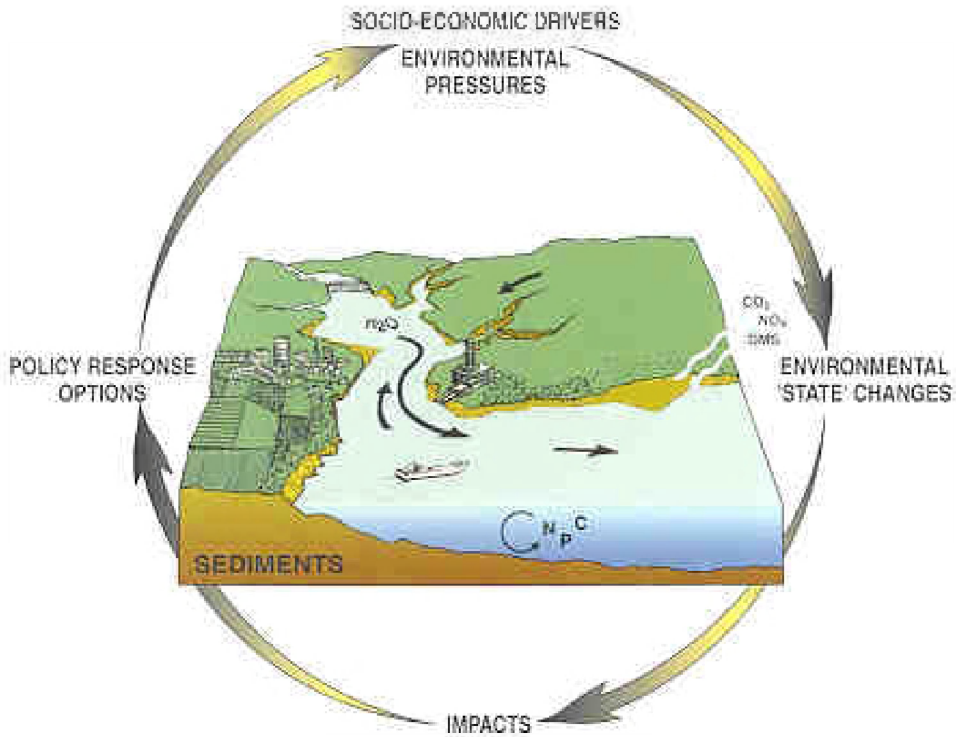
LAND-OCEAN INTERACTIONS IN THE COASTAL ZONE (LOICZ)

Core Project of the
International Geosphere-Biosphere Programme: A Study Of Global Change (IGBP)
of the International Council of Scientific Unions (ICSU)



TOWARDS INTEGRATED MODELLING AND ANALYSIS IN COASTAL ZONES: PRINCIPLES AND PRACTICES

R. Kerry Turner, W. Neil Adger and Irene Lorenzoni

With contributions from: I. J. Bateman, P. Boudreau, B. T. Bower, R. Buddemeier, P. Burbridge, Chan Huan-Chiang, J.I. Marshall Crossland, N. Harvey, P. Holligan, J-L. de Kok, D. McGlone, R. Sidle, S. Smith, K. Takao, N. H. Tri and other participants in the LOICZ workshops held in Norwich, UK and Kuala Lumpur, Malaysia in 1997.

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Cover: The cover design represents the need for combined natural and socio-economic approaches to the P-S-I-R concept in both research and wise management of people and their activities in the coastal zone.

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TOWARDS INTEGRATED MODELLING AND ANALYSIS IN COASTAL ZONES: PRINCIPLES AND PRACTICE

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1. BACKGROUND AND RATIONALE FOR INTEGRATION

All countries with a coastline have an interest in the sustainable management of the coastal resource systems. The task of sustainable management, defined here as sustainable utilisation of the multiple goods and services provision generated by coastal resources (processes, functions and their interrelationships), is likely to be made more difficult because of the consequences of global environmental change (including climate change). The Land Ocean Interactions in the Coastal Zone (LOICZ) Core Project of the IGBP focuses on the role of the world's coastal zones in the functioning of the Earth system: the way in which global changes will influence that role; the way in which such changes will affect the use of coastal space and resources by humanity; and the consequences of such changes for human welfare.

The general goals of LOICZ as stated in the Implementation Plan (Pernetta and Milliman, 1995) are as follows:

I. To determine at global and regional scales:
a) the fluxes of materials between land, sea and atmosphere through the coastal zone
b) the capacity of coastal systems to transform and store particulate and dissolved matter
c) the effects of changes in external forcing conditions on the structure and functioning of coastal ecosystems.

II. To determine how changes in land use, climate, sea level and human activities alter the fluxes and retention of particulate matter in the coastal zone, and affect coastal morphodynamics.

III. To determine how changes in coastal systems, including responses to varying terrestrial and oceanic inputs of organic matter and nutrients, will affect the global carbon cycle and the trace gas composition of the atmosphere.

IV. To assess how responses of coastal systems to global change will affect the habitation and usage by humans of coastal environments, and to develop further the scientific and socio-economic bases for the integrated management of the coastal environment.

Understanding the interactions between the coastal zone and global changes cannot be achieved by observational studies alone. Modelling of key environmental processes is a vital tool that must be used if LOICZ is to achieve its overall goals and objectives, particularly in view of the fact that many of the uncertainties in global carbon flow models may represent unquantified processes occurring within the coastal zone. This document is intended as an initial guide for those wishing to contribute to the objective of combining modelling work on the dynamics of carbon, nitrogen, phosphorus, sediments and water in the coastal ocean with socio-economic analysis of the drivers of C, N, P and sediment fluxes and the human welfare consequences of the changes in C, N, P and sediment fluxes in the coastal zone over time.

The LOICZ approach is to encourage researchers around the world to develop models of the fluxes of C, N, P and sediment for their local geographic areas of interest. If constructed in a similar manner, these models would provide estimates which can be aggregated at regional and global scales. For any group of researchers wishing to investigate and model a particular local coastal system (or aspects of that system) for subsequent scaling up into larger models or wider regional estimates, there are initially two types of information required:

- estimations of biogeochemical fluxes in the system as it is now, for eventual incorporation into global estimates of flux through the coastal zone; and
- dynamic simulations of processes in the coastal system which can be used to explore the consequences of environmental change, and produce forecasts of future fluxes.

The second type of information set will require the integration of socio-economic and natural science data and models in two basic analytical contexts:

- to provide an understanding of the external forcing effects of socio-economic changes such as, for example, population growth, urbanisation, and other land use changes on fluxes of C, N, P and sediment; and
- to assess the human welfare impacts of flux changes due to changes in processes and functions in coastal resource systems. Such assessments of the social costs and benefits involved will provide essential coastal management intelligence based on social science and possible resource and value trade-offs.

The second analytical context poses a more formidable research task, not just because of the data requirements and the problem of integration involving data which differ in form and in spatial and temporal scale, but because the long-term goal is the development of an integrated prognostic assessment capability. LOICZ does not underestimate the difficulties and challenges that are posed and has sought to evolve a tactical and strategic approach centred on the initial development of simple budget models for water, nutrients and other materials and the production of biogeochemical flux budgets of the system, integrating over annual or multi-annual scales. The comparison of fluxes through systems that differ in certain environmental parameters should allow tentative predictions of the consequences of environmental change.

