Union (EU) as an example of the imbalance in resource distribution and exploitation. A recent example of conflicting interests in the use of local resources is the exploitation of petroleum in southern Nigeria, where the local population is denied sustainable management of their environment for the sake of energy provision, primarily to industrial nations, and for the profit of the national government. Unequal distribution of and access to resources also applies at smaller scales; for example, between the rural and urban populations within a country and between men and women at the household level. These conditions must be taken into account when defining sustainability strategies.

   In economic terms this problem is translated into the relationship between local and global needs and strategies, or, as expressed by Costanza (1991), "making local and short-term goals consistent with global and long-term goals." Strategies for sustainable development must fulfill this demand.

Returning to the valuation of nature, such a strategy implies a common value system, which to the economist is implicitly the monetary value of resources. However, monetary valuation may not completely capture the true value of environmental goods, considering the appreciation of non-monetary values (e.g., to members of a reciprocal subsistence culture as opposed to participants in a market economy).

In analogy to Hampicke's conclusion that an anthropogenic approach is sufficient to justify concern for the environment, it can be concluded that consideration of the interests of powerless contemporaries and care for their needs indicates the validity and credibility when claiming concern for future generations. If the rights of and participation by today's powerless are denied in sustainability strategies, then it must be asked if such strategies, supposedly for the benefit of future generations, are not guided by self-interest.

At the same time a portion of self-interest has been shown as a valid common basis for policy development in democratic societies. Neither ethical appeals for altruism for contemporary human beings (and even less for future generations) nor controversial views of nature are valid bases for sustainability strategies. Such policies must be based on (reciprocal) individual and communal benefits, on costs and incentives, and on legislative measures.

Any policy for sustainable development is subject to value judgment. Sustainability assessment is not pure, neutral, objective science, but rather reflects these implicit value judgments. From these considerations, criteria can be derived for decisionmaking among the typical alternatives in sustainability strategies:

- Between the interests and rights of human beings and other species.
- Between the interests and rights of present and future generations.
- Between the interests and rights of different social groups.

From Concept to Measurement

In the preceding section the normative aspects of sustainability and its assessment were pointed out. Only then should its scientific operationalization be considered. Scientific operationalization will be viewed in the context of system theory in order to adequately capture the complexity of sustainability in time and space, and its tradeoffs among different components and aggregation levels. Although it must be taken into account that even apparently holistic system analysis is reductionist in nature (cf. Röling 1994), it is considered the most appropriate "meta-language" (Lantermann 1996) for an interdisciplinary approach to sustainability assessment.

Context of System Theory

Conway (1983) used the term *sustainability* to describe a characteristic of an agroecosystem long before it was used in the context of political development. According to Conway (1985), *sustainability* is a measurable agroecosystem function in addition to productivity, stability, and equitability. This definition and interpretation have been widely accepted for sustainability assessment of agricultural systems, and, therefore his approach will be analyzed in more detail (cf. Tisdell 1988).

Conway uses the term in the sense of resilience, a system's ability to respond to stress. Dalsgaard and others (1995) pointed out that Conway's interpretation of *resilience* is not consistent with its use in ecosystem theory: he relates *resilience* to maintaining productivity rather than to maintaining the structure and patterns of behavior (Holling 1987). Conway deliberately chose productivity trends as the most appropriate parameter for resilience because he

suggests a pragmatic methodology for agroecosystem analysis in a participatory workshop setting. Similarly, the other system properties were selected to facilitate relatively quick and easy analyses of agroecosystems, but not for sustainability assessment (Conway 1993).

His examples of trend analysis use classical yield or price development curves developed from measurements over the last decade or two (Conway 1985). These are typical *ex post* analyses with a short time frame, as he did not intend to discuss the long-term predictability of system behavior. Therefore, his approach cannot immediately be applied to *ex ante* analysis with a long-term perspective.

Ex ante analysis is the most difficult aspect of sustainability assessment. In terms of system theory, these difficulties relate to system properties such as complexity, interrelatedness, nontransparency, and dynamics due to feedback mechanisms, cumulative effects, time lags, or evolution (Dörner 1989).

The predictability or extrapolation of system behavior is possible to only a limited extent. In principle, such predictability requires knowledge of the dynamics of the entire life cycle of the observed components, and needs to cover at least the time span of the life cycles. This is not always possible, however, in *ex ante* analysis of long-term developments. The most relevant experiences with respect to ecosystem management have been gained in fisheries, a typical field for the "tragedy of the commons" (i.e., the conflict between short-term individual interests and long-term common interests). Ludwig and others (1993) reviewed the concept of maximum sustained yield (MSY) for fisheries management based on their analysis of historical statistics. They concluded that this concept encouraged overexploitation of a fluctuating resource due to a "hatchet effect": the lack of limits on investment during good periods, but strong pressure not to disinvest during poor periods, leading to a heavily subsidized industry that overharvests the resource. They concluded that predictions of future events, in particular the effects of global warming and other possible atmospheric changes, are extremely difficult because the time scales involved

are so long that observational studies are unlikely to provide timely indications of required actions or the consequences of failing to take remedial actions.

Wissel (1995) showed the potential of ecological modeling for predicting system behavior. In particular, he pointed out that modeling forces one to clearly define which system component or property is to be sustained and which time frame and spatial scale is to be used as a reference. While Conway (1985) stresses the need to sustain productivity in agroecosystems, environmentalists often argue to sustain species number, composition, or spatial arrangement.

In ecosystem theory it is generally agreed that systems may exist in any one of several stable states, depending on environmental conditions (multiple stability). As a consequence, small changes in factors that influence the system may lead to sudden and strong changes or even collapse of the system. Such sudden changes have been experienced in fisheries when a slight increase in the harvest rate has led to complete destruction of the fish resource (Wissel 1995). Thus, recognizing threshold levels of factors that may cause undesired or irreversible changes to an ecosystem is the greatest challenge in sustainability assessment. (This presumably, however, is not feasible.)

In contrast to Conway's concept of resilience, in which productivity declines slowly before breakdown (with different patterns of recovery), the challenge of sustainability assessment in agroecosystems is to detect hidden stress before it becomes apparent in yield decline or before the system is irreversibly damaged. Thus, for example, if we are to anticipate changes that are not yet apparent in yield decline, yield trend analysis must be complemented by indirect measures of the ecosystem's capacity to respond to stress (e.g., by observing symptoms of change not directly linked to the yield trend, such as species composition or rates of flow and turnover [Becker 1995; Dalsgaard *et al.* 1995]).

Wissel (1995) concluded that modeling, under clearly defined conditions and assumptions, can be used to predict system behavior,

not as a deterministic prognosis but to indicate the level of probability of a system behaving in a particular way. This provides a way to assess the remaining risk of failure at applying certain strategies or policies.

Such *risk* assessment has become common practice in environmental impact analysis over the last decade. However, predictability is limited not only by the risk of a more-or-less well known probability (e.g., the risk of drought in a region with long-term meteorological records), but also by *uncertainty* (i.e., unpredictable events whose effects cannot be assessed, either in quality or in intensity). An example of such uncertainty is the production and release of a new and potentially polluting chemical (Costanza 1993). How uncertainty can be operationalized in economic strategies was discussed by Funtowitz and Ravetz (1991) and Perrings (1991). Going beyond risk and uncertainty, Dovers (1995) included *ignorance* in designing sustainability policies. Ignorance refers to unprecedented effects that cannot be recognized (e.g., the toxic effects of DDT in the first years of its use before observing its harmful consequences on vertebrates [cf. Carson 1962]). Dörner (1989) showed the limitations of human cognition, in particular when dealing with the prediction of complex system behavior. Thus, there are inherent difficulties in *ex ante* analysis for sustainability assessment.

Despite these shortcomings, system theory has proven valid for sustainability assessment. First, it contributes to clarifying the conditions of sustainability. By definition, system theory forces one to define the boundaries of the system under consideration and the hierarchy of aggregation levels. In agricultural land use systems the most relevant subsystems (or levels) are the cropping system (plot level); farming system (farm level); watershed/village (local level); and landscape/district (regional level). Higher levels (national, supranational, and global) influence agriculture more indirectly by policy decisions or large-scale environmental changes (e.g., acid rain or global warming). Izac and Swift (1994) discussed the appropriate scale for sustainability assessment of agricultural systems in Sub-Saharan Africa.

By identifying the system hierarchy, externalities between levels and tradeoffs among components can be traced and explicitly taken into consideration. For example, in an agroecological system analyzed at the farm level, the effects of national policies are externalities as long as they are outside the decision context of the farmer (Olembo 1994). Typical tradeoffs among components within a farming system include unproductive fallow lands in a rotation system for the sake of soil recovery for future use. In resource economics the aspect of externalities has gained great importance in that methodologies are being developed to convert such externalities into accountable quantities (cf. Steger 1995), as well as the assignment of "opportunity costs" to tradeoff effects.

Similarly, the "tragedy of the commons" (i.e., individual use of common resources) can be analyzed adequately only by considering the higher system level to find proper policies for sustainable use (e.g., the case of overgrazing in pastoral societies). Such conflicting interests among different groups—or hierarchical levels of the system—is a typical problem in sustainability strategies. Problem analysis is greatly facilitated by system theory to derive alternative scenarios of future development, depending on the policy chosen.

System analysis, in addition to indicating negative interactions of system components and conflicts among system levels, allows one to detect and describe synergies (i.e., positive interactions) among system components. In agroecosystems, crop mixtures with a Land Equivalent Ratio (LER), i.e. the total land area required in monoculture to produce the same yield as a given area of mixed crop, expressed as a ratio, greater than 1 would be an example of such synergetic effects. Ikerd (1993) pointed out that synergy is a crucial element in sustainable agriculture. To be used for sustainability assessment, however, synergy must be converted into measurable factors. These factors often are approximated by measures of the complexity of the system, which is considered a stabilizing quality. Although there is no immediate correlation between complexity and stability, the contribution of complexity and its synergetic and stabilizing effects on system performance need to be

included in sustainability assessment (Dalsgaard *et al.* 1995; Piepho 1996).

To demonstrate the difficulties of *ex ante* analysis, the deep-sea fishery industry was selected as an example because it is large scale and it has a long-term nature, thus it does not allow experiments with replications in space or in time. There are, however, systems with shorter cycles that can be described or modeled successfully by system analysis (e.g., cropping cycles in agriculture). In such cases, system theory can contribute to the predictability of system behavior, but it is necessary to choose the correct time scale for the purpose (cf. IBSRAM 1994). For example, although Izac and Swift (1994) mentioned the medium-term climatic cycle of 11 years in Southern Africa due to the ENSO (El Niño – Southern Oscillation) effect of 10 to 15 years (Schönwiese 1995; Powell 1995), they arbitrarily proposed a 3-year basis of climatic differences for sustainability assessment.

While Conway (1985, 1993) treated agricultural productivity as a dynamic system property with regard to time, it should also be viewed as a spatially differentiated system property that reflects the resource endowment of the specific site. This includes not only the total amount of nutrients, but also the dynamics of biological activity and the genetic potential of crop varieties. Izac and Swift (1994), therefore, extended the concept of nondeclining productivity in sustainable agroecosystems to byproducts and impacts on ecosystem amenities, beyond harvestable outputs. Haber (1995) showed that intragenerational transfer (the spatially unbalanced resource endowment and unbalanced exploitation and transfer from the poor to the rich) is a potential reason for unsustainability.

Approaches to Sustainability Assessment

To present current approaches to the operationalization of sustainability, the—admittedly artificial but most common—division of economic, environmental, and social indicator concepts as well as composite indicators will be discussed. This is consistent with the underlying definitions; that is, focusing on economic, ecological, and social or holistic interpretations of sustainability.

Two basic approaches to sustainability assessment have been developed:

- The exact measurement of single factors and their combination into meaningful parameters.
- Indicators as an expression of complex situations, where an *indicator* is "a variable that compresses information concerning a relatively complex process, trend or state into a more readily understandable form" (Harrington *et al.* 1993).

The term *sustainability indicator* will be used here as a generic expression for quantitative or qualitative sustainability variables.

The WCED (1987) definition, which focuses on intergenerational equity, and Conway's (1983) definition, which focuses on productivity trends, both concentrate on the dynamic aspect of sustainability over time. Indicators to capture this aspect belong to the group of trend indicators, while state indicators reflect the condition of the respective (eco)system (Bernstein 1992). In developing environmental indicators for national and international policies it has become common practice to distinguish pressure, state, and response indicators (OECD 1991; Adriaanse 1993; Hammond *et al.* 1995; Pieri *et al.* 1995; Winograd 1995). An overview on current sustainability indicators is presented in Table 1.

Economic indicators. For economic trend indicators two basic approaches have been proposed: the valuation of *discount rates* of resource depletion and *Total Factor Productivity (TFP)*. In the neoclassical economic tradition discount rates are derived from the concept of intergenerational equity, or more precisely, from its predecessor concept of limited unrenewable resources (Meadows *et al.* 1972). The concept of discount rates in the context of sustainable development was first proposed by Barbier (1989) and Pearce and others (1990). Assuming a certain "natural capital" stock of a given resource, rates of potential use are calculated to maintain this resource for a given time before its depletion. Pollution rates also are calculated by assuming a given

Table 1. **Indicators and Parameters for Sustainability Assessment**

ECONOMIC INDICATORS	ENVIRONMENTAL INDICATORS
• Modified gross national product • Discount rates -depletion costs -pollution costs • Total factor productivity • Total social factor productivity • Willingness to pay • Contingent valuation method • Hedonic price method • Travel cost approach	• Yield trends • Coefficients for limited resources -depletion rates -pollution rates • Material and energy flows and balances • Soil health • Modeling -empirical -deterministic-analytical -deterministic-numerical • Bioindicators
SOCIAL INDICATORS	COMPOSITE INDICATORS
• Equity coefficients • Disposable family income • Social costs Quantifiable parameters? • Participation • Tenure rights	• Unranked lists of indicators • Scoring systems • Integrated system properties

absorption or sink capacity of the environment and a desirable time horizon. In the last few years the aspect of sink capacity has gained increasing importance as a more serious environmental constraint than resource scarcity.

The concept of discount rates for limited natural resources is based on the assumption that human economy is a subsystem of the finite natural global system (Daly 1991) "in which the limiting factor in development is no longer manmade capital but remaining natural capital" (Costanza *et al.* 1991). Most economists view manmade and natural capital as substitutes rather than as complements. If human

activity is viewed as a substitute for natural capital, then the scarcity of natural capital is weighted in a fundamentally different way. This has led to the distinction of "soft" and "hard" sustainability, the former assuming substitutability of human activity for natural capital, the latter not assuming such substitution.

Among economists the precise determination of discount rates has been an issue of the sustainability debate in the last few years (e.g., Pearce 1990; Spendjian 1991; Steger 1995). This debate will not be repeated here, but it has clearly shown the different ethical values (i.e., the "weight" assigned to the demands of future generations and to the intergenerational transfer of [potential] wealth). Equally, the distinction of "soft" and "hard" sustainability based on the assessment of the substitutability of human capital for natural resources reflects different concepts of the value of nature (van Pelt *et al.* 1995). If, for example, a rain forest species threatened by extinction is viewed as a potential source of a pharmaceutical product, it can be substituted for with a synthetic production of this drug. If it is considered a living being with an intrinsic value, it can never be substituted for by human creativity and action.

The concept of discount rates was developed in the context of global development and recognition of the scarcity of global resources (e.g., petroleum) or the absorption capacity of the atmosphere (e.g., for greenhouse effect gases). Thus, spatial scales are large and time frames are long (Figure 3). The second approach to economic trend assessment of sustainability, Total Factor Productivity (TFP), is at the other end of the space and time scale, at the farm level. It calculates the ratio of the total value of all outputs to the total value of all inputs for a given production system. The TFP approach was first suggested by Lynam and Herdt (1989). A modified version that included soil nutrients and land degradation in the valuation was presented by Ehui and Spencer (1990). The TFP approach has been criticized because it does not internalize external costs, such as environmental effects (Hailu and Runge-Metzger 1993). Herdt and Lynam (1992) tried to overcome this shortcoming by proposing the Total Social Factor Productivity (TSFP) as a more advanced approach than the TFP. The TSFP

included the environmental costs of production, but the question remains about how to value environmental costs appropriately and where to draw the boundary of internalization.

Both approaches, TFP and TSFP, assume steadily increasing production, defining *sustainability* as the "capacity of a system to maintain output at a level approximately equal to or greater than its historical average" and "technology contributes to sustainability if it increases the slope of the trend line" (Lynam and Herdt 1989). From an ecosystem analysis point of view it is clear that no system with finite material resources can grow limitlessly without eventually collapsing. Thus, there is the underlying contradiction of *sustainable growth* in this concept. The same applies to Conway's (1993) upward productivity trend when he referred to *resilience* as sustainability. Recently, a close correlation between the TSFP index and yield has been observed when the TSFP has been used to evaluate long-term agricultural experiments "because most of these experiments have well controlled if not constant inputs. However, on typical farms this would not necessarily be the case because inputs are constantly being adjusted" (Barnett *et al.* 1995). Still, yield trend and TSFP refer to the same spatial and hierarchical level (i.e., the agroecosystem at the farm level) and cover similar time spans.

If human economy is viewed as a subsystem of the global ecosystem where natural resources are considered material assets, then the question of adequate methods and values for their monetarization arises. Only after adequate monetarization can they be internalized in the economic assessment of sustainable development. Different approaches have been suggested: modified gross national products, cost estimates for conservation and rehabilitation measures, and contingent valuation methods.

The Gross National Product (GNP) is the classical indicator of economic development; it counts environmental damages or social rehabilitation as positive values because they increase overall expenses of the national economy. This deficiency has been taken into account in neoclassical economics by adjusting the GNP to create the *Modified*

Gross National Product, subtracting resource depletion and pollution and considering income distribution (e.g., by an "index of sustainable economic welfare" [Costanza 1991]). This does not, however, solve the problem of accounting for nonmonetarized natural resources such as air or aesthetic values.

Valuation of nonmonetarized resources is attempted either by calculating real or fictitious costs of production, conservation, and rehabilitation of natural resources, or by assessing their subjective value to the population. The latter is carried out either by (1) surveys on the *Willingness-to-Pay* or *Contingent Valuation Method* for certain natural resources or amenities or (2) indirect evaluation of their appreciation (e.g., by the *Hedonic Price Method* or *Travel Cost Approach* [cf. Engelhardt and Waibel 1993]). Costanza (1991) questioned the quality of such results in that they do not adequately incorporate long-term goals because this methodological approach excludes future generations from claiming their interests.

Engelhard and Waibel (1993) compiled a list of different approaches to monetarization of natural resources. They favored assigning monetary value to the entire natural environment as an instrument for policy planning and monitoring (e.g., they claim that it is more helpful to say that a spider has a value of US$10 than to assign it an "unappreciable" value). In contrast, under the assumption of "hard" sustainability, Hampicke (1994) concluded that each species is invaluable because of the risk of its extinction, and because it is impossible to predict the monetary value that future generations would assign to it if they chose to monetarize the value of species. Using biodiversity as an example, von Braun and Virchov (1995) showed the potential and limitations of monetarization and economic valuation of the different aspects of biodiversity. They presented a gradient of declining quantifiability, from the direct consumptive value of a species or its products to its potential genetic information for future generations.

Economic sustainability assessment is strongly value-laden. First, it assumes the possibility and right to monetarize all aspects of life and

nature (Nelson 1995). The conversion of sustainability aspects into monetary terms has the advantage that it permits comparison of and calculation with these different quantities in a uniform dimension. On the other hand, it assumes a congruent basis of calculation, which in reality differs considerably depending on the underlying value judgment. Furthermore, monetary values are not sufficiently consistent with ecosystem structure and function, and therefore their aggregation may lead to inadequate environmental policies (RSU 1994).

Environmental indicators. *Yield trend* is the most obvious environmental indicator to assess the sustainability of agroecosystems. However, its suitability must be questioned because yield trend can be assessed either by *ex post* analysis or by modeling. In both cases extrapolation of the results is risky because agricultural systems are not static and because environmental stress is not necessarily reflected by yield trend changes, and sudden collapse may occur (Wissel 1995). Yield trend is highly specific to the site and to the crop variety, and thus is ultimately data intensive. Walsh (1991) and Hailu and Runge-Metzger (1993) pointed out that at least 20 to 25 years of observation are necessary to obtain results of 5 to 10 percent accuracy. Barnett and others (1995) showed the potential and limitations of productivity trends for sustainability assessment under controlled environmental conditions by evaluating long-term agricultural experiments around the world.

The calculation of *discount rates of resource depletion and pollution* can be used as an environmental, as well as economic, trend indicator. In this case, the dimension would not be monetary values but physical units (e.g., tons or parts per million). In contrast to yield trends with small spatial scales and short time spans, this approach is applied mainly for large-scale resource exploitation and (nonpoint) long-term pollution, such as gaseous emissions or global warming. Frequently, these physical calculations are used as a basis for economic valuation, in particular to extrapolate the potential and limitations of industrial development. According to Der Rat von Sachverständigen für Umweltfragen (RSU 1994), physical indicators should be prioritized; monetary indicators should be used as complementary.

The search for environmental indicators originated with industrialization and the accompanying pollution. In the pre-industrial era, canaries were used in mines as an early warning system to detect increases in carbon monoxide so that miners could be evacuated. Canaries belong to the group of *monofactorial bioindicators*, the first generation of environmental indicators. This type of bioindicator includes species that react sensitively to changes in their environment (e.g., lichens to increased SO_2 or heavy metals, or tobacco to increased ozone in the atmosphere). With increasing environmental awareness and experience the use of ecological indicators was further refined. The second generation of ecological indicators focused on *ecosystem dynamics*, on the structure and function of entire ecosystems. Parameters for stress/response assessment were developed, and chemical compounds and processes or metabolic products were measured and quantified. This ecosystems approach included the assessment of values such as "purity of nature," "amenities," or "ecosystem integrity," as expressed by the "Index of Biotic Integrity" (Regier 1992). The latter ecosystem properties have strong normative components that require clearly defined value systems. A comprehensive review of the different approaches to ecological indication is given in McKenzie and others (1992).

Only recently has identification of ecological indicators been discussed in the context of socioeconomic and cultural conditions (Rapport 1992); that is, the growing environmental movements in industrial nations "discovered" sustainability as a concept for environmental quality assessment. Thus, the concept of ecological indicators merges with the concept of sustainability indicators (which have come into the debate in the context of international development) into one coherent concept of global relevance (Becker 1995).

In this sense environmental indicators for policy planning and monitoring comprise ecological indicators of pollution (first generation), of ecosystem structure and function (second generation), and of socioeconomic aspects (third generation). Thus *ecological* indicators in the narrower sense are ecosystem descriptors, while *environmental* indi-

cators have a more general meaning, are policy-oriented, and are not necessarily based on ecosystem analysis.

Recognizing the need for such policy-oriented indicators at the national and international levels, the Organisation for Economic Co-operation and Development (OECD 1991) compiled a list of some 20 indicators, defined as pressure/state/response indicators. For example, on an *ex post* data basis, the OECD proposed emission rates of key gases to assess atmospheric pollution, nitrate concentrations for river quality assessment, and numbers of threatened species. These biophysical aspects are complemented by socioeconomic indicators, such as population data or energy intensity and supply. These indicators are clearly rooted in the tradition of the assessment of environmental damages by industrialization, oriented toward negative tradeoffs of past and present developments. They are not guided by an objective or target future sustainability scenario. The OECD list is driven by data availability rather than being designed on a solid theoretical basis (RSU 1994).

Target-oriented approaches to environmental sustainability assessment for national policymaking have been developed in the Netherlands. Although among the most advanced approaches to sustainability assessment for actual policy planning, they are based on static concepts and do not precisely reflect ecosystem behavior ("quick and dirty indicators," Kuik and Verbruggen 1991).

Material and energy flows are important ecosystem properties and have been recognized for their relevance to sustainability assessment and policy development (e.g., Daly 1991). Henseling and Schwanhold (1995) proposed material flow management as a strategy for sustainable development. Although it appears directly derived from ecosystem theory, it is a pragmatic approach related to industrial production and trade, similar to *Production Chain Analysis*, as a monitoring instrument for material flows (Meissner 1993; RSU 1994).

In contrast to these not precisely ecological environmental indicators, the RSU (1994) claimed that it was necessary to orient indica-

tor selection on ecosystem theory. The RSU proposed indicators to reflect critical levels, critical loads, and critical structural changes at different levels of the ecosystem hierarchy (e.g., the frequency of ozone levels beyond a defined threshold, pesticide concentrations in rivers, or habitat integrity). They admitted, however, that the operationalization of such indicators still requires considerable research, and therefore they proposed to use the most sensitive components of an ecosystem as stress indicators. Identifying stress symptoms in the most vulnerable and sensitive ecosystem element presents an early warning for the entire ecosystem. Thus, these indicators are representative of the complete cause/response chain in that particular ecosystem. Becker (1995) showed how indicator plants may be used to detect agroecosystem change in a way that complements analysis of productivity trends.

Social indicators. Social indicators are intended to translate aspects of intragenerational equity into measurable quantities, or at least into operationalized terms. However, approaches to quantification and operationalization of social dimensions must be carefully restricted to those aspects that can be described meaningfully by numerical or analytical tools and methods (Hardin 1991).

The most direct *quantification of equity* involves calculation of the distribution of wealth within a society. Two similar parameters have been proposed: the less common Herfindahl Measure for absolute concentration and the Gini coefficient for relative concentration, derived from the Lorenz curve (Conway 1993; Müller 1995). The Gini coefficient tends toward zero in a perfectly egalitarian society and toward one at increasing inequality (Izac and Swift 1994). Winograd (1995) used the Gini coefficient to list the agricultural land concentration in Latin American countries and to show how it has changed over the last three decades.

At first, such a numerical expression seems convincing as an indicator of equity. However, it has several shortcomings that must be considered. From a mathematical point of view, the Gini coefficient tends to give relatively greater weight to changes in different parts of the

range (Conway 1993). More important, it is based on a static perception of social and cultural values and conditions, and pretends total uniformity of people, which is obviously not valid. People differ in their endowment and in the way they use and appreciate their resources (cf. Nelson 1995), and they are conscious of social justice. Conway (1993) therefore preferred Atkinson's weighted index of income distribution. Huber (1995) made the distinction of social justice based on need, on performance, and on property as different dimensions of equity. These differences are not taken into account in static, target-oriented sustainability policies (e.g., Levi 1995).

In early economic theory, as conceived by Pareto (1935), economic optimality included the maximization of economic efficiency along with the maximization of social welfare (Izac and Swift 1994), while in neoclassical economics the focus was more on economic efficiency only (Nelson 1995), separating it from social welfare. Taking into account that these two aspects are different sides of one coin, *economic efficiency* measures have been proposed as social indicators as well. Two common approaches are *disposable family income* at the household level and the calculation of *social costs* (Hailu and Runge-Metzger 1993). Apart from the level of wealth (or poverty) and its distribution across the society, two other aspects are particularly relevant for social sustainability assessment: legal aspects of *land tenure* and *participation.* Although land tenure can be expressed by the Gini coefficient, its assessment in different societies with varying cultural and legal practices is a complex issue that cannot easily be operationalized.

From a social science point of view, participation is a central element of sustainability (Busch-Lüty 1995). Participation, however, is difficult to translate meaningfully into quantitative terms as a social indicator. In particular, with respect to sustainable development, Lélé (1991) pointed out that participation does not ensure equity in resource use, which is fundamentally tied to the land reform issue; that is, tied to land tenure (Adger and Grohs 1994).

Participation has also been demanded as an instrument of Environmental Impact Assessment (EIA). Since *ex ante* analysis of

potential impacts of planned projects on the environment is difficult, participation is intended to reduce uncertainty by intrasubjective judgment; furthermore, participation increases the transparency of the decisionmaking process (Meissner 1993). Recent experience with participatory EIA of release trials of genetically modified crops has revealed, however, that the power structures in this process and the subsequent decisionmaking could not be overcome by integrating NGO groups into the process (Stoeppler-Zimmer 1993).

Social indicators refer to all levels of the spatial hierarchy, with varying degrees of relevance. Equity can be assessed within a family or across a village community as well as among different countries. The temporal dimension of these social indicators is of comparatively minor importance.

Composite indicators and systems approaches. None of the single indicators presented above can adequately represent the level of sustainability of an ecosystem or society. Therefore, several ways of combining or aggregating indicators have been proposed. Three basic approaches can be distinguished: unranked lists of heterogeneous indicators, scoring systems with unified dimensions, and system descriptors.

The first, and simplest, approach to sustainability assessment by composite indicators are *lists of indicators* without the intent to aggregate or unify them into a single dimension, and without assigning different weights to the different components. The common lists of environmental indicators belong to this group, as proposed by, for example, the OECD, the World Resources Institute (WRI), the United Nations Development Programme (UNDP), the World Bank, or the Food and Agriculture Organization of the United Nations (FAO) (Müller 1995). Such lists of indicators have been used in recent attempts to apply the information gained from the scientific sustainability discussion to policy development (e.g., RSU 1994; Winograd 1995; Bund and Misereor 1996). The advantage of these lists is their transparency. Because they are predominantly action oriented for practical policy, they are based on available data and on the normative

principles of the generating institution, while the scientific operationalization of sustainability is less obvious in the choice of components. The definition of threshold values of resource exploitation or pollution is often interest driven rather than based on "hard" scientific knowledge of resource availability or scarcity (e.g., the target to reduce petroleum consumption of cars to three liters per hundred kilometers).

Scoring system approaches combine different components of the "sustainability complex" into one measure; the components may be given different weights according to the objectives or preferences of the authors. Examples of scoring systems at the local level are the Sustainable Livelihood Security Index (SLSI) defined by Swaminathan (1991) and the Agroecosystems Analysis Framework (e.g., Tabora 1991). The SLSI combines ecological, economic, employment, and equity factors by weighting three components: carrying capacity (human and animal), number of economically active adults in the village, and the degree of female literacy and employment. The SLSI thus is a subjective measure of sustainability, or, more precisely, a measure of "sustainable food and nutrition security at the household level," as defined by the author. The Agroecosystems Analysis Framework is a scoring system that distinguishes biophysical aspects, economic and social impacts, and policy and administrative aspects, subdividing each category into four or five components with a certain score as a basis for comparison of different land use systems. This method, although more detailed than the SLSI, also is subjective in its choice of components and weights of scores. In the Netherlands, national-level scoring systems have been proposed (cf. van Pelt *et al.* 1995) in which indicators similar to those in the unranked lists are calibrated to uniform dimensions by target values (e.g., in the AMOEBA approach [ten Brink 1991]). The problem with scoring systems is that they pretend objectivity and uniformity, while the choice of components and their assigned weights is highly subjective, and the aggregation of different spatial, temporal, and sectoral dimensions is often not meaningful (cf. Müller 1995).

System-based approaches apply strict rules of system theory to select a number of *system properties* as sustainability indicators, and a

set of rules that specify how to integrate them into a meaningful assessment of the system's sustainability. Examples of this approach, which are described below, include the quantification of ecological sustainability in farming systems analysis by Dalsgaard and others (1995) and the system-based operationalization of economic indicators at different spatial scales by van Pelt and others (1995).

Dalsgaard and others (1995) selected four system properties that they considered crucial for sustainability—*diversity, cycling, stability,* and *capacity*—and they explicitly explained their selection criteria based on ecosystem theory. Their methodology was applied in a participatory process with local farmers for sustainability assessment of local agroecosystems. They based their calculations on parameters that can be easily measured by the farmers themselves, but they used sophisticated mathematical models to convert the results into sustainability measures. Although the underlying assumptions used as the basis for selecting the four system properties could be debated (e.g., high species diversity is not necessarily correlated with sustainability, as indicated by stable, but species-poor, natural ecosystems), the authors' explicit ecosystem-based approach of critical system properties is a promising step toward sustainability assessment. This method, however, is currently restricted to the local level, and does not explicitly consider economic or social aspects. Beyond that, it is based on the assumption that the chosen system properties of agricultural systems follow the same patterns as in natural ecosystems. Nor did the authors explicitly assess human intervention in farming systems. The strength of this approach lies in its systematic selection of system properties, and less in the applied set of rules for their integration. Its focus is on state indicators of system conditions, without considering trends over time. It is intended for spatial system comparison rather than for analyzing the dynamics of the resource base in one site.

Van Pelt and others (1995) used discount rates as basic parameters, considering the substitutability of natural resources by distinguishing between "hard" and "soft" sustainability. Identifying the normative decisions in this distinction (intergenerational equity) as well as the conditions of social welfare (intragenerational equity), they developed

an algorithm to derive policy decisions based on an integrated model of intra- and intergenerational equity for different spatial levels in a hierarchical order (multicriteria analysis [MCA]). They discussed the relative advantages and disadvantages of ecological sustainability indicators proposed by other authors. The strength of this approach lies more in the set of rules used to integrate the different aspects of sustainability into a coherent system than in the selection of sustainability indicators. The MCA model does not, however, consider the interaction between different system components. The model is clearly based on trend analysis with less emphasis on detailed description of system conditions.

In both examples the final results are expressed as a single figure (i.e., in the same way as in scoring systems). The difference between the two approaches lies in the system-based selection and integration of the single components. As for the other scoring techniques, final figures are most easily interpreted by the authors themselves, or they may be used as a relative measure for system comparison, as suggested in both examples.

Ecosystem health. Even before *sustainability* became fashionable, environmentalists had proposed "ecosystem health" as a concept for managing environmental resources. Since then, the usefulness of this approach to assess environmental quality has been subject to debate (Shrader-Frechette 1994).

There are strong arguments that favor such an approach; the most obvious is its intuitive appeal, which makes ecosystem health particularly attractive for policy recommendations. Proponents of this concept (e.g., Rapport 1992; Costanza 1991) stress its holistic perspective based on a positive vision instead of focusing on single degradation symptoms.

Rapport (1992) compared the three generations of ecological indicators to different stages of diagnostic experience in human health. According to his interpretation the first generation, monofactorial indicators, focused on "clinical signs" of environmental degradation. The

second generation, ecosystem structure and function assessment, detected "preclinical signs and symptoms" of ecosystem breakdown. The third generation, which sought to establish connectivity between ecological considerations and economic and social factors, tried to define a larger and more appropriate context for assessing the health of the environment. He proposed three requirements for future ecological indicators: early warning by single monofactorial indicator species and forecasting models, groups of indicator species to reflect multiple stress, and indicators for ecosystem health based on properties of mature ecosystems, such as increasing complexity, feedback, and so on (cf. Becker 1995).

Despite the advantages of intuitive appeal, holistic perspective, and positive vision, the concept of ecosystem health must be applied with care in sustainability assessment. The term *ecosystem health*, like the term *sustainability*, is a strongly value-laden and normative concept. As *human health* may be defined differently by those with different perspectives and by different interest groups (e.g., the smokers' lobby or a health insurance company), the meaning of *ecosystem health* is neither precisely defined nor unanimously accepted by the scientific community.

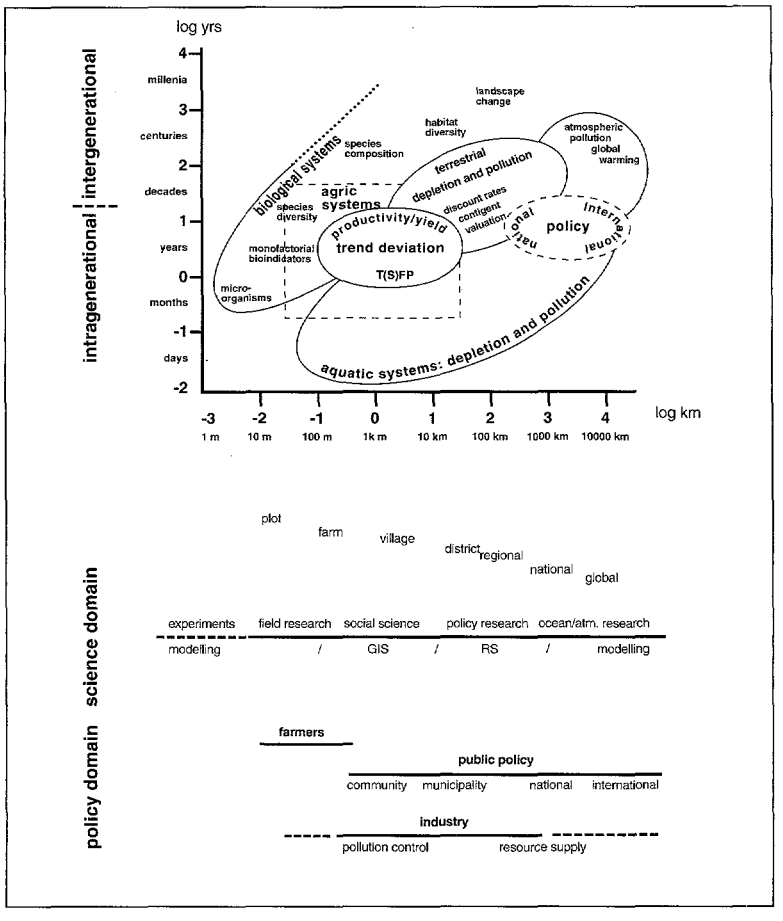
By origin, the ecosystem health concept has a strong affinity to Leopold's holistic biocentrism. Norton (1991) spoke of "metaphysical holism" when ecosystem health is defined in the sense of biocentrism, and he distinguished it from "contextual holism" as an ecosystem health approach based on system theory. Its use in the latter context is based primarily on resilience as the most relevant system property. Wolman (quoted by Shrader-Frechette 1994) suggested that resilience be complemented by categories such as persistence, equity, attention to scale, and compassion, but this combination of categories leaves the domain of system theory, in particular by introducing compassion as an ethical aspect. Operationalization of ecosystem health presents the same problems as the operationalization of sustainability. Compared with the definition of *human health* as a basis for medical intervention, the definition of *ecosystem health* for environmental management is even less clear. In contrast to human health, which is centered on the

individual patient as the target "system," the boundary and hierarchical level of the "patient" ecosystem is less obvious and there is less agreement on it. Therefore, Norton (1991) pointed out that it is necessary to define the proper scale for ecosystem health using a system theory approach ("contextual holism"). Allen (quoted by Shrader-Frechette 1994) emphasized that it is necessary to explicitly define the spatial and temporal boundaries of the respective ecosystem.

Thus, all that has been discussed above that pertains to sustainability assessment also applies to the concept of operationalizing ecosystem health. Barnett and others (1995) therefore concluded that "good indicators of ecosystem health are good indicators of sustainability." They proposed soil properties such as erosion, organic matter content, pH, phosphorus, potassium, micronutrient availability, microbial respiration rate, and bulk density as ecosystem health indicators to assess the quality of the resource base. They pointed out, however, that such indicators, which are based on experience, do not permit precise conclusions and that a "soil health index" would require further study before validation.

In this sense, the ecosystem health approach is valuable for sustainability assessment because it complements "hard" system analysis with "soft" heuristic data and indicators, which are derived from experience and observation of symptoms. Somerville (quoted by Shrader-Frechette 1994) pointed out that ecology is not only science but also art, and that therefore ecosystem health is an appropriate approach to reflect the art aspect of environmental management. The combined emphasis on art and science is taken for granted in medical diagnosis and treatment; equally, ethical aspects have always been an integral part of medical science. Thus, sustainability assessment carried out using the ecosystem health concept may benefit from the experience and knowledge acquired in the political debate on human health. The controversial debate on the consideration of "alternative" medical approaches, on the economic value of human life, organs, and medical services, as well as on technical, mechanistic versus holistic medical treatment may be viewed as a source of insight for the debate on ecosystem health and sustainability.

Figure 3. Space and Time Matrix for Sustainability Assessment



Operationalizing Sustainability Assessment in Space and Time

Sustainability indicators can be divided roughly into state and trend indicators and have specific spatial scales and time horizons. Figure 3 presents current sustainability indicators in a space/time frame; a logarithmic scale was chosen so that the entire range of local to global dimensions and of short to long durations can be shown.

Although short time spans (i.e., hours or less) are relevant for biological and physical processes (e.g., soil biology or sudden point-source pollution), sustainability assessment is focused on longer time periods. Thus, these fast processes are not covered by this diagram, although they may enter into the modeling of system components. The dashed lines indicate agriculture and policy, two realms of human activity that are central to sustainability research. Agriculture comprises spatial scales from plot-level and small-scale gathering to agribusiness and large plantations, and temporal scales of short-season crops to perennials and long rotations in swidden agriculture (cf. Fresco and Kroonenberg 1992; Fresco 1994).

The diagram shows that the two approaches central to the assessment of agroecosystem sustainability—yield trend deviation and Total Factor Productivity—are very limited in scale and scope. In particular, they cover time spans of only one or two decades. Thus, they do not assess intergenerational change, which, according to the WCED definition, is the principal criterion for sustainability. Similarly, policy measures have a temporal scope of at most one to two decades, but generally much less (e.g., one legislative period). The most prominent approach derived from the concept of intergenerational equity is to use discount rates of resource depletion or absorption capacities for pollution. This concept can be applied to trend assessment of abiotic, physical, and terrestrial resources and to the assumed sink capacity of aquatic, terrestrial, and atmospheric systems over intergenerational time periods.

Only recently has the importance of biodiversity for sustainability been recognized. The focus on the Biodiversity Convention at the United Nations Conference on Environment and Development (UNCED) can be viewed as a reflection of this new awareness. This strong interest in biodiversity as a crucial element for sustainable development was generated at UNCED mainly by the industrial countries, not only for conservationist reasons but also, and primarily, for economic reasons; that is, to maintain genetic resources for future use and exploitation. Redclift (1994) pointed out that the UNCED follow-up agenda of the Global

Environmental Facility (GEF) is dominated by the interests of industrial countries, one of which is to halt genetic erosion. Flitner (1995) documented how the history of the conservation of plant genetic resources, since their "discovery" a century ago, has been driven by economic interests.

Although research on biodiversity has been high on the scientific agenda for the past decade (e.g., Wilson 1988; Schulze and Mooney 1993; Gaston 1996), the link between biodiversity and sustainability assessment is still weak. Biodiversity has an ambiguous role in sustainability assessment. On the one hand, it is in the focus of what needs to be sustained. On the other hand, biodiversity is proposed as a means to assess the sustainability of complex systems.

Considering the threat of irreversible species extinction, the conservation of biodiversity is the greatest challenge for sustainability strategies with the longest-term implications. For sustainability assessment focusing on biodiversity, the Center for International Forestry Research (CIFOR), *inter alia*, suggest to distinguish threat assessment and sensitivity analysis: the (static) assessment of threat to genetic resources *in situ* and the (dynamic) analysis of the impact of changing conditions (e.g., land prices, access to markets, etc.) on the level of threat (Boyle 1995).

With regard to the use of biodiversity as a parameter in sustainability assessment, several approaches were presented above. Single species are used as monofactorial pollution indicators; groups of species are used to reflect multiple ecosystem stress. Applied to plant communities, such observations may include phytosociological, structural parameters that go beyond simple counting of species numbers (Becker 1995). To assess sustainability at the local level it may be necessary to analyze biodiversity below the species level (i.e., intraspecific diversity). Patterns of phenotypical or genotypical differences and their relation to site conditions may be indicative of forces that determine if a system is sustainable. Diversity at the variety level of crops can indicate sustainable land use, reflecting ecological site conditions as well as the socioeconomic conditions of farmers (de Boef *et al.* 1993).

Species numbers are the basis of biodiversity for sustainability assessment as proposed by Daalsgard and others (1995), OECD (1993), and Winograd (1995). Daalsgard and others (1995) use the Shannon Index, while Piepho (1996) pointed out that this combined index may be a misleading parameter for biodiversity in agricultural systems. OECD (1991) used the percentage of threatened species in relation to all species of a country, admitting, however, that these data are not standardized internationally. Winograd (1995) complemented the number and ratio of threatened and rare species per country with the Species Risk Index, which is defined as the number of endemic species per unit area multiplied by the percentage of loss of original area (Reid *et al.* 1992).

Loss of habitats by conversion into agricultural land or by intensification of production is the major cause for extinction hazard (OECD 1991). The RSU (1994) therefore concluded that the assessment of structural landscape change should be a crucial element in the operationalization of sustainability. The decision about which landscape structure is to be sustained is highly culture dependent and value-driven. Apart from regions with extremely adverse conditions for human settlement, landscape patterns have been created by mankind over millennia. These regions have undergone "creative destruction" (cf. Schumpeter quoted by Fritz *et al.* 1995), which in many cases has increased their biodiversity (e.g., in the Amazon forest, Posey 1985). Thus, in order to maintain this habitat diversity, nature conservation alone is not sufficient. Strategies must include a consensus on the objectives of conservation in a dynamic process as well as use of appropriate conservation methods, including human action. Ultimately, the "integrity of nature," as claimed by biocentric holism, would change manmade landscapes into a less diverse pattern.

In Figure 3 it was assumed that extinct species are unsubstitutable ("hard" sustainability), leading to time horizons on the order of magnitude of evolutionary processes. If substitutability of the value of species were assumed ("soft" sustainability), the temporal scope for their replacement would depend on the creativity of man (i.e., on technical progress).

The difference in temporal scope and speed between biological and technical processes is increasingly being recognized in the sustainability debate (Gowdy and Daniel 1995). The underlying hypothesis is that technical progress occurs much too quickly to be adjusted to biological cycles, which ultimately govern all life and development on earth. Bund and Misereor (1996) thus concluded that it was necessary to demand "de-acceleration" (*Entschleunigung*) of development (cf. Gronemeyer 1993).

Criteria for Indicator Selection

Table 2 presents criteria for the selection and evaluation of sustainability indicators. The first demand on sustainability indicators is their scientific validity (BML 1995). Bernstein (1992) demanded that "the ideal trend indicator should be both ecologically realistic and meaningful and managerially useful." These two key properties should be complemented by the requirement that appropriate indicators be based on the sustainability paradigm (cf. RSU 1994). This last property explicitly introduces the normative element, guiding selection of the indicator according to the value system of the respective author, institution, or society.

The requirements for sustainability indicators cover a broad range of aspects, not all of which can be equally met. This is most obvious in the category of ecosystem relevance (second column of Table 2). The first set of criteria in the second column (Ecosystem Relevance) describe desirable properties to determine the sustainability of ecosystems. However, they are not explicit with regard to system theory; thus, they still require operationalization. The second group of indicators in column 2 is based on system theory; however, they cannot necessarily be put into practice (e.g., large scale and long-term effects, such as global warming, are beyond the scope of experimental evidence from full system cycles). Thus, a balance must be found between scientific accuracy and pragmatic decisionmaking. Table 2 can be used to evaluate existing indicators by applying a matrix approach. Selected criteria of all columns can be matched with the indicators to be evaluated by assigning *yes / no / not valid* scores to each

Table 2. Criteria for the Selection of Sustainability Indicators

Scientific Quality	Ecosystem Relevance	Data Management	Sustainability Paradigm
• Indicator really measures what it is supposed to detect • Indicator measures significant aspect • Problem specific • Distinguishes between causes and effects • Can be reproduced and repeated over time • Uncorrelated, independent • Unambiguous	• Changes as the system moves away from equilibrium • Distinguishes agroecosystems moving toward sustainability • Identifies key factors leading to unsustainability • Warning of irreversible degradation processes • Proactive in forecasting future trends • Covers full cycle of the system through time • Corresponds to aggregation level • Highlights links to other system levels • Permits tradeoff detection and assessment between system components and levels • Can be related to other indicators	• Easy to measure • Easy to document • Easy to interpret • Cost effective • Data available • Comparable across borders and over time • Quantifiable • Representative • Transparent • Geographically relevant • Relevant to users • User friendly • Widely accepted	• What is to be sustained? • Resource efficiency • Carrying capacity • Health protection • Target values • Time horizon • Social welfare • Equity • Participatory definition • Adequate rating of single aspects

Sources: Bernstein (1992); BML (1995); Hamblin (1992); Harrington *et al.* (1993); Meissner (1993); Müller (1995); Pettit and Sarwal (1993); RSU (1994); Van Keulen (1993).

(cf. RSU 1994). This can help guide the selection of indicators according to the purpose of the study.

In Figure 3 most methodologies important to sustainability research are related to the spatial scale. From an ecological perspective it can be concluded that biodiversity research, application of geographic information systems (GIS) and remote sensing for determining spatial characteristics, and modeling to assess the temporal dimension will become the dominant fields of sustainability research. Small-scale models will likely cover full system cycles with more or less well-researched determinants for temporal extrapolation (e.g., crop growth models), while large-scale models will in most cases cover long time spans, and therefore will need to incorporate risk assessment based on probability assumptions (cf. Ludwig *et al.* 1993; Wissel 1995; Gowdy and Daniel 1995).

From a social science point of view, Hailu and Runge-Metzger (1993) identified the following methodologies, but they did not relate them to spatial dimensions: (1) expert guesses (inter alia to capture uncertainty): Delphi techniques, consultations, and intelligent guesses (Meissner 1993 complemented this category with heuristic approaches); (2) experiments; (3) field surveys: social science, aerial surveys and remote sensing, direct field observations; and (4) modeling: empirical models, deterministic-analytical models, and deterministic-numerical models.

From Sustainability Assessment to Sustainability Policy

Ideally, policy decisions should coincide with the results of scientific analysis; however, because of conflicting interests this is frequently not the case. On the one hand, policy decisions are embedded in a cultural environment that is shaped by rational and by irrational traditions, by ethical consensus and by discourse (Figure 1). In a modern society, on the other hand, policy decisions are based on scientific analysis, apart from the normative aspects (Figure 2). Thus, relating

scientific sustainability assessment to the cultural, ethical, and political context, it becomes apparent that scientific sustainability assessment can contribute significantly to policy decisionmaking, but that it is not the only basis for such decisions.

Scientific analysis of sustainability and the selection of valid indicators serve as tools for rational decisionmaking and evaluation. Figure 3 shows the three major "clients" for sustainability indicators in the policy domain: farmers at the local level, industry managers at the local level to assess the environment of their plants and the markets at the regional to global levels, and public policymakers (Hamblin 1992; Harrington *et al.* 1993). They all need sustainability indicators for effective decisionmaking in space and time: between "here" and "there" and between "now" and "later" (Costanza 1991; Redclift 1994).

Dovers (1995) presented a general framework for scaling and framing policy problems in sustainability. He proposed a heuristic list of attributes for problem-framing and response-framing. Problem-framing is based on scientific sustainability assessment, while response-framing in the first place comprises normative and political criteria.

Apart from criteria linked to spatial and temporal impacts, Dovers (1995) included the degree of complexity and connectivity as a criterion for problem-framing. Similarly, the RSU (1994) pointed to the "retinity principle" (i.e., the interaction of components and dimensions in space and time) as the basic principle for sustainability, integrating economic, social, and ecological development. At the same time, the RSU admitted that the relationship between economy and ecology is inherently conflictive. As a result, this requires a consensus on values, including minimizing, but tolerating, (unavoidable) "evil" and restrictions on current consumption.

Forging conflicting interests into a coherent agenda based on maximum consensus is the primary goal of democratic policy. Conflicts may arise between economy and ecology, between short-term and

long-term goals, and between local and global goals. These conflicts also can occur concurrently (connectivity, retinity).

Specific policy recommendations and choices of specific policy measures are not the intent of this paper. Rather, its purpose is to present the basic values for sustainability assessment and to describe the set of instruments available for sustainability assessment, as well as the limitations of their application. Thus, this paper is restricted to problem-framing; it does not address response-farming in detail (Dovers 1995).

Ecology and Economy

Because natural resources are scarce (Daly 1991), their monetary valuation has a significant, although limited, role in the policy domain. Monetary valuation, as a policy instrument, is necessary for resource allocation. In particular, it permits the internalization of externalities, and contributes to problem analysis from a cost-benefit point of view.

With regard to the value of nature, anthropocentric principles may lead to valuations different from those of biocentric holism. For example, this may occur when a decision must be made about the interests of the inhabitants of a mountain reserve versus the protection of the gorillas living there. Careful analysis of such a situation, however, frequently reveals that respecting the rights of the human inhabitants also serves the creatures to be protected (cf. Hampicke 1994; Steger 1995).

Three ethical principles for decisionmaking are proposed from an economic point of view: efficiency, sufficiency, and consistency. Efficiency is the most conventional policy recommendation. For example, the CGIAR Task Force on Sustainable Agriculture recommended that research focus on productivity and efficiency (CGIAR 1995). Proponents of this policy presume that technical progress will ultimately lead to such an efficient use of natural resources that depletion and pollution rates will no longer exceed the supply and absorption capacity of the environment. This is closely related to

"soft" sustainability, presuming the substitutability of natural resources by human intervention. On the contrary, sufficiency recognizes the scarcity of unsubstitutable resources ("hard" sustainability). Sufficiency is the most controversial policy principle because it is associated with uncomfortable restrictions on wealth. The RSU (1994) demands working toward moral consensus in society to achieve long-term goals that are to be valued more than short-term wealth, although a combination, with restrictions and prohibitive measures, may be unavoidable. Consistency, related to "sensible" sustainability, has been proposed as a "third way," as economic management consistent with natural cycles and phases (Huber 1995). It may, however, only pretend to harmonize economy and ecology. Ultimately, it consists of efficiency increase (though ideally cyclical) up to the limits of pool size and turnover rates when resource restrictions need to be faced (sufficiency).

Specific policy measures will have to consider all three principles, or strategies. Efficiency increase may be applied in cases of low resource use. When applicable, consistency is a desirable strategy, but it is limited to cyclical processes. Ultimately, there is no alternative for restricting resource use (i.e., sufficiency) of limited, unsubstitutable natural resources. System theory for sustainability assessment can help identify resource pools, turnover rates, component interactions, externalities, and such. It also can shed light on the complexity and connectivity of policy problems, and on their hierarchical order. It cannot, however, replace political value judgment on the limits of use and policy strategies to achieve the envisaged goals.

Short-Term and Long-Term Goals

Translating the principle of intergenerational equity into policy requires one to define the priorities between short-term and long-term goals, to choose realistic time horizons, and to deal with uncertainty.

Since the introduction of *Nachhaltigkeit* into forestry 200 years ago, the principle of present renunciation of wood harvest for the sake of securing future timber production has been claimed. This principle was forced on traditional smallholders by aristocratic landowners,

depriving them of their source of fuel wood and fodder (Trossbach 1996). This situation is similar to that in which some contemporary policymakers of industrial countries demand restrictions on rain forest use by traditional forest dwellers.

With regard to the temporal dimension of problem-framing, Dovers (1995) distinguished among the timing of possible impacts, their longevity, their reversibility, and their mensurability (i.e., their degree of risk, uncertainty, and ignorance). In modern society policy decisions generally have short time horizons (e.g., they are guided by interests in winning elections); that is, on the order of years rather than decades. Yet the impact of such decisions may have much longer effects; for example, in the case of nuclear energy (Meadows *et al.* 1992). Today's decisions frequently do not take into account the well-known potential for future damage. In the case of long-term environmental damage, Ludwig and others (1993) pointed out that "scientific certainty and consensus in itself would not prevent overexploitation and destruction of resources. Many practices continue even in cases where there is abundant scientific evidence that they are ultimately destructive." Using irrigation in the Euphrates and Tigris valley as an example, they concluded that "3000 years of experience and a good scientific understanding of the phenomena, their causes, and the appropriate prophylactic measures are not sufficient to prevent the misuse and consequent destruction of resources."

If policy decisions even in the case of well-known risk are not guided by advocating the interests of future generations, then it is even more difficult to institute preventive policies in cases of unknown risk, uncertainty, and ignorance of long-term impacts. For such cases the "precautionary principle" has been proposed. Dovers (1995) discussed the value of this principle for sustainability policies. He showed that—as has been shown for the ecosystem health approach—the precautionary principle is as normative and value-laden as sustainability itself, and is composed of loose, qualitative descriptors that are difficult to operationalize. Proposed techniques for operationalization of the precautionary principle are limited in scale and scope (e.g., quantitative or ecolog-

ical risk assessment, minimax criteria, safe minimum standards, or environmental bonds [Dovers 1995]). Dovers concluded that the precautionary principle is "primarily a moral or political notion that may be informed (or misinformed!) by science." The decision on when to apply the principle and the definition of threshold values may be supported by science, but it is still a normative, political decision based on informed judgment (cf. Wissel 1995).

Local and Global Goals

The majority of policy papers on global issues have their roots in industrial countries, while local strategies are proposed for agroecosystem management mainly in developing countries. This reflects the dominating interests and concerns: large-scale resource exploitation and pollution by industry versus agricultural production.

Large-scale policy proposals tend to be closer to static, top-down (neoclassical) concepts, proposing target values for resource use, whereby target definition is strongly normative. For example, Levi (1995) recommended that estimates of future nuclear energy use be based on the assumption that the population of developing countries will reach a target energy consumption of one-quarter that of the current consumption levels by inhabitants of industrial nations. The opposite policy approach involves local agendas with broad participation based on process-oriented, discursive development (Redclift 1994; Busch-Lüty 1995).

Bottom-up policy strategies derived from local agendas have their limitations in a system hierarchy: many problems cannot be solved by simple aggregation of subsystems or components. Sustainability assessment based on system theory can be used to define hierarchy structures, connectivity, and system boundaries.

There are few examples of spatial cross-boundary impacts of sustainability policies. Reardon and Vosti (1992) and Reardon and others (1995) analyzed such policy impacts on household-level decisionmaking in developing countries, as well as the interrelationship between environmental protection and poverty alleviation

(Reardon and Vosti 1995). Other cross-boundary examples include policies for interest compensation between local and large-scale goals, which may be developed by participatory optimization processes (Müller 1993).

Although human society and social policymaking are distinct from ecosystem behavior, one can conclude—in analogy to biodiversity—that the advantage of a dynamic and participatory local agenda is that it maintains or increases cultural diversity and complexity, thus enhancing system stability and resilience (cf. Norgaard 1989).

Conclusions

Ludwig and others (1993), after their analysis of sustainability as a guiding policy principle for deep sea fisheries as a large-scale system with a long time horizon, came, in brief, to the following conclusions: "Include human motivation, shortsightedness and greed; act before scientific consensus is achieved; rely on scientists to recognize problems but not to remedy them; distrust claims of sustainability; confront uncertainty; consider a variety of possible strategies; favor actions that are robust to uncertainties; hedge; favor actions that are informative; probe and experiment; monitor results; update assessments and modify policy accordingly; and favor actions that are reversible."

Their conclusions are confirmed by the analyses and results of this article, which follow:

Be honest. Since its inception 200 years ago the concept of sustainability has been applied in the spirit of industrial development. It remains in this tradition in that its main focus is the repair of actual and potential environmental damages caused by industrial development (internalization of externalities). Sustainability principles can be upheld only as long as they do not interfere with dominating economic interests. Frequently, arguments for environmental protection in the interest of future generations deny contemporaries in developing countries the same rights to resource consumption as the citizens of industrial nations.

Be modest. Scientific analysis of sustainability is a necessary and useful tool for problem-framing, but its role in the derivation of specific policies is limited. Dovers (1995), following Funtowitz and Ravetz (1991), recognized a gradient for response-framing from a micro to a macro level. Micro-problems can be solved by "applied science," meso-issues by "professional consultancy," while macro-issues require "post-normal science," reflecting a gradient of increasing system uncertainty and increasing decision stakes. Consequently, with increasing scale, policy decisions become more and more normative and value-guided, possibly even irrational, rather than based on "hard" scientific facts.

Sustainability assessment of intergenerational equity should be claimed with care. The majority of methodologies for sustainability assessment of agroecosystems do not go beyond a couple of decades; thus, they do not cover intergenerational time spans (Figure 3). Similarly, the time horizon of policy measures (not impacts) has the same order of magnitude. If longer time spans are considered, the scientific operationalization of intergenerational equity is increasingly insecure. From the perspective of system theory it implies long-term *ex ante* analysis of (eco)system development, which lacks experimental foundation.

The claim of intergenerational equity is an ethically well-supported moral appeal, but it is not a commonly agreed upon policy principle (cf. Agarwal 1993). It lacks sufficient ground for consensus in society because future generations cannot retaliate against present injustice. History has proven that greed and self-interest generally gain priority over unselfish, altruistic care for a yet nonexistent population.

Be clear. Sustainability assessment demands clarity in two aspects. First, it requires the explicit definition of objectives, time scales, and spatial dimensions when applying scientific operationalization. Only then can methodologies be adjusted to the objective and scope of the problem. Second, the underlying norms and values need to be clarified. Figure 2 shows that sustainable development is a strongly normative concept that requires scientific operationalization. For scientific sustainability assessment, the holistic ecosystem health

concept appears to be an appealing approach. Similarly, the precautionary principle appears to be an adequate policy for sustainable development. However, it has been shown that both concepts are as normative and vague as sustainability itself. Thus, scientific operationalization of sustainability on the basis of either of these concepts needs special care. If applied with the necessary caution and with an awareness of their limitations, however, these normative concepts do have some justification. First, they are useful tools for defining targets and visions for policy development. (It should be made explicit, however, which normative decisions and values have entered into the definition of the target system.) Second, they show that reductionist science—even most complex modeling and system theory—cannot capture the full range of aspects associated with sustainability, just as a description of symptoms does not fully define health.

Be cautious. Despite its operational limitations, the normative aspect of sustainability and its scientific assessment is a powerful and necessary concept for decisionmaking at all levels, from farming to international policy. Noting the potential long-term impacts of environmental management practices is certainly a prerequisite for decisionmaking, as long as alternative strategies exist and are viable. In view of potential damage to the environment, the precautionary principle is a useful tool despite its conceptual weakness. If possible, the reversibility of impacts should be aimed at. Careful scientific problem analysis using the presented methodologies for sustainability assessment can contribute to understanding often complex situations.

Frequently, policy decisions are required without a full understanding of the problem and its impact or without agreement on its scope. To limit the impact of policy decisions based on insufficient knowledge, discursive and iterative approaches are preferable. Participatory processes for seeking consensus on norms and strategies help define acceptable policy targets, although they do not guarantee that the diverse positions and interests of otherwise neglected groups will be considered.

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