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# New advances in economic modelling and evaluation of environmental issues

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**Abstract**

This paper gives an overview of recent progress made in modelling economic environmental systems and in environmental policy analysis. In the modelling part attention will be given to new integrating frameworks offered inter alia by materials balance approaches, especially in the context of linkages between physical environmental phenomena and economic production and valuation. These can be relevant for studying materials–product chains, multisectoral materials flows, or even multiple use of complex ecosystems. Modern approaches will be dealt with, such as analysis for sustainable development, and ways of incorporating scenario experiments in environmental modelling approaches. In the context of sustainable development, modelling of multiple use of ecosystems and of spatial dimensions is also discussed. In the last part of the paper new advances in the area of environmental policy analysis will be dealt with. The main focus will be on methods for addressing uncertainty in evaluating environmental policy strategies, in particular fuzzy information and the use of meta-analysis. © 1997 Elsevier Science B.V.

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## 1. Analysis of environmental–economic issues and conflict

Environmental economic analysis and conflict management is nowadays a major challenge to policy analysts. In the past decades, the threatened state of the natural environment has become a key issue in policy evaluation because of the great many externalities involved. It is also increasingly recognized that environmental and resource conflicts will generally have far-reaching economic and ecological impacts which cannot always be encapsulated by a prevailing market system. The limits inherent in conventional economic evaluation methodologies and the necessity of analyzing unresolvable conflicts between diverse policy objectives have led to a need for more appropriate and fine-tuned analytical tools for strategic evaluation of environmental policies or plans (Van Pelt, 1993). *Environmental management* is essentially based on conflict analysis characterized by technical, socio-economic, environmental and political value judgements, since in an environmental planning process straightforward and unambiguous solutions are hard to be attained. Such a multi-faceted planning process will always give rise to a search for

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acceptable compromise solutions, an activity that requires a proper modelling framework and an adequate evaluation methodology with a strong applied orientation.

This also explains the popularity of multicriteria evaluation for project and policy evaluation in the context of economic analysis of environmental issues. A proper use of multicriteria analysis presupposes inter alia the existence of an adequate quantitative environmental–economic model. Models aiming at depicting, predicting or analyzing problems of an integrated economic–environmental nature are commonly referred to as *economic–environmental* and *economic–ecological* models (see Hafkamp, 1984; James, 1985; Braat and Van Lierop, 1987; Van den Bergh, 1996). They are characterized by a variety in technical structure (nonlinear/linear, static/dynamic, descriptive/forecasting/optimizing), and may be considered as the result of a trade-off between generality, precision and realism (see Costanza et al., 1993). New developments in this type of modelling focus on extending existing monodisciplinary models to incorporate processes usually studied by other disciplines. Some interesting examples are entropy from physics to economics and ecology (see Ruth, 1993), evolution from biology to economics (Hinterberger, 1994) and resource management (Munro, 1994), and decentralized decision making and externalities from economics to the study of economy–ecosystem interactions (Crocker and Tschirhart, 1992). Technically, translation of models from one discipline to others has recently focused much on game-theoretic and chaotic nonlinear models.

In view of the complexity of the interdisciplinary problem, involving description, estimation, analysis and evaluation, there is a need for an appropriate analytical framework allowing for a comprehensible and operational representation of a real-world environmental economic system. The strong quantitative tradition in economics has in the recent past enabled researchers to include environmental elements fairly easily in conventional models. Nevertheless, in integrating economic and environmental or ecological models various difficult methodological problems have to be faced, such as: *differences in time scales* (general economics focuses on short to medium term effects, whereas most of ecology is based on medium to long turn processes); *differences in aggregation levels* (in economic models high relative to most ecological models), *differences in spatial scales* (the spatial scale of many ecological variables is sometimes very low, whereas that of many economic variables is rather high) and *differences in measurement levels of the variables* (the level of precision may vary, and thus there is a clear need for methods taking also into account information of a 'mixed' type).

In the areas of environmental and resource management and policy planning for sustainable development, many conflicting issues and interests emerge. In real-world situations of public decision analysis two main cases can be distinguished (Stewart, 1991):

(1) *Broad commonality of goals*, i.e. differences among parties are revealed through various trade-offs which they perceive to be most in their interest.

(2) *Direct conflict of goals*, e.g, a case where public policy involves an explicit division of resources among different sectors of the society or where attitudes are leading to strong unreconcilable differences (e.g. environmentalists versus industrialists).

It is thus clear that environmental policy analysis has to be positioned in a complex force field of multiple actors and objectives. From an operational point of view, the major strength of multicriteria methods is their ability to address problems marked by various conflicting interests. Multicriteria evaluating techniques cannot solve all these conflicts, but they can help to provide more insight into the nature of these conflicts by providing systematic information into ways of arriving at political compromises in case of divergent preferences in a multi-group or committee system by making the trade-offs in a complex situation more transparent to decision-makers. A necessary condition for a systematic evaluation is, however, the availability of systematic environmental information, preferably based on a structural systemic impact model.

In models for environmental and resource policy-making, the following three main types of policy objectives may be distinguished (Braat and Van Lierop, 1987):

1. nature conservation objectives, e.g., 'minimum exploitation of goods and services at minimum (private and social) cost';

2. socio-economic objectives, e.g., 'production of goods and services at minimum (private and social) cost';
3. mixed objectives, e.g., 'maximum sustainable use of resources and environmental services'.

It is widely recognized that in policy-relevant economic–environmental evaluation models, socio-economic and nature conservation objectives are to be considered simultaneously. Consequently, multi-criteria methods are, in principle, an appropriate modelling tool for combined economic–environmental evaluation issues. Such methods seem to be an adequate response to policy choice situation characterized by a high degree of market failure.

In the context of conflicting interests, it is noteworthy that in environmental management there is often an interference from local, regional or national government agencies, while there is at the same time a high degree of diverging public interests and conflicts among groups in society. At an *intra-regional level* many conflicting objectives may exist between different actors (consumers, firms, institutions, etc.) which can formally be represented as multiple objective problems and which have a clear impact on the emerging spatial configuration of a certain policy field (e.g., industrialization, housing construction, road infrastructure construction). At a *multi-regional level* various spatial linkages exist that may affect a spatial system through spatial interaction and spillovers (e.g., diffusion of environmental pollution, spatial price discrimination). In a formal sense these can be described by means of a multiple objective programming framework. At a *supra-regional level* various hierarchical conflicts may emerge between regional government institutions and the central government or between regional branches and the central office of a firm, which implies again a multiple objective decision situation. Thus there are many cases of conflicting policy regimes.

According to Daly (1989), three main conflicting goals in economics may be identified: *optimal allocation* (efficiency), *acceptable distribution* (equity) and *optimal scale* (sustainability). While an optimal allocation may result from the operation of markets, the attainment of both distributional issues and optimal scale (or at least any scale that is not above the maximum carrying capacity) requires collective action by the community on the relevant regional, national or international-level according to the nature of the problems at hand.

In this respect, one may refer to Tinbergen (1956), who made a useful distinction between the analytical aspect and the political aspect of public decision-making. The *analytical aspect* is concerned with links between all variables relevant in the decision-making process as well as with all side-conditions imposed by the economic, social and technological structure of society. This analytical aspect of a decision problem can in theory be represented by a set of formal statements or an impact model (or structural model). The *political aspect* concentrates on the choice of instruments and there being manipulated to realize the policy objectives. These policy objectives can be operationalized as a set of fixed targets to be strived for or as arguments of a community welfare function to be optimized. In particular the latter approach has received much attention in the literature about policy-making and in welfare economics, and has become an important element of the economic theory of environmental policy (see Baumol and Oates, 1988).

The use of conventional optimization models has been criticized from many sides. The optimizing approach is based on the assumption that different objectives can be expressed in a common denominator by means of trade-offs, so that the loss in one objective can be evaluated against the gain in another. This idea of *compensatory changes* underlies both the classical economic utility theory and the traditional cost–benefit analysis and has been applied as 'externality theory' to specific environmental economic issues. The determination of a common denominator is, however, fraught with many difficulties. From a theoretical point of view, the optimizing principle is very elegant, since it provides an unambiguous tool to evaluate alternative strategies on the basis of their contribution to social welfare. From an operational point of view, the value of the traditional optimizing approach is however, rather limited, because the specification of a social welfare function requires complete information about all possible combinations of actions, about the relative trade-offs between all actions and about all constraints prevailing in the decision making process. This certainly applies to environmental policy which exhibits glaring examples of conflict analysis.

## 2. Environmentally sustainable development

Environmentally sustainable development has in the past decade become an important policy objective. The concept of sustainability has however, already a long history. The most well known definition of sustainable development is probably the one given by the World Commission on Environment and Development (WCED, 1987): "... paths of human progress which meet the needs and aspirations of the present generation without compromising the ability of future generations to meet their needs". Goodland and Ledec (1987) define sustainable development as: "... a pattern of social and structural economic transformations which optimizes the benefits available in the present without jeopardizing the likely potential for similar benefits in the future". This definition implicitly assumes a need to maintain yields from renewable natural systems over long periods of time. Other approaches to the concept of sustainable development focus on the physical or natural resource base of any economy. Pearce and Turner (1990) claim that sustainable development implies maintenance over time of aggregate resource stocks, such that the potential to generate welfare is not allowed to fall below the current level. Clearly, this viewpoint raises also important questions concerning the measurability of environmental quality and environmental capital (see, e.g., Pezzey, 1993; Van den Bergh and Van der Straaten, 1994; Jansson et al., 1994).

According to Costanza (1987), "... sustainability does not necessarily mean a stagnant economy, but we must be careful to distinguish between growth and development". Economic growth which is an increase in quantity cannot be sustained indefinitely on a finite planet. Economic development which is an improvement in the quality of life without necessarily causing an increase in quantity of resources consumed, may be sustainable. Sustainable growth is in the long term essentially an impossibility. Sustainable development should therefore become our primary long-term policy goal. Hence the sustainability of natural resources and the environment are to direct explicitly economic development. Motivated by Costanza et al. (1991, p. 8), a practical definition of global sustainability is the following: Sustainability characterizes a relationship between quickly-changing human economic systems and larger dynamic, but normally slower-changing ecological systems, in which: (i) human life can continue for a very long period of time (say more than 1000 years), (ii) human individuals can flourish (are free and happy); and (iii) human cultures can develop; and (iv) effects of human activities remain within bounds, so as not to destroy the diversity, resilience, and functioning of ecological systems. Although it is not difficult to be critical on any definition of sustainable development, this one, while leaving sufficient freedom, to fill in the details at the same time enlightens the four major components in the debate as well as the modelling of the concept. In practice, there is a multiplicity of complementary and sometimes alternative definitions.

The foregoing discussion also leads us to the ecology–economy perspective (Costanza, 1991). The economic system is an open dynamic system of the overall finite global ecosystem. Similar in many ways to ecological subsystems. The two systems are physically connected by the throughput of energy and matter from ecosystem sources and by other environmental goods and services sustaining economic activity. This means that economic production of any commodity needs natural resources, and gives rise to the transformation of natural resources — from discovery, extraction, refinement and so on — into useful raw materials and eventually into humanely produced goods and services; it also requires the use of industrial energy as well as the support by ecosystems that are being driven by solar energy input. The economic subsystem rests on these bio-physical foundations, which may be formalized through ecosystems' theory and the laws of thermodynamics. One important implication is that the economy must behave, to be sustained, in a way that is consistent with these bio-physical laws.

Since the market prices do not reflect exactly the relative scarcity of environmental resources, it is necessary from a political economic point of view, in order to avoid an overexploitation of these resources, to impose appropriate *regulatory measures* by public authorities. In fact, since the rational decisions of individual agents lead necessarily to an outcome that is inconsistent with the best interests of society, a '*social trap*' (Costanza, 1987) exists. The exploiters of a common resource stock may have little incentive for the conservation of that resource; Hardin (1968) has called this the "tragedy of the commons". The situation may even be worse in the case of open access resources. For instance, as long as fish in the sea can be caught profitably, fishermen will wish to do so, and this may lead to severe over-fishing not taking into account long-term effects (Clark, 1990). In order to cope with such externalities, policy measures may have to be introduced. As pointed out by Baumol and Oates (1988), these measures can take the form of *direct regulations* (e.g. maximum pollution emissions) or the form of *economic disincentives* (pricing systems based on social costs in the form of taxes and subsidies).

A different economic perspective leads to a property rights approach. The 'Coase theorem' (Coase, 1960) provides the theoretical basis for a non-interventionist pollution control policy. Environmental pollution is a form of *market failure* because of the overexploitation of resources held as common property or not owned at all; therefore, the market fails when property rights are inadequately specified. According to the property rights approach, increased government intervention should be resisted, because public ownership of many natural resources is the real root of resource control conflicts; then there is a *policy failure*. According to Coase, given certain assumptions, the most efficient solution to pollution damage is a bargaining process based on tradeable rights: if the polluter has the right, the sufferer can compensate him in order not to pollute; if the sufferer has the right, the polluter can compensate him to tolerate damage.

From an ecological–economic perspective, the expansion of the economic subsystem is limited by the size of the overall finite global ecosystem and by its dependence on the life support sustained by intricate ecological connections which are more easily disrupted as the scale of the economic subsystem grows relative to the overall system. Since the human expansion, with the associated exploitation and disposal of waste and pollutants, not only affects the natural environment as such, but also the level and composition of environmentally produced goods and services required to sustain society, the economic subsystem will be limited by the impacts of its own actions on the environment (Folke and Käberger, 1991).

Generally, ecosystems are used in several ways at the same time by a number of different users, referred to as multiple use. Such situations lead almost always to conflicts of interest and damage to the environment. The consequences range from suboptimal use due to unregulated access, to degradation of resource systems due to limited knowledge of the ecological processes involved. It goes without saying that mapping out such consequences is a major task of environmental impact assessment. Dynamic simulation analysis has been applied to deal with this issue, where multi-criteria evaluation can be used to deal with a variety of physical, ecological and economic indicators (see for a nice example Braat, 1992). Dynamic models in resource economics addressing multiple use are rare (see Bowes and Krutilla, 1985), and have focused on welfare optimization (single criterion) and the optimal temporal trade-off between conflicting activities (see, e.g., Bishop and Samples, 1980). In a static context the multiple use of ecosystems has been placed in the context of equilibrium analysis, and it has been shown that it can be regarded as an application or — in some directions — generalization of externality theory (see Crocker and Tschirhart, 1992). More work, both theoretical and applied, is required to decide about policy and management of such pressing multiple use problems as, for instance, wetland ecosystem degradation (see, e.g., Costanza et al., 1989, Gren et al., 1994).

In the context of sustainable development especially the spatial dimension has received little attention. The importance of the spatial element arises from a reciprocal relationship: (1) local processes have global impacts; and (2) global trends give rise to local effects. For example, the loss of ecosystems in some regions may have a large impact on global climatological conditions and geochemical cycles. Over-grazing and deforestation may lead to large-scale soil erosion, downstream sedimentation, flooding and salinisation (see, e.g., Clark and Munn, 1986). Furthermore, environmental processes do not uniformly and smoothly impact all regions, but may have important different consequences at a regional scale (see, e.g., Alcamo et al., 1990). The specific regional environmental and economic structure determines the sensitivity of a region to external environmental and economic forces (Siebert, 1985, 1987).

Regional sustainable development is a concept which has received only little attention, which is somewhat strange in view of the large literature that has evolved on (general or global) sustainable development. Two characteristics of regions are responsible for the difference between sustainable development and regional sustainable development (RSD); they are: (i) cross-boundary flows of environmental and economic goods and services; and (ii) external determinants of regional development. A realization of regional sustainable development can therefore be regarded as based on the sustainable provision of natural resources in the region and the sustainable import and export — from the viewpoint of regional sustainable development in other regions — of resources, goods, services and waste. The problem of unsustainable development of a region is linked to the fact that the size of a regional population and economy are not checked sufficiently by the region's carrying capacity, and therefore overshooting may occur. In many cases this may be acceptable if at a higher level of spatial aggregation overall sustainability was ensured. A second reason for unsustainable development of a region may be the existence of the negative external impact of regional development, cross-boundary pollution and global phenomena (e.g., climate change) from which regional control is separated. Both the regional cross-boundary flows supporting the economy, and the cross-boundary pollution cause the regional carrying capacity to be exceeded for a while, from which the environment may be harmed permanently. This has negative consequences for the carrying capacity itself and thus for the long-run performance of the regional economy (see Van den Bergh and Nijkamp, 1994b,c,d). Consequently regional sustainable development has to fulfil two goals: (1) it should ensure an acceptable level of welfare for the regional population, which can be sustained in the future; and (2) it should not be in conflict with sustainable development at a supra-regional level (see Van den Bergh, 1991).

Studying sustainability in a multi-regional system may also be useful to deal with the spatial implications of global sustainability, in terms of regional activities, and inter-regional trade flows. Verhoef and Van den Bergh (1995a,Verhoef and Van den Bergh, 1995b present analytical and numerical results of such a type of investigation, in the context of sustainable transport. Based on an extended spatial price equilibrium model, an optimal trade-off can be made between mobile and immobile sources of pollution, between regional production (with autarky as an extreme case) and trade dependence, and between volume reductions and technological solutions. The model also allows to consider to what extent partial — such as isolated, single sector — policies can lead to sustainability goals. Although transport is pre-eminently linked to issues of spatial sustainability, one can also translate the results to other types of open systems, such as countries, sectors and ecosystems. A similar issue is studied in Van den Bergh and Nijkamp (1995), now in an explicit dynamic simulation modelling context where economic and environmental processes of two regions, and their trade and environmental interactions, are dynamically specified. The resulting model is used to trace, among others, sustainable growth in an open economy, the effect of dissimilarity between regional environmental processes, and the role of technological progress and diffusion. Essential for the outcomes is the endogenous pattern of interregional trade in the model.

Especially the trade-off between efficiency and sustainability in a multi-sector production–consumption system is interesting in the above examples, since it can be analytically linked to a trade-off between the absolute volume or size of each sector, its relative size in the economic structure, and the level of 'environmental technology' adopted in each sector. In an operational sense this may be done by using indicators for efficiency and sustainability. Both on the theoretical and operational level many opportunities exists for further model-based analysis.

## 3. The material balance approach in environmental economic modelling

Environmental policy analysis requires the use of a structured impact system. In the past decades a wide variety of economic–environmental models have been developed with more or less success (see for overviews inter alia Hafkamp, 1984; Van den Bergh, 1996). The present section is based on the viewpoint that a long run economic–environmental analysis, which is needed for issues of sustainable development, should be based on two main elements: (i) the long run relationship between the economy and the natural environment is characterized by two-way interactions between on the one hand population growth, investment, technology and productivity, and on the other hand declining environmental quality and resource exhaustion; and (ii) a more realistic representation of the interdependence between various environmental effects, from extraction to emission, can be realized by adopting a materials balance perspective on economic processes. This means that *direct* mutual impacts between the economy and the environment, as well as *indirect* economic–environmental influences are taken into account, by separating between individual effects from and on production, consumption and welfare. Wilkinson (1973) already introduced the idea of ecological disequilibrium to link economic change to environment–economy relationships in the long run. Inclusion of such environmentally influenced economic change in impact models has been undertaken inter alia by Faber and Proops (1990) and Van den Bergh (1993). Complementary and related models stem from applied systems theory, notably in the fields of biophysical "macroscopic mini-models" derived from energy language diagrams (Odum, 1987) and of global modelling (Meadows et al., 1982). The main shortcomings of these models in comparison with a materials balance approach are the lack of consistent description of the relation between substitution of productive inputs (or components of welfare), virgin resource extraction and waste generation and residuals emission.

For an investigation of economic–ecological integration at both a theoretical and operational level of modelling one may — as indicated above — include materials balance conditions to account in a consistent way for material flows that lead to various interlinked effects. Although the use of material balance models was already propagated more than two decades ago (see Ayres and Kneese, 1969 and Kneese et al., 1970), it has unfortunately seen little application. Furthermore, the combination of non-linear models and materials balance conditions is rare, in both theory and applications. The main reason is that materials balance analysis or materials accounting can be done much more easily with linear production functions. Exceptions are Faber et al. (1987), Gross and Veendorp (1990), Van den Bergh (1991), Ruth (1993), Kandelaars and Van den Bergh (1996).

The concept of materials balance applies to all natural and economic processes. It means that materials in a physical system are not lost, and that material inputs in processes end up in either stock accumulation or material output flows. It should be mentioned here, that the material input is larger than the useful goods output, especially in view of spillage and auxiliary materials (like water and fertilizer in agriculture) (see Ayres and Kneese, 1969).

In formalizing the materials balance principle in environmental economic models, the following steps are required: (i) relevant variables should be in material units; (ii) where necessary, transformations must be modelled between (variables in) material units and other units; and (iii) materials balance conditions should be specified for economic variables in the economic system, for ecological/physical variables in the environmental system, for economic–environmental interactions (which include both economic and environmental variables).

Production functions can be formulated in various ways to satisfy the materials balance principle. Application of the materials balance principle to the production process expresses that all material input must end up somewhere: in final or capital goods or in waste. The link between production theory and materials balance is rarely touched upon in the literature. Some theoretical and conceptual steps taken in this direction were set by Anderson (1987) and Smith and Weber (1989). Some properties for a Cobb-Douglas production function that satisfy the materials balance condition are derived by Gross and Veendorp (1990). They show with a standard economic growth analysis that such a function sets a limit to growth for the case of an economy that obtains its material inputs from a non-renewable resource.

We will now devote more attention to characteristics of non-linear production functions that are consistent with the law of materials balance. This will lead to several possible formulations some of which are probably more useful than others, dependent on the context. The production process may be envisioned as a transformation of resource inputs into goods and waste outputs by actors (funds/agents; see Georgescu-Roegen, 1971). One may expect considerable potential for substitution between sub-categories of actors (labor and capital), since they play a similar role in the production process; therefore, they are aggregated into the variable 'actors' ($A$). An increase in the use of agents may reduce the amount of waste output. Substitution is possible within the category of resource inputs to production ($R$). Substitution between the categories of actors and resources is limited. This is not assumed a priori; it follows from the application of the materials balance condition to the production function.

A materials balance can be expressed as follows: (i) the inequality $R > Q$ ($R$ and $Q$ denote the levels of material input and goods output from production, respectively) as a minimal consistency condition; or a more strict condition, such as $R > Q + x$ ($x$ is a lower bound for waste residuals from production), based on knowledge of technical and physical constraints; and (ii) the equality $R = Q + W$ ($W$ is waste residuals from production), if material accounting is strived for (and possible); clearly, (ii) encompasses (i), since $W$ is always positive.

The equations in Eq. (1) show a general relationship between the output of goods $Q$ on the one hand, and all factors involved in its production on the other hand. These factors include $A$, $R$ and $W$. The materials balance principle is explicitly stated in terms of an equality condition that relates the total resource input to the output of produced goods and waste. The parameter $t$ in this (and the subsequent) production functions denotes the change resulting from technical progress. In the formal representation of Eq. (1), $A$, $R$ and $W$ are treated identically, i.e. their conceptual difference is not made explicit in $F(\cdot)$, but becomes clear only after the material constraint is added.

$$Q = F(A, R, W, t), \qquad R = Q + W \tag{1}$$

It should be noted that all partial derivatives of $F(\cdot)$ are positive. Of course, we have here an aggregate description of the production process, namely only in material terms. Specific characteristics of the final product are not considered, so that a simple waste production function can be derived, namely as $W = R - F(A, R, W, t)$. Therefore, an 'ex post' relationship can be established between the production functions for useful output and waste, i.e. after the application of the materials balance condition.

The second type of formulation of production subject to a materials balance shown in Eq. (2) starts with two separate, (ex ante) *independent* production functions for goods and for waste, viz:

$$Q = F_1(A, t) \text{ and } W = F_2(A, t),$$
$$R = Q + W \text{ or } Q + W \leq R_c. \tag{2}$$

The formulation in Eq. (2) with the equality constraint can be interpreted as follows: a given actor (and activity) level $A$ determines the levels of useful and waste outputs; the sum of these gives the resource requirement. The interpretation based on the inequality constraint is that for a given amount of resources $R_c$, the inequality constraint should be satisfied, namely by choosing an actor level $A$ such that the sum of the resulting values of $Q$ and $W$ is feasible.

As a special case of such a constrained process we may distinguish between actors allocated to production and to an activity (denoted by the function $F_3(\cdot)$) which diminishes the amount of waste resulting from the (regular or main) production process (denoted by $F_1(\cdot)$ and $F_2(\cdot)$). It is assumed that the available levels of capital, labor and resources are given. This is formalized in Eq. (3) as follows:

$$Q = F_1(A_1), \qquad W = F_2(A_1) - F_3(A_2),$$
$$A_1 + A_2 \leq A_c, \qquad Q + W \leq R_c. \tag{3}$$

The conditions that apply to these functions are that all derivatives be positive, and that $F_2$ always has a higher value than $F_3$. In contrast to the formulation in Eq. (2), one may add the objective of maximizing $Q$ or minimizing $W$ to the formulation in Eq. (3).

A third type of formulation of materials balance production functions uses a production function $F$ that (automatically) satisfies the consistency restriction of a materials balance, i.e.: $F(A, R) \leq R$ for all values of $R > 0$. This can be accomplished in two ways. The first is shown in Eq. (4), and is characterized by taking the minimum of any general production function and some share of the resource input, viz:

$$F(A, R, t) = \text{MIN}\{F(A, R), a(t) \cdot R\}. \tag{4}$$

The parameter $a(\cdot)$ describes technological efficiency in resource use; it falls between zero and one, and has a positive derivative. It can be defined as follows:

$$a(t) = 1 - \frac{W}{R} = 1 - \frac{F_2(A, R, t)}{R} \tag{5}$$

where we regard waste in terms of a production process as formalized in Eq. (2). From Eq. (5) it is clear that $a(\cdot)$ is to be interpreted as the efficiency of resource use in production at time $t$ (i.e., with the technology available at $t$), which has a lower bound zero.

The second way, shown in Eq. (6), uses a function that is based on a resource efficiency coefficient $r(\cdot)$ which can be regarded as a variable coefficient that relates useful output to resource input. This coefficient is increasing in all its arguments:

$$Q = r(A, R, t) \cdot R, \quad 0 \leq r(\cdot) < 1. \tag{6}$$

Resource efficiency in production is thus assumed to be improved either by increasing the intensity of the production activity factors relative to the resource input (indicated by an activity–resource ratio $A/R$), or by technological progress (indicated by $t$).

Since the materials balance condition implies a linear (in)equality, it complies easily with linear types of models, such as fixed proportions and linear production functions (see Van den Bergh, 1991). From the above distinction between the three types of representations of production processes satisfying materials balance, it is clear that one may choose between various specifications of materials balance production functions. For further details and applications we refer to Van den Bergh and Nijkamp (1994a) and Kandelaars and Van den Bergh (1996).

The use of materials balance conditions can provide insight in many areas of environmental and environmental–economic research. Examples are: studies on the physical and ecological limits to economic growth, based on the notion that resource and assimilative capacities may restrain materials flows entering and leaving economic systems; resource scarcity over longer periods of time given various production and consumption scenarios; integrated materials–product chain policies such as materials and product recycling, e.g. via deposit–refund systems, waste taxation or subsidies on technology; materials flows analysis between economic and environmental systems, such as nutrient flows in wetland areas, on the boundary of hydrological, ecological and agricultural production processes.

It is clear that a materials balance representation may also be helpful in depicting the spatial aspects of a complex economic–environmental system, including the distribution of pollution. The physical dimensions incorporated in a materials balance model allow for a proper and consistent mathematical representation of both physical and economic linkages, including their geographical distributions. It is clear that — despite the progress made in environmental modelling — uncertainty is still a dominant feature. This will be discussed in the next section.

## 4. Uncertainty and scenarios in environmental modelling

Apart from conflicting objectives and complex interactions environmental conflict and policy analysis is also characterized by uncertainty. Furthermore, it has been argued that the presence of qualitative information in evaluation problems concerning socio-economic environmental and physical planning is a rule rather than an exception (Nijkamp et al., 1990). Thus there is a clear need for methods taking into account qualitative and imprecise information. In multicriteria evaluation theory, a clear distinction is made between quantitative and qualitative methods. The strong quantitative tradition in economics has enabled researchers to include environmental elements — measured in a cardinal metric — fairly easily in conventional models focusing on the interface of economics and the environment. However, qualitative aspects are harder to deal with in traditional models and, therefore, there is a clear need for methods that are able to take into account information of a 'mixed' type (both qualitative and quantitative measurements). Another problem related to the available information concerns the uncertainty contained in this information. Ideally, the information should be precise, certain, exhaustive and unequivocal. But in reality, it is often necessary to resort to information that does not have those characteristics so that one has to face uncertainty of a stochastic and/or fuzzy nature present in the data (Munda, 1993). If it is impossible to exactly identify or establish the future state of the problem faced, a *stochastic uncertainty* is created. This type of uncertainty is well known; it has been thoroughly studied in probability theory and statistics. Another type of uncertainty derives from the ambiguity of this information, since in the majority of the particularly complex problems addressing the interface of environment and men, much of the information is expressed in linguistic terms so that it is essential to come to grips with the fuzziness that is either intrinsic or informational typical of all natural languages. Therefore, a combination of the different levels of measurement with the different types of uncertainty has to be taken into consideration. The following taxonomy can then offer a useful framework for typifying empirical studies (see Table 1).

*Fuzzy uncertainty* does not concern the occurrence of an event, but the event itself in the sense that it cannot be described unambiguously. This situation is very common in human systems. Spatial–environmental systems in particular are complex systems characterized by subjectivity, incompleteness and imprecision (e.g., ecological processes are sometimes uncertain and little is known about their sensitivity to stress factors such as various types of pollution). Zadeh (1965) writes: "as the complexity of a system increases, our ability to make a precise and yet significant statement about its behavior diminishes until a threshold is reached beyond which precision and significance (or relevance) become almost mutually exclusive characteristics" (*incompatibility principle*). Therefore, in these situations statements such as "the quality of the environment is good", or "the unemployment rate is low" are quite common. Fuzzy set theory is a mathematical theory for modelling situations in which traditional

Table 1
Occurrence of combinations of information measurements levels and uncertainty

| Uncertainty level | Information measurement level | |
|---|---|---|
| | quantitative information | qualitative information |
| Certainty | very rare | rare |
| Uncertainty | common | very common |

modelling languages that are dichotomous in character and unambiguous in their description cannot be used. Human judgements, especially in linguistic form, appear to be plausible and natural representations of cognitive observations. We can explain this phenomenon by *cognitive distance*. A linguistic representation of an observation may require a less complicated transformation than a numerical representation, and therefore, less distortion may be introduced in the former than in the latter. In traditional mathematics variables are assumed to be precise, but when we are dealing with our daily language, imprecision usually prevails. Intrinsically, daily languages cannot be precisely characterized on either the syntactic or semantic level. Therefore, a word in our daily language can formally be regarded as a fuzzy set.

Fuzzy information can be represented in decision models in two different ways:
- by using linguistic variables;
- by using fuzzy numbers.

In a decision problem it is possible to distinguish two main elements, available information and manipulation rules for this information. A fuzzy decision model is essentially characterized by the presence of a set of membership functions. These membership functions can be defined on one or more of the other components of the model; therefore, the degree of fuzziness of the model may vary accordingly. Both continuous and discrete fuzzy multicriteria methods exist in the literature. Recently a new discrete multicriteria model whose impact (or evaluation) matrix may include either crisp, stochastic or fuzzy measurements of the performance of an alternative $a_n$ with respect to a criterion $g_m$ has been developed (see for details Munda et al., 1994). Applications can be found in forestry management, or landscape planning where linguistic information or value statements are preponderant. It may be concluded that fuzzy approaches are a crucial component of modern decision analysis. There are also other approaches to the treatment of uncertainty, and these will now be discussed.

Fuzziness and uncertainty are indigenous features of environmental management. The main aim of evaluation methods is to improve the quality of environmental policy by using the most appropriate tools, given the available data. Thus a kind of 'plausible reasoning' (see Polya, 1954) is in order here to derive justifiable inferences about 'states of the world'.

Evaluation methods and techniques can also be extended with complementary analytical tools, such as *scenario analysis*. Scenario analysis is one of the methods and techniques of prospective policy research that have become very popular since the late sixties. Especially in the case of unstructured decision problems with uncertain and fuzzy outcomes, scenario analysis may be an appropriate instrument. The main difference between scenario analysis and conventional methods of policy analysis is that scenarios do not only contain a description of one or more future situations, but also a description of a consistent series of events that may connect the present situation with the described future situation(s).

Scenarios can be identified by four characteristics (cf. Van Doorn and Van Vught, 1981):

- A scenario is either *descriptive* or *normative*. The prospective paths and pictures of a descriptive scenario are based on the know-how developed in the past and present. The question whether these paths and pictures are desirable or not, is not raised. The first scenarios designed by Kahn and Wiener (1967), are in agreement with this description. The construction of normative scenarios is based upon the ideas of the scenario-writers or scenario-users. The future paths and pictures are selected by these writers and users. The so-called Ozbekhan-scenarios (see Ozbekhan, 1969), as a response to Kahn and Wiener, may be regarded as member of this category (cf. Van Doorn and Van Vught, 1981).
- Another distinction that can be made is the difference in direction of the scenario analysis. If future pictures are based upon the present situation and future paths leading to it, then the scenario is said to be *projective*. On the other hand, if at first the future situations are determined and next the paths leading to this situation, then in fact these paths lead from the future backwards to the present. As they are composed afterwards, these scenarios belong to the class of *prospective* scenarios. Prospective scenarios are always normative, while projective scenarios are either descriptive or normative.
- A scenario can be characterized as a *trend* scenario or as an *extreme* (or contrast) scenario. Trend scenarios are in fact an extrapolation of the present situation. Extreme scenarios on the other hand, try to construct future paths and future situations that are considered to be in principle feasible, though very unlikely. They are both always projective scenarios.
- The last distinction to be made is whether a normative scenario is based upon the preferences of the *majority* of people, or whether it is based on the preferences of a *small minority*. The first group may be characterized as "common opinion" scenarios, and the second as "happy few" scenarios.

It is evident that the use of scenarios is of great

Table 2
Differences between forecasting and scenario analysis

| Forecasting models | Scenarios |
|---|---|
| □ Focus on quantified variables | □ Focus on qualitative pictures |
| □ Model based on quantitative data available | □ Quantitative and qualitative information |
| □ More emphasis on accuracy/detail | □ More emphasis on global trends/shocks |
| □ Focus on a partial perspective and certainty | □ Focused on uncertainty |
| □ Results determined by status quo | □ Results determined by future images |
| □ From present to future | □ From future to present |
| □ Deterministic analysis | □ Creative thinking |
| □ Closed future | □ Open future |
| □ Statistical–econometric tests | □ Plausible reasoning |
| □ From simple to complex | □ From complex to simple |
| □ From quantitative to qualitative | □ From qualitative to quantitative |
| □ Many quantitative temporal data required | □ Expert information useful |
| □ Policy analysis based on past experiences | □ Analysis of new policies/instruments |

importance — as a complementary tool — for multidimensional environmental planning problems. In recent global environmental change models and climatological models scenarios have become an intrinsic component to map out uncertain futures. A series of examples can be found in Zwerver et al. (1995).

In view of the uncertainty incorporated in many planning analyses also information systems should be given due attention in environmental planning. This does not only hold true for monitoring systems, but also for decision support systems and expert systems. Clearly, such systems also form an extremely useful contribution to a rationalization of complex planning problems. Examples can be found in land use planning (using geographic information systems - GIS), regional and environmental management and infrastructure planning (see for details Giaoutzi and Nijkamp, 1993).

Thus, both scenario experiments and information systems may provide useful decision support methods for environmental management under uncertainty.

In many practical situations, researchers have to create visions on the future as a frame of reference for judging unexpected developments. Scenarios are different from forecasts, as realism is not necessarily a main feature. The differences between forecasting models and scenarios are illustrated in Table 2 (see Zwier et al., 1995).

It may thus be concluded that scenarios are essentially communication instruments. They aim to explore uncertain futures by depicting the consequences of imaginary (though possible) futures.

## 5. The use of meta-analysis

Recent years have witnessed an increasing interest in generalizing results from case studies. This has led to the popularity of meta-analytic techniques, e.g. in the medical sciences. The application of assessment procedures in environmental analysis is clearly complicated. Nevertheless, over the years very many assessments have been conducted. These obviously vary in their level of sophistication and, needless to say, in their inherent objectivity. In other words we have a considerable, existing body of knowledge upon which we may try to gain additional insights to assist in the development of appraisal methods. This is true even beyond the confines of environmental economic analysis (Oswald, 1991). Of course, gaps in our information frame remain and many aspects of the procedure need further original research but there is, nevertheless, a significant body of existing knowledge which may fruitfully be mixed. Furthermore, when many results are available, sometimes different conclusions or estimates are implied. This raises the possibility for gaining insight by performing meta-analysis based on existing case study results in environmental management.

Generally speaking, meta-analysis offers a range of techniques designed to generate additional information from an existing body of knowledge, and it essentially involves synthesis. Having looked into the nature of environmental evaluation problems, the question emerges as to the role meta-analysis might play in helping improve our understanding of environmental policy measures based on plan or project evaluation methods in a variety of different circumstances.

The use of meta-analysis allows one to assess common features and variations across a range of prior studies (Van den Berg et al., 1997). Its historical basis in the pure sciences, medical science and areas such as social psychology shows that much of the previous work has been concerned with quantifiable, or at least quasi-quantifiable, effects (e.g., Hedges and Olkin, 1985; Hunter et al., 1982; Rosenthal, 1991). Further, although meta-analysis has tended to be used for evaluation in cause–effect types of experimental situations, it has also been adapted to circumstances where there has been felt a need to summarize particular phenomena, for instance, the estimation of point values (e.g., the demand elasticity of a rise in gasoline tax; and monetary values for environmental damage or, alternatively, environmental policy benefits). Studies in the latter areas are still rare (see Smith and Kaoru, 1990; Smith and Chin Huang, 1993; and Smith and Kaoru, 1995).

While relatively little used in the environmental economic field it does have the potential to offer new insights into a number of important areas (Van den Bergh et al., 1995, 1997). Its strength lies not in originality per se but rather in extracting additional information from work which has already been done.

It allows for the useful consideration of the pool of existing work and studies constructed on environmental issues and to draw from this pool common threads, outliers, linkages and generalizable interpretations.

It should be noted that at present there are limitations associated with the use of meta-analysis in the field of environmental policy assessment, and in particular in those areas of policy which are less easily expressed in some quantitative form. These problems related to the qualitative nature of environmental policy problems are, however, largely the same types of problems which existed in the past in econometrics and related fields and which are now routinely handled in the work which is done. The problem may well be quite simply that the efforts to date in applying meta-analysis to environmental policy assessment have focused on issues which can be viewed in qualitative terms, e.g. exploring why studies produce quite wide variations in the values placed on traffic noise nuisance or the value of safety improvements. The quantification issue should not, therefore, be seen as a binding constraint (see also Button and Nijkamp, 1996).

A list, which is in no way meant to be exhaustive, of the types of environmental management and evaluation issue which might be addressed using meta-analysis can be defined within the boundaries of the following criteria. There must be an existing body of studies which can be subjected to statistical procedures in the broadest meaning of the term. The number of studies need not be large (e.g. some medical meta-analyses have involved as few as three studies), but ideally it should not be too small. Normal statistical criteria favor the use of as many observations as possible. Clearly, there are also limitations.

There may be various objectives behind the use of meta analysis in environmental policy assessment. This may be in terms, for example, of attempting to seek out the common treats of previous successful packages of environmental policy measures or it may be in terms of looking for a summary measure from a body of prior analysis (e.g. relating to the damage done to property by particular atmospheric pollution). The need for this clarity of objective relates, in part, to ensuring the minimum bias introduced when selecting the studies for inclusion. There must be a degree of commonality in the prior information to be examined. This may, for example, be spatial, temporal or subject specific, but without this the realm of analysis would be excessively diverse. Defining the extent of commonality is itself a potential problem and is inevitably subjective. In some cases the problem of diversity may be contained if it is possible to isolate the peculiarities of studies and to normalize them in the meta-analysis itself.

The subject matter of the study must be such that it can be handled within a statistical framework, although given the nature of work in the environmental field this may require the application of what might generally be termed 'soft modelling' (Nijkamp et al., 1984). The modelling, however, must go beyond simple data description and statistical tabulations, as there is a need for a more rigorous analysis of various distinct experiences. Within these boundaries there is a wide range of environmental evaluation issues which seem amenable to meta-analysis. We will mention here six particularly promising issues without excluding others (for a complete overview, see Van den Bergh et al., 1997).

First, evaluation of a variety of environmental costs and benefits. Here we are concerned mainly with point estimates which could include such issues as: traffic noise nuisance, accident levels, local air pollution, crop damage, recreation benefits, ecosystem functions (e.g., flood protection, non-use values).

In addition, and using the same basic framework of meta-analysis, the technique could be deployed to explore the differing influence exerted in evaluation of adopting various different methods of valuation (e.g. hedonic property values, contingent valuation or travel cost method) used in policy assessment (in technical terms, this involves seeking moderator variables). For instance, it may be used to explore whether some techniques systematically give higher valuations than others and whether there are area or group biases.

Second, assessment of the effectiveness of alternative policy instruments in containing environmental damage. There is a growing body of individual case studies which have sought to examine how successful different policy strategies (e.g. fiscal policies, regulation, moral suasion) have been in the environmental area. While the meta-analysis may require soft modelling (e.g., qualitative response models), rather than conventional statistical analysis, there is both the background material becoming available and the techniques being developed to conduct fruitful meta-analysis in this field. It is also clearly a subject of considerable practical importance at a time when the traditional command-and-control approach to environmental policy is being supplemented by fiscal instruments. It also has relevance in the context of combining appropriate environmental protection policy with infrastructure investment when the latter is being initiated rather than on a later add-on.

Third, exploration of the appropriate political level of intervention to contain environmental damage. While the number of case studies to bring together here is still relatively small and the application of 'soft modelling' would be inevitable there are now studies which have explained the success in adopting different policy options at the local, state and federal levels, or at a cross-section of different levels.

Fourth, the political acceptability of alternative environmental instruments by decision makers in a topic which has been addressed in a number of studies. There seems, for example, to be a strong traditional bias in favor of using regulatory rather than fiscal tools to continue adverse environmental impacts. The works which have been done in this field are, however, of differing international origin, look at different sets of factors and so on. There is scope for systematically bringing together this information and analyzing it from a generalizable perspective.

Fifth, the levels of various multiplier effects associated with different decisions. Studies have tended to adopt different approaches to measuring these environmental policy effects and to the extent they are traced through the economic and social structure. While not always quantifiable, the importance of making due allowance for these effects at different levels of aggregation justifies exploring the potential for, at least, some qualitative look at what could be achieved using a quasi-meta analysis (i.e., semi-statistical procedures). For example, an important question may be under what conditions do the secondary effects cease to be of any real importance to the outcome of an environmental policy option or strategy.

Finally, physically forecasting the direct non-environmental impacts of any policy is difficult but these impacts are frequently the main determinant of the ultimate scale of the environmental damage done. For instance, environmental economists have expended considerable energies trying to place a money value on factors such as traffic noise nuisance or changes in the probability of fatal accidents taking place but from the overall societal perspective if the actual traffic forecasts are seriously incorrect then the accuracy of these evaluations becomes of secondary relevance. A 5% reduction in the valuation of the cost of noise is easily swamped by a 20% under-prediction of traffic volume. Meta-analysis provides a rigorous basis for improving the forecasts of the direct physical implications of decisions which may then be fed into the assessment process or the evaluation procedure.

The conclusion from the above list of possible environmental application fields of meta-analysis is that there is a vast range of interesting assessment opportunities at different levels of policy-making and for different environmental concerns. Clearly, this list is by no means limitative, but merely illustrative. It illustrates the wide scope of modern meta-analytic methods.

## 6. Conclusions

This paper has shown that the field of environmental modelling and analysis is extremely dynamic and covers a wide spectrum of methodologies. The need to more fully embrace environmental considerations in policy-making is now fully accepted. The practical problems of doing this remain quite formidable although significant progress has been made in recent years. This paper has sought to offer a typology of the issues involved and to offer suggestions as to how we might be able to extend current lines of research, deal with fundamental uncertainty issues and extract additional information from prior studies and policy analyses. This may involve the wider use of model integration, combining physical, biological, spatial-interaction and market-process theories, such as materials flow ecosystem, equilibrium and multi-regional models. In addition, meta-analysis in connection with fuzzy multicriteria analysis and qualitative impact modelling can be useful to extend and combine the results of existing studies characterized by uncertainty. Needless to say, there is still much work to do in this largely unexplored area, in particular in modelling qualitative systems features.

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