In the short term, budget calculations and empirical models are likely to have a greater value as predictive tools of different management strategies and different environmental change scenario simulations, reflecting, for example, predicted population and land use changes. Empirical models usually possess only limited predictive capabilities, defined by the range of observed data for which they were generated. As a result process-oriented system models also need to be developed in order to evaluate the effects of perturbations (linked to the socio-economic drivers of population, economic activity, land use and land cover changes in the relevant drainage basins) outside the range for which there exists C-, N- and P- related and other empirical data.

In the longer term, an holistic approach is necessary (combining process-oriented and empirical models) where the goals of the models and the critical scales are defined prior to model formulation and simulation. The development of more detailed models, through a number of iterative steps where model evaluations interact with field measurements, will only be possible for coastal regions possessing the necessary scientific, data-base and institutional capacities. The development of typological relationships among coastal regions could play a role in exporting such detailed models to other areas.

Since most model development under LOICZ is expected to be supported primarily at the national or regional level, the spatial extent will probably be relatively small in order to address important local management issues. In order to meet the long-term, global objectives of LOICZ, it is essential to scale up or compile the results of these local and regional models into global coastal zone models.
For LOICZ, the scaling procedure may be accomplished in a number of ways. A preliminary step has involved the development of a **Coastal Typology** based on the central objective of categorising the world's coastal zone on the basis of national features, into a realistic number of geographic units, which will serve as a framework for, among other things:

- organisation of data bases
- selection of regions for extensive studies (remote sensing, long term monitoring)
- selection of appropriate sites for new studies
- scaling local and regional models to regional and global scales
- analysis, compilation and reporting of LOICZ results in the form of regional and global syntheses.

The general scheme for the classification and development of models under this initiative utilises existing data sets and site-based studies in an attempt to apply them to data-poor sites or situations. Ideally, natural and environmental data combined with demographic and other socio-economic data within the typology would provide proper descriptions of the functioning of the world's coastal zones. LOICZ has concluded that it would be inappropriate to develop a single typology that would meet the needs of all LOICZ applications. Multiple typologies, linked and made compatible by common data and conceptual elements, are seen as the logical suite of products to meet the goals of the LOICZ Implementation Plan (see Appendix 1).

The assessment of the impact of changes in the coastal zone on human use of resources (wealth creation) and habitation (quality of life aspects) requires a further element in LOICZ's modelling and assessment strategy - the application of socio-economic research methods and techniques in the context of coastal resource assessment and management. A particular contribution of socio-economic research is the incorporation of evaluation methods and techniques which can be applied to specific resource damage and utilisation situations (projects, policies or courses of action which change land use/cover, alter or modify residuals from point and non-point sources etc.) because of C, N and P flux changes and related consequences, including loss of functions and even habitats. Again most of these valuation studies will be at a local/regional level and the same scaling up problem presents itself. However, the transfer of economic valuation estimates (known as benefits transfer) across time and geographical and cultural space is controversial.

The last 20 to 30 years have seen the gradual evolution of a strategy aimed at an integrated assessment of environmental science, technology and policy problems. A multi-disciplinary tool kit has been presented whichglobal climatic change researchers, for example, have tapped into (Schneider, 1997). An integrated assessment framework must include integrated or coupled models (biogeochemical and socio-economic) but it is not limited to just this. According to Rotmans and Van Asselt (1996) integrated assessment is "an interdisciplinary and participatory process of combining, interpreting and communicating knowledge from diverse scientific disciplines to achieve a better understanding of complex phenomena". The critical importance of making value-laden assumptions highly transparent in both natural and social scientific components of integrated assessment models (IAMs) needs to be highlighted; practitioners now argue that incorporating decision-makers and other stakeholders into the early design of IAMs greatly facilitates this process. Valuation in this process is more than the assignment of monetary values and includes multi-criteria assessment methods and techniques to enable identification of practicable trade-offs. The LOICZ work should therefore be seen as fundamental but also rudimentary as far as fully-fledged integrated assessment is concerned.

In summary, progress in integrated modelling/assessment is required particularly in relation to two LOICZ general goals:

- the determination of how changes in land use, climate, sea level and human activities alter the fluxes and retention of particulate matter in the coastal zone, and affect coastal morphodynamics; and
- the assessment of how responses of coastal systems to global change will affect the habitation and usage by humans of coastal environments, and to develop further the scientific and socio-economic bases for the integrated management of the coastal environment (Pernetta & Milliman (1995) LOICZ Implementation Plan).

The various elements in the LOICZ strategy leading to the eventual achievement of these goals can be summarised in simplified form in Table 1.1.

Table 1.1. Elements of the LOICZ modelling/assessment strategy

- stimulation of a large number of local/regional C, N & P budget models carried out on a consistent basis to allow for eventual scaling up
- coastal typology development and other approaches to facilitate scaling up; initial construction of input, transformation and exchange component typologies
- continuing development of more detailed models to increase predictive capability
- first order incorporation of socio-economic data and environmental change scenarios into models in order to understand current and predict future C, N & P fluxes
- possible expansion of typology to increase the comprehensiveness of the input component typology; and possible creation of a human welfare-related component typology
- incorporation of socio-economic analysis into models to predict future human welfare consequences of changes in C, N & P fluxes; scaling issue.

In the next section, the requirements of and guidelines for a more integrated approach to coastal resources assessment will be presented. Elements of the basic analytical framework presented are based on discussions at and contributions to the Norwich 'Integrated Guidelines' workshop, held in March 1997 and the SARCS/WOTRO/LOICZ Workshop on Integrated Modelling Guidelines, Kuala Lumpur, July 1997. These elements have subsequently been amended and incorporated with concepts and methods presented in previous LOICZ reports (see Pernetta and Milliman, 1995; Gordon *et al.*, 1996; Buddemeier and Boudreau, 1997; Turner and Adger, 1996).

2. TERMS AND DEFINITIONS

2.1 Introduction: Global Environmental Change

Global Environmental Change (GEC) is a cumulative process of change, driven by human use of environmental space and resources, these pressures being particularly intense in coastal areas around the globe. The pressures may result in changes to the Earth system which in turn will impact on future human use of coastal space and resources (thereby affecting human welfare (in terms of wealth creation and the quality of life). LOICZ is a project designed to improve our scientific understanding of this global feedback loop and hence provide a sound scientific basis for the sustainable management of the world's coastal areas. Progress is therefore required in methods and techniques that will enable the formulation and testing of a more integrated coastal resources assessment. Socio-economic systems and 'natural' systems are, to a variable extent, now locked into a co-evolutionary path, characterised by joint determinism and complex feedback effects. Modelling and assessment exercises should, over time, be reoriented to properly capture the causes and consequences of the joint system change as manifested in coastal areas. This will require collaborative work among a range of science and social science disciplines.

A particular characteristic of modern economic development (encompassing population and population density increases, urbanisation, intensification of agriculture and industrial processing) is that it has led to the progressive opening of biogenic nutrient cycles e.g. much increased mobility of nitrogen and phosphorus. The increased mobility of nutrients has meant increased exchanges between land and surface water and consequent impacts on ecological functioning of aquatic systems.

The coastal interface between the continents and the ocean is comprised of a continuum of aquatic systems including the network of rivers and estuaries, the coastal fringe of the sea, the continental shelf and its slope. These interdependent systems are characterised by very significant biogeochemical processes - primary productivity generation, organic matter and nutrient sinks for example. Significant inputs of nutrients to the coastal zone arrive via rivers, groundwater, and the atmosphere. The major flux of nutrients from land to sea occurs through river transport, via the drainage basins network. The network contains various 'filters' (e.g. wetlands) retaining or eliminating nutrients during their downstream passage to the sea. The effectiveness and selectivity of these filters depend on the strong biogeochemical coupling that exists between carbon, nitrogen, phosphorus and silica circulation and they are also affected by hydrology and land use/cover (Howarth *et al.*, 1996). Nutrient fluxes have been increased by human activity; in addition, the N:P:Si ratios of these inputs have been perturbed and many coastal management practices exacerbate these perturbations. There is evidence of impacts arising from these changes in areas of restricted water exchange (Jickells, 1998).

2.2 Pressure-State-Impact-Response (P-S-I-R) Framework

A useful starting point for both LOICZ natural and social science research would be to seek (via a more integrated modelling and assessment process) to better describe and understand the functioning of the ecosystems forming the coastal interface, and in particular the filter effect it exerts for nutrients in response to environmental pressures, both anthropogenic and non-anthropogenic - climate change, land use/cover change, urbanisation and effluent treatment from both point and non-point sources. But first we need some broad analytical framework (rather than a specific model) in which to set the more detailed analysis.

The P-S-I-R cycle offers such a generalised context and Figure 2.1 illustrates the approach for a coastal zone and linked drainage basin.

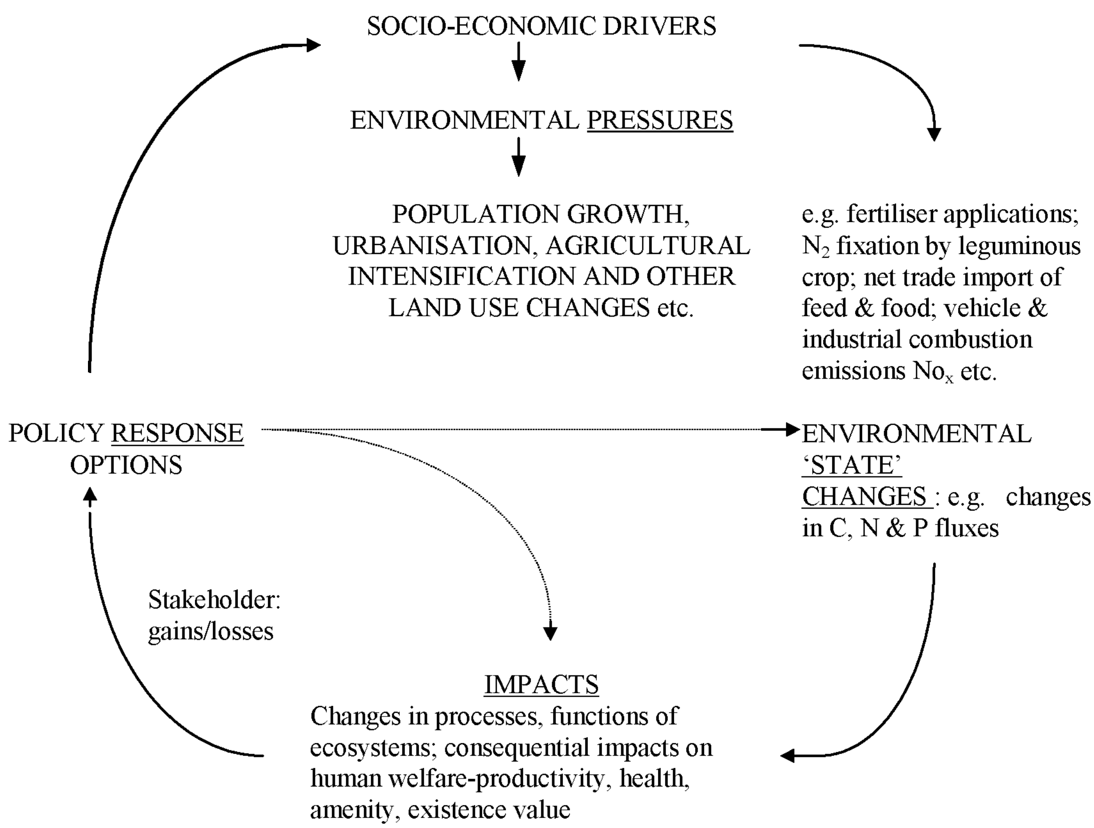
This framework provides a way of identifying the key issues, questions, data/information availability, land use patterns, proposed developments, existing institutional frameworks, timing and spatial considerations etc. (Turner *et al.* 1998).

For any given coastal area (defined to encompass the entire drainage network) there will exist a spatial distribution of socio-economic activities and related land uses - urban, industry mining, agriculture/forestry/aquaculture and fisheries, commerce and transportation. This spatial distribution of human activities reflects the final demand for a variety of goods and services within the defined area and from outside the area. Environmental pressure builds up via these socio-economic driving forces causing changes in environmental systems states.

The production activities result in different types and quantities of residuals, as well as goods and services measured in Gross National Product (GNP) terms. LOICZ is particularly concerned with changes in C, N, P and sediment fluxes as a result of land use change and other activities. Environmental processes will transform the time and spatial pattern of the discharged/emitted residuals into a consequent short-run and long-run time and spatial ambient environmental quality pattern.

These state environmental changes impact on human and non-human receptors resulting in a number of perceived social welfare changes (benefits and costs). Such welfare changes provide the stimulus for management action which depends on the institutional structure, culture/value system and competing demands for scarce resources and for other goods and services in the coastal zone. An integrated modelling approach will need to encompass within its analytical framework the socio-economic and biophysical drivers that generate the spatially distributed economic activities and related ambient environmental quality, in order to provide information on future environmental states.

Figure 2.1 P-S-I-R Cycle and Continuous Feedback Process



Source: Adapted from Turner *et al.* (1998)

2.3 Sustainable Coastal Development

Sustainable coastal development can be described as 'the proper use and care of the coastal environment borrowed from future generations'. The concept of sustainable development achieved global attention following the World Commission on Environment and Development (WCED) report known as the 'Brundtland Report' or 'Our Common Future' (WCED, 1987). This was given further impetus at the United Nations Conference on Environment and Development (UNCED), also known as the 'Earth Summit' in Rio de Janeiro, with the production of Agenda 21 (UNCED, 1992). Agenda 21 has a separate chapter (17) relating to coastal management.

Sustainable development was defined by the WCED as that which "meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987 p8) and it was suggested that economic development and environmental well-being are not mutually exclusive goals. The key elements of sustainable development relate to the concept of needs and the restricted environmental ability to meet these needs, both present and future. Sustainable development is a process of change in which "the exploitation of resources, the direction of investments, the orientation of technological development, and institutional change are all in harmony and enhance both current and future potential to meet human needs and aspirations" (WCED, 1987 p90).

In order to achieve critical sustainable development objectives for coastal environments and coastal development policies, it is important to have strategies such as: conserving and enhancing the coastal environment, managing risk and coastal vulnerability, and merging coastal environmental considerations with economics in decision making. At the UNESCO conference on Coastal Systems Studies and Sustainable Development, it was noted that modern industrialised development and associated population growth have subjected coastal environments to severe pressures and degradation through over-exploitation of resources, pollution of various kinds and destabilisation of the littoral zone, as well as through more global climatic and other changes. Similarly it was pointed out that there is a spread of modern-style industrialised development problems such as congestion, pollution and high resource consumption into the coastal zone which contain some of the richest and most diverse resource areas of the planet.

The need for sustainable development was given its strongest support by UNCED at the Earth Summit, which included four main agreements: the Rio Declaration on Environment and Development; the Framework Convention on Climatic Change; the Convention on Biological Diversity; and Agenda 21. All of these impact in some way on coastal environments. The Rio Declaration contains 27 principles relating to international behaviour in relation to development and the environment and requires all nations to co-operate in trying to achieve sustainable development. The Framework Convention on Climatic Change is directed towards reducing harmful emissions of greenhouse gases and specifically mentions regional programmes to lessen the effects of climatic change and the need to incorporate climatic change into policies and actions. These are directly relevant to coastal sustainability in terms of greenhouse sea-level rise predictions. The Biological Diversity Convention which refers to ecosystem, species and genetic diversity is important in the sustainability of coastal ecosystems, where there is greatest pressure of population growth and development. Agenda 21 is a complex 800-page action plan on global environment and development for the 21st century which contains reference to the sustainable use of ocean and coastal resources. In order to follow up on Agenda 21 a Commission on Sustainable Development has been created within the United Nations.

Sustainable development has been approached in different ways around the world. However, there is a danger in using a sectoral approach to holistic environmental matters such as sustainable development. For example, in Australia nine sectoral reports on sustainable development were found to be so lacking that it was necessary to set up 37 inter-sectoral groups including one on coastal development.

2.4 Sustainable Development Policy Objective

Sustainability from an economic perspective requires a non-declining capital stock over time to be consistent with the criterion of intergenerational equity. Sustainability therefore requires a development process that allows for an increase in the well-being of the current generation, with particular emphasis on the welfare of the poorest members of society, while simultaneously avoiding uncompensated and 'significant' costs on future generations. Policy would be based on a long-term perspective, incorporating an equity as well as an efficiency criterion, and would also emphasise the need to maintain a 'healthy' global ecological system.

The **'constant capital'** condition for sustainable development can be interpreted in a weak and strong form. The **weak sustainability** condition can be written as

$$K/N = \{K_m + K_h + K_n + K_{sm}\}/N \qquad (4)$$

(4) should be constant or rising over time.

The **strong sustainability** condition in its environmental form should be:

K_{n}/N constant or rising over time (5); **and** weak sustainability (4) must also hold

where K_m = man-made capital
K_h = human capital
K_n = natural capital
K_{sm} = social/moral capital
N = population

Weak sustainability effectively assumes unlimited substitution possibilities (via technical progress) between the different forms of capital. Strong environmental sustainability assumes that natural capital (or 'critical' components of such environmental systems) cannot be substituted for by other forms of capital.

Because the coastal zone is the most biodiverse zone, a strong sustainability strategy would impose a 'zero net loss' principle or constraint on resource utilisation (affecting habitats, biodiversity and the operation of natural processes). Wetlands, for example, provide a range of valuable functions and related goods/services flows. Such systems have also been subjected to severe environmental pressures and have suffered extensive degradation and destruction. They may therefore be good candidates for a 'zero net loss' rule depending on how critical the functions and systems involved might be. The opportunity costs of the wetland conservation policy (i.e. foregone development project net benefits) should be calculated and presented to policy makers. If the wetland area requires a more proactive management approach i.e. buffer zone creation, monitoring and enforcement costs, then an aggregate valuation calculation will be required.

2.5 Resource Valuation

Given the P-S-I-R analytical framework there is a further requirement for a conceptual model which can formally link natural science to social science and to the different dimensions of environmental/social values. The functional diversity concept is a key feature of the required approach because it can link ecosystem processes and functions with outputs of goods and services, which can then be assigned monetary economic and/or other values, see Figure 2.2. Functional diversity can be defined as the variety of different responses to environmental change, in particular the variety of spatial and temporal scales with which organisms react to each other and to the environment (Steele, 1991). Marine and terrestrial ecosystems differ significantly in their functional responses to environmental change and this will have practical implications for management strategies. Thus although marine systems may be much more sensitive to changes in their environments, they may also be much more resilient (i.e. more adaptable in terms of their recovery responses to stress and shock). The functional diversity concept encourages analysts to take a wider perspective and examine changes in large-scale ecological processes, together with the relevant socio-economic driving forces. The focus is then on the ability of interdependent ecological-economic systems to maintain functionality under a range of stress and shock conditions (Folke, Holling and Perrings, 1996).

A note of caution is also necessary to alert researchers to an operationally significant problem associated with terminology across the disciplines which require integration. Some agreed terminology is necessary to facilitate the modelling exercises, see Box 2.1 and Figure 2.2.

Box 2.1 Definitions and Terminology

- Problem Orientation

Any assessment should take account of the prevailing political economy context, equity issues and possible 'stakeholder' interests. Data limitations must be acknowledged and recommendations made conditional upon these.

- Typology

A useful common terminology which regards **processes and functions** as relationships within and between natural systems; **uses** refer to use, potential use, and non-use interactions between human and natural systems; and **values** refer to assessment of human preferences for a range of natural or non-natural 'objects' and attributes.

- Scale

The spatial, temporal, quantitative, or analytical dimensions used to measure and study any phenomenon. The size of the spatial, temporal, quantitative or analytical dimensions of a scale is termed its extent.

The drainage network should be the spatial unit for assessing ecological variables, with possible zonation within this. In terms of benefit estimation, the minimum extent is determined by the relevant population affected by any impacts. Temporal scale and extent of analysis is also fundamentally important.

- Thresholds

These relate to the extent and frequency of impacts. Their occurrence can be presented in a simple three-part classification: no discernible effects; discernible effects; discernible effects that influence economic welfare.

- Economic Valuation

Three broad approaches to a valuation exercise: impact assessment; partial analysis; and total valuation. For each function or impact, a number of techniques exist for attributing economic value to environmental benefits.

- Transferability

Transferring scientific results across sites is required for global scaling but transfer of some economic benefits is problematic. Accuracy of benefits transfer may be improved if based on scientific variables divided into separate components depending on processes, functions, and 'state variables'.

Source: see Ahn, Ostrom and Gibson (1998) for a summary.

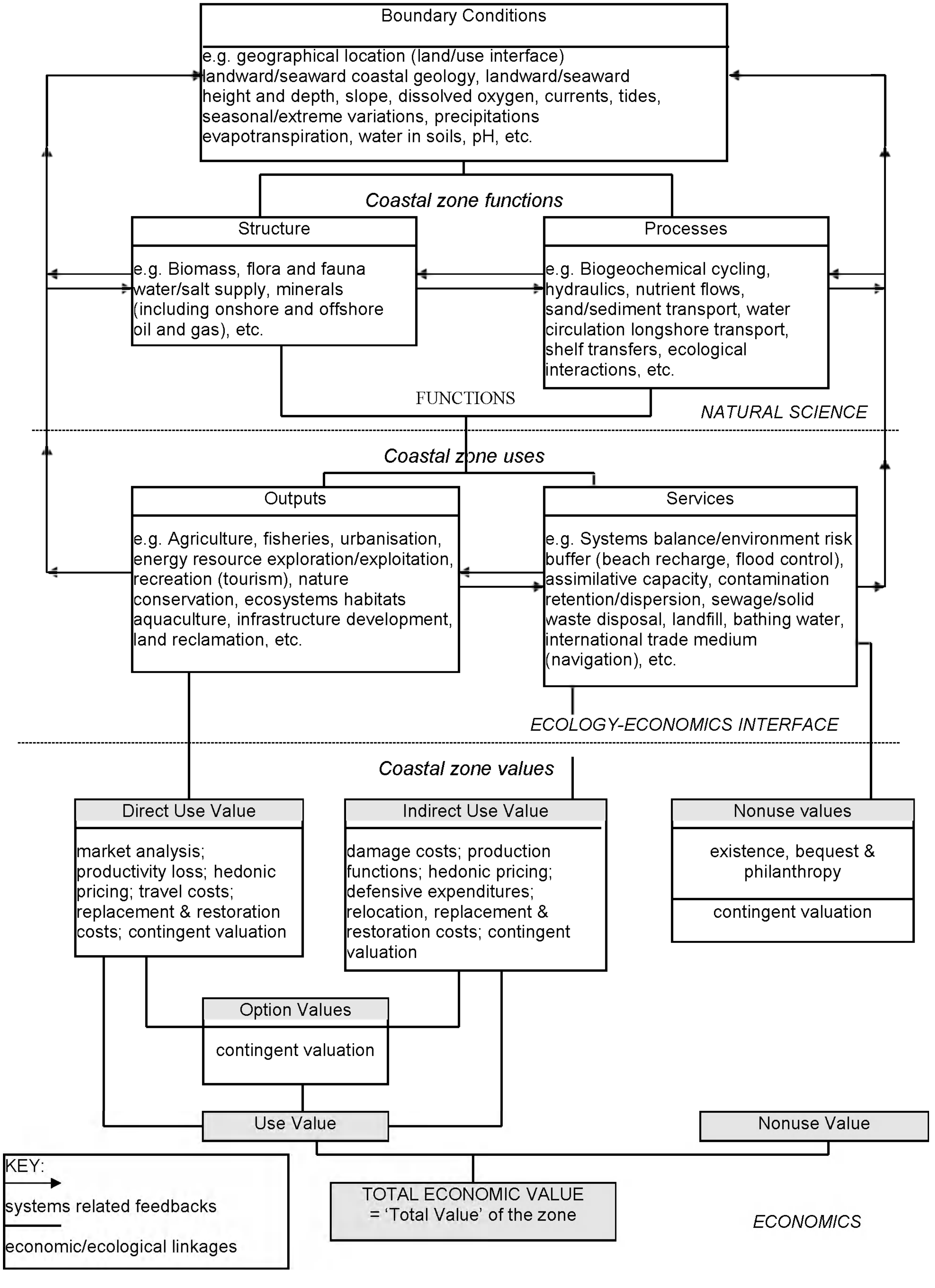
The choice of resource valuation approaches will consequently depend on the spatial extent of the cause and effect relationship subject to assessment:

- **impact analysis**: related to identified impacts generated by nutrients flux change and other state changes usually within a restricted spatial area, but sometimes requiring drainage basin-wide data/analysis;
- **partial valuation analysis**: of given ecosystems, their functions and valued outputs,normally requiring more extensive spatial area analysis;

and

- **total valuation analysis**: of a defined and perhaps very extensive coastal marine area.

Figure 2.2 Coastal zone functions, uses and values



In addition, problems may arise when datasets based on different timescales have to be related to each other. From a statistical point of view, any data with associated confidence intervals can be incorporated into the studies. Therefore, the characteristics and relative advantage of the information used should be clearly stated, preferably in comparison to other datasets that could have been equally relevant in that specific project/area (e.g. annualised/homogenised data versus extreme event data; cross-section versus longitudinal data; discounting).

2.6 Programme Level Sustainability Rules

Instead of just concentrating on single or a small number of impacts and their human welfare implications as discussed above, it is possible to take a more comprehensive and strategic approach across a set of pressures throughout the coastal zone and connected drainage basins in line with the P-S-I-R framework. The constant natural capital rule at this programme level can be interpreted as a process of netting out environmental damage costs (NB_{fet} + TEC_t) across a set of activity changes, such that the sum of individual damages should be zero or negative (Barbier *et al.*, 1990).

$$\sum_i E_i \leq 0$$

where E_i = environmental damage (NB_{fet} + TEC) generated by the ith change.

Under a strong sustainability rule, $\sum E_i$ is constrained to be non-positive for each period of time. If it is not feasible for E_i to be zero or negative for all activities, it may be possible to include within any portfolio of projects, one or more **shadow projects**. These shadow projects aim to compensate for the environmental damage generated by the existing/planned set of economic activities, and are not subject to normal cost-benefit rules.

Environmentally compensating project(s), j, would be chosen such that for strong sustainability:

$$\sum_j A_{jt} \geq \sum E_{it}\ , \forall t$$

where A_j = net environmental benefits of jth project.

Thus the loss of a wetland at some particular location may be compensated for by wetland relocation, creation or restoration investments elsewhere in the zone (the concept of 'strategic mitigation'). This shadow project rule as an interpretation of how to implement sustainability has been adopted by various coastal zone management agencies. But it remains controversial, and may form only one element in management for sustainability. In the LOICZ context it is important that the sustainability of the whole coastal system be incorporated into analysis of Impact and Response.

2.7 Indicators

For the purpose of developing the systems model representing the P-S-I-R Framework, indicators identifying three complementary sets of factors need to be identified, namely:

1. Bio-geochemical and physical fluxes represented by C, N, P, water, sediments and other factors which influence the state of coastal systems, the functions they perform and resources generated (LOICZ objective I).

2. Economic fluxes relating to changes in resource flows from coastal systems, their value and changes in economic activity (LOICZ objectives II and IV).

3. Social fluxes - e.g. food supply and price relating to food security, public health, welfare, flooding hazards (LOICZ objective IV).

The indicators chosen should be easily understood by the different disciplines contributing to the formulation of the systems model and should foster communication across disciplines to support inter-disciplinary integration. The actual establishment of such indicators will require focused, inter-disciplinary research linking natural and social sciences.

Three levels of indicators may be considered. The first are C, N, P, sediments, water flows and financial/economic indicators such as values in terms representing changes in value related to physical, chemical, biological fluxes. The second level of indicators would be represented in changes in coastal system properties for example, primary production, rates of sedimentation, reduction in habitat functions supporting fish stocks. The third level of indicators would relate to the implications for human welfare conditions resulting from fluxes. Examples would include changes in fish stocks and therefore productivity and economic value, public health, environmental amenity and more philosophical/moral aspects of environmental change.

The identification of relevant indicators should also reflect the LOICZ programme's design of developing the interdisciplinary science to measure fluxes, to interpret the significances of those fluxes in the state or condition of coastal ecosystems/environments and the implications for the human condition which can help inform policies, management and investment. This can be viewed as a sequence of 3 phases of end uses, namely:

1. Development of a global model of major fluxes by the amalgamation of information on C, N, P and other key variables based on original case studies and established data bases (LOICZ objectives I, III);

2. Translation of the information on fluxes to determine the state of coastal systems (LOICZ objective II); and

3. Interpretation of state of coastal systems and implications for human use of the resources generated by coastal systems in meeting social and economic development objectives (LOICZ objective IV).

Indicators (3 sets of factors) required are:

1. Biogeochemical and physical fluxes (state flux)

2. Economic fluxes - changes in resource use (if related to (1)).

3. Social fluxes – e.g. food supply (human welfare significant).

Three levels are represented:

1. CNP/$ represent change in value

2. Change in coastal system properties (rates of sedimentation)

3. Implication for human welfare (change in fish stocks)

Three phases need to be undertaken:

- Global model of major fluxes (goals (I) + part (III));

- Fluxes, their changes and the consequent effects on the state of the coastal system (goal II); and

- Interpretation of resulting environmental state and the implications for human resource use and the achievement of socio-economic objectives (goal IV).

2.8 Illustrative Contexts

In order to move from a general level of analysis to the more practical and detailed level required by researchers actually conducting research of relevance to LOICZ, some illustrative study area contexts and case studies will be useful. These contexts and case studies, however, have to be both relevant to and integrated into the existing LOICZ modelling and typology research strategy.

Integrated assessments of the coastal zone will be representative of certain subsets of environmental and human welfare contexts that reflect the particular times and places at which they were undertaken. The LOICZ typology can be used both for demonstrating the range of coastal contexts for which the results of assessments are available (as well as indicating gaps in knowledge) and for extrapolating from the results of particular investigations to wider relevant spatial and temporal scales. Such analyses, which are themselves a topic needing further research, represent the means for advancing LOICZ investigations on integrated assessment to a more practical and relevant level.

The contexts for assessment are most usefully defined in terms of a matrix of environmental and socio-economic indicators. The matrix should be constructed in such a way to enable scaling up from the results of assessments in order to take account of continuing and future changes in the boundary conditions or drivers that define any given type of context.

The primary environmental indicators are physical: **Climatic** - temperature (tropical to polar), precipitation (wet to arid), and wind direction and strength (ocean dynamics, atmospheric transport); **Topographic** - continental margin type (passive to active) and relief (high to low), rock type (hard to soft), morphological features – e.g. deltas, lagoons; **Dynamic state** - variability in physical conditions (seasonal climatic and hydrological extremes, coastal uplift or subsidence), trends in physical conditions (global warming, sea level change, sediment starvation etc.).

For coastal systems that are unperturbed by human activities, these factors define the ecological state, the boundary conditions of inputs of energy and materials (e.g. from the ocean and from catchment systems) and, therefore, the biogeochemical fluxes of central interest to the LOICZ project. The ecological state incorporates properties such as biological productivity, biodiversity, and ecosystem sensitivity to environmental change.

Socio-economic indicators include:

Population density and growth rate in the coastal zone;

Gross National Product *per capita* (economic activity);

Waste emissions/discharges.

The number of contexts that might be defined in this way is potentially large, so that the LOICZ typology would be used to identify those that can be merged for practical purposes and to prioritise those that are significant in terms of global change and biogeochemical properties.

Examples of illustrative contexts that meet such criteria are:

Semi-arid coastal areas subject to intense tourism pressures and hydrological perturbation, both at or near shorelines (e.g. Mediterranean)

Tropical and sub-tropical areas experiencing rapid change via population growth, urban development and economic activity, with supporting infrastructure (e.g. SE Asia)

Deltaic areas subject to the impacts of rapid land use/cover changes (e.g. Nile)

Areas with rich natural resources that are now being exploited in a non-sustainable manner (e.g. SE Asia)

Low-lying coastal regions at risk from flooding due to sea level rise, subsidence and storms (e.g. Bay of Bengal)

Enclosed and semi-enclosed coastal seas where changes in biogeochemical fluxes have large scale effects on hydrographic properties (e.g. Baltic, Black Sea).

3. MODELLING PROCEDURES

3.1 Generic stages

Three overlapping procedural stages can be identified in the process by which more integrated modelling and resource assessment can be achieved:

- Scoping and resources audit stage
- Actual modelling stage
- Evaluation stage

(a) **Scoping and audit**

Initially the problem needs to be formulated i.e. the significant issues to be included in the system study need to be identified in order to fix the scope of the research to be carried out. More specifically, the problem formulation should result in (Miser and Quade, 1985):

- definition of problem owner and problems
- identification of system boundaries
- inventory of constraints
- identification of objectives
- identification of decision criteria and values

The best starting point in any overall modelling strategy is to generate a basic description of the particular coastal system being studied (including the socio-economic activity levels present and predicted [the pressures]). In some cases it may be possible to compare the system to be studied with similar systems that have already been well described and understood. The answers to the basic scoping questions will influence the type of model to be used, data collection/analysis and impacts evaluation requirements. They will also *inter alia* raise **'scale' issues**, including the problem of defining system boundaries and the temporal extent.

Because LOICZ requires regional and global estimates of flux a coastal typology effort has been mounted. The intention of such a typology will be to subdivide the world's coastal zone into clusters of discrete, scientifically valid units, based on both natural and socio-economic features and processes (see Table 3.1 for examples)

Table 3.1 Factors relating to the definition of the coastal zone areas for the development of coastal budget models

Physical Description	
Topography/bathymetry	Shelf edge, bay mouth, estuary, coastal lagoons
Current system	Tidal excursion, boundary of residual circulation
Gradient of material concentration	Frontal structure
Energy regime	Tidal or river dominated, waves, currents, closed
Drainage basin	Soil type, runoff, input of dissolved & particulate material
Biological Description	
Habitat type	Coral reef, seagrass, mangrove, salt marsh
Biological production	Length of growing season, production
Chemical Description	
Nutrients	C,N,P concentration and flux
Socio-Economic Description	
Demographics	Population density, growth
Land use	Land cover, crop type, human activity

Given the LOICZ strategy of concentrating on C, N, P and sediment fluxes, the initial requirement of a more integrated modelling/assessment approach is the compilation and analysis of more comprehensive socio-economic pressures data sets. These data could then be fed into the coastal typology exercise and into nutrient balance modelling. The data represent environmental pressure in the form of residuals generated by populations and their economic activities, together with land use changes. Conceptually what we have are a multiplicity of input-output (IO) relationships (see Appendix 2), with the outputs being joint products (combinations of goods and services and non-product outputs or residuals which if not recycled become wastes emitted/discharged into the ambient environment). We will have IOs at the individual industrial process/plant level, through population settlements IOs, agricultural cropping regimes/practices IOs, and up to regional scale IOs. These residual estimates will serve as the input to the natural science nutrient budget models.

Pressures data in summarised form is represented by GNP calculations for countries and regions. There are also World Bank forecasts of future GNP on a national basis. Food and fibre consumption and land use statistics can be obtained from the FAO computerised database Agrostat (FAO Agrostat Database, 1990 FAO, Rome). Some analysts have used supply as defined by FAO as a measure of total consumption, rather than direct or actual *per capita* consumption. Supply data includes losses incurred e.g. on storage, transport and processing.

Population growth and density data are available nationally, regionally and in global data sets such as United Nations Environment Programme, 'Environmental Data Report' (annual) and World Resources, Guide to the Global Environment; (biannual - also available on diskette). At the regional level, population data, for example, for the Baltic Sea Drainage Basin (an extensive area containing 29 cities with a population of 250,000 or more) has been collected by Swedish researchers (see Sweitzer *et al,* 1996; World Bank, World Development Report (annual - also available on diskette).

Tourism data can be found in World Tourism Organisation, Year Book of Tourism Statistics, WTO, Madrid; and for Europe in EEATF (1995) Europe's Environment: the Dobris Assessment Report, EEA, Belgium.

Data on shelf sea areas and marine exclusive economic zones can also be found the World Resources Institute Diskette Database. Overall, given the range and amount of data requiring collection, GIS applications will be essential.

The scoping stage is also an appropriate time for researchers to consider the predictive capability of their analytical approach. From the pressures side, an element of prediction can be introduced by the identification of trends in GNP, population, land use/cover change, urban settlements and other factors (trend scenarios) and the feeding of these into N and P budget calculations. The trend scenarios, once established, could then be compared with alternative futures scenarios e.g. low growth, medium growth, high growth variants. In studies of a more localised nature, e.g. bays or estuaries within drainage networks, different management strategies might be modelled and compared.

In summary, the scoping/audit phase should raise, among others, the following fundamental issues/questions/problems:

- the need for, and feasibility of, a basic characterisation of the study area encompassing both natural science and social science (socio-economic activity patterns and drivers) data;
- the extent of scale, particularly the system boundaries for the proposed study;
- the modelling/analysis goals, the need for, and feasibility of, some predictive power in the analysis to be adopted e.g. via environmental change scenarios, management strategies;
- the contribution the chosen study can make to the scaling-up process and the typology exercise.

(b) **Modelling Stage**

Based on the identified structure of system elements and their cause-effect relationships one can draw a schematised *causal diagram* for the system, which provides a rough visualisation of the qualitative structure of the coastal-zone system. Causal diagrams must be considered as preliminary modelling tools which only serve the purpose of clarifying what interactions must be modelled. This means that the perfect causal diagram does not exist, and attempts to improve the diagram should terminate at a certain point. When necessary, imperfections in the diagram can be corrected during later phases of the modelling process.

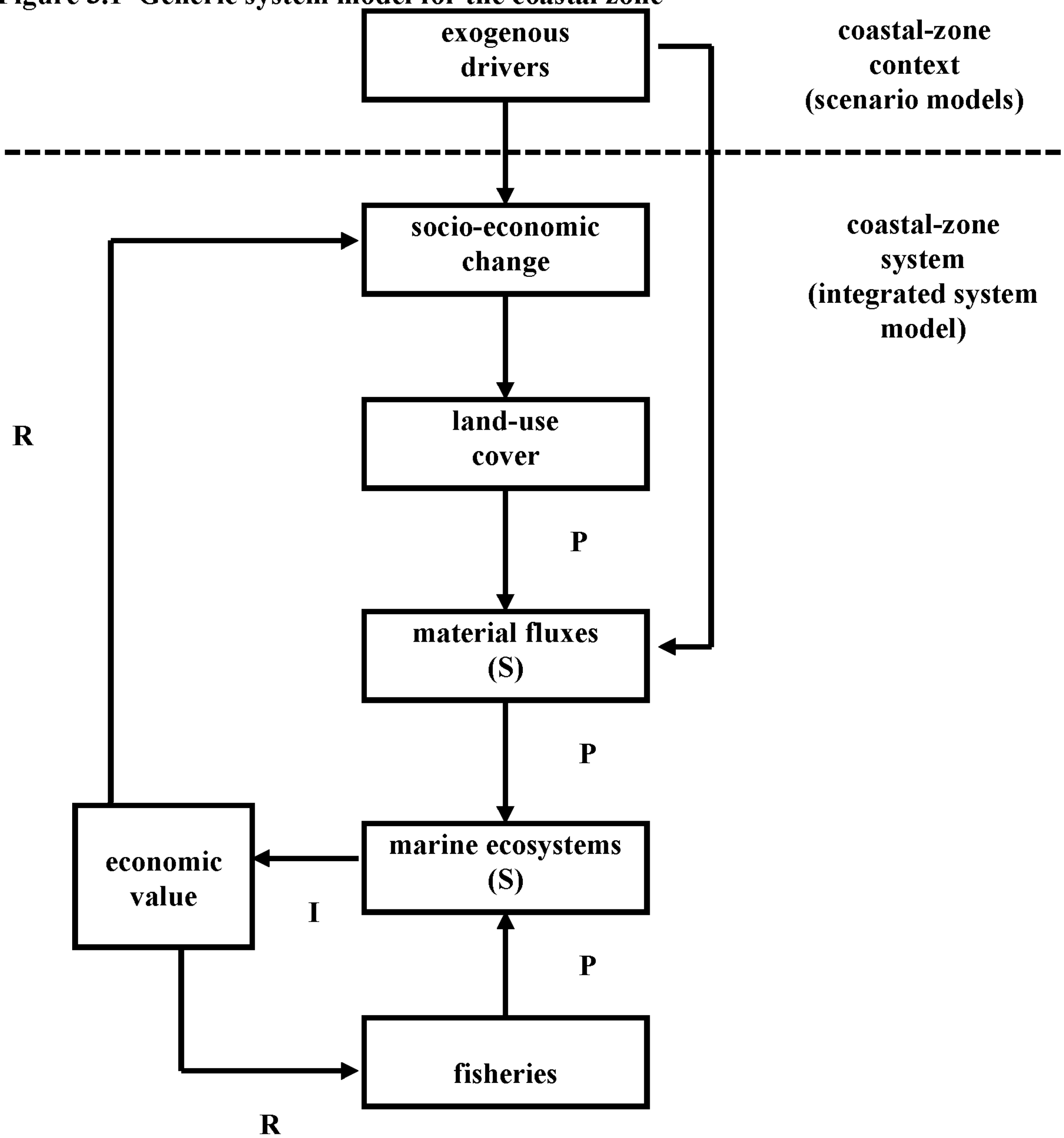
Figure 3.1 shows an example of a generic system model for the coastal zone. Reading the diagram from top to bottom we see how socio-economic and physical forces drive the coastal system. Examples of such external forcing mechanisms are demographic change, market demand and climatological conditions. Land-based economic activities such as industry, agriculture, or residential land use influence the fluxes of C, N, P, particulate matter, or toxics into the coastal waters. These in turn affect the functioning of marine ecosystems such as coral reefs and seagrass beds. Marine resource use forms such as fisheries can also directly affect ecosystems. For example, fish stocks may decline as a result of overexploitation whereas coral reefs can be damaged due to the application of destructive fishing methods. The pressures (P) exerted on the ecosystems cause a state change (S), with a possible loss of functioning. This may result in socio-economic impacts (I). In response, the driving mechanisms may change. For example, declining fish catches may result in increasing prices, which can lead to a change in the market demand for fish.

The LOICZ Implementation Plan (Pernetta and Milliman, 1995) has identified four general kinds of numerical modelling approaches (not necessarily discrete) that are of use to LOICZ research. They are budget models, process models, system models and prognostic models. The strategy suggested is to start modelling by preparing a simple mass balance budget for the variable(s) of interest. It may then be possible to move along the spectrum from budget models to the more complicated systems modelling if required and if the necessary resources and scientific capacity are available.

For the needs of LOICZ it is initially most important to get good estimates of the inputs and outputs of a coastal system than to capture the details of processes within the system. LOICZ has therefore begun to develop biogeochemical budgets which incorporate major physical oceanographic exchange and mixing processes. To make more progress initially, in 'integration' terms, the budget models require better socio-economic pressures/drivers data and analysis to assist their prognostic capabilities. Thus what is proposed is that by treating the budget (simple model of fluxes) as a first step in the modelling procedure rather than an end in itself, it should be possible to identify the major socio-economic drivers and system processes which determine the fluxes. LOICZ would then have started to make the important transition from a purely descriptive budget to a predictive process-based model. Ultimately the goal would be to move to numerical simulation models which focus on the internal dynamics of coastal systems and describe how critical biogeochemical processes are influenced by a whole range of anthropogenic and non-anthropogenic environmental variables. The biogeochemical guidelines document (LOICZ Report No. 5) lays out some more detailed procedures for developing a class of mass balance budgets "stoichiometrically linked water-salt-nutrient budgets" (Report No. 5, Section 5).

From the social science perspective, progress towards a more integrated assessment of coastal systems should incorporate three forms of models: **activity models, natural systems models** and **models with a valuation dimension**. The social science terminology has been used here but essentially what is being proposed as an analytical strategy is not incompatible with the modelling strategy adopted by LOICZ and other natural scientists. Thus activity models are the ways in which socio-economic drivers/pressures variables are related to C, N and P (among others) fluxes in drainage basin networks.

Figure 3.1 Generic system model for the coastal zone



Integrated system model for Land-Ocean Interaction. P = Pressure, S = State, I = Impact, R = Response.

They encompass residuals generation and modification activities (e.g. different agricultural cropping regimes and N releases; levels of sewage effluent treatment and consequent N and P releases) across all relevant socio-economic activities within a drainage basin. The IO modelling approach can be usefully applied up to perhaps the scale of a regional (within country) IO model, which could predict residuals generation (nutrients, but also sediments and other substances) for a geographical set of economic activities and population settlements, under a number of different economic growth scenarios.

Traditionally IO models are limited by their lack of dynamic properties (they are based on comparative statics i.e. a snapshot at current time T (and fixed coefficients) versus some defined future time point T + 1 with new fixed coefficients). Nevertheless, combined with change scenarios they could provide useful initial research findings. Much more complicated regional computable general equilibrium models offer increased flexibility but at the cost of much increased complexity and computing capacity and expertise (see SARCZ/WOTRO/LOICZ WORKSHOP Report No 20 p. 27-28 and Appendix 2 of this report for brief explanation of the application of IO models).

What social scientists have called natural systems models, other scientists would call budget models through to numerical systems models. However, at the prognostic systems model end of the spectrum there is a further 'interface' between natural and social science. In flux change contexts, for example, information on dose-response relationships would indicate what the impacts and implications for habitats, ecosystems and human welfare were because of changed C, N and P outputs.

In social science (and in terms of benefits to resource managers and policymakers) a prognostic model should have a specified and explicit objective function which relates to aspects of human welfare. The function will include ambient environmental quality indicators either in the function itself or as constraints. The policy goal of sustainable development of coastal resources is one such objective function in this context. Finally, because of the needs of policymaking (in which relative valuations of costs and benefits and trade-offs are inevitable) social science analysis is concerned with the development and application of criteria for evaluating strategies (see next sub-section (C)).

While the terminology might differ, the underlying approaches to activity and mass balance/budget models are entirely compatible and offer opportunities to initiate the integration exercise, and provide some prognostic capability. A number of different modelling shells have been used and both the STELLA and ECOS packages, for example, show useful initial results (see Merbok Mangrove case study in LOICZ Report No. 5 and LOICZ Meeting Report No. 20:SARCS/WOTRO/LOICZ Workshop on Integrated Modelling, 1996).

In order to offer sound management as well as science advice to managers and policymakers, evaluation analysis is required. What is proposed here is 'technical' information communicated to managers/agencies, not ready-made decisions or institutional management systems/approaches, which are not part of the LOICZ remit.

Summarising, the modelling phase should be initiated by simple nutrient budget models and then incrementally expanded to incorporate, where feasible and necessary, more complex systems models. In the absence of a suite of reliable models, LOICZ researchers need some pragmatic interim strategy in order to identify and weigh (magnitude and significance) the impacts and 'wider-system' implications of changes in fluxes across a range of geographical sites. Informed by the natural sciences, social scientists could then proceed with some impacts valuation studies even though the precise scientific 'cause and effect' mechanisms for the flux change impacts and implications may not have been fully quantified and modelled.

(c) Evaluation Stage

In any multiple resource use problem context, it will be necessary to identify the complete range of stakeholders present and their pressure impacts and influences. Multiple stakeholders have multiple worldviews and potential **values conflicts**. One way of conceptualising this values conflicts problem over time is via the formulation and analysis of **environmental change scenarios.** For this approach to produce meaningful results a **trend scenario** (i.e. the implications of current trends remaining substantially unaltered until some chosen terminal date in the future) needs to be contrasted with the results derived from one or more **alternative futures scenarios.**

The stakeholder/revenue conflict situations that may be identified in any given coastal zone could be assessed and evaluated via **multi-criteria** evaluation methods which encompass both monetary and non-monetary valuation procedures (see Figure 3.2).

Figure 3.2 Spectrum of appraisal methods

Financial Appraisal	Economic Appraisal	Multi-Criteria Approach
Based on private costs and benefits in cash flow terms.	Based on social costs and benefits, expressed in monetary terms, including environmental effects.	Based on non-monetary and monetary estimates of a diverse range of effects, social, political and environmental.
Analysis related to an individual economic agent, i.e. farmer, householder, firm or agency.	Social costs/benefits = private costs/benefits + external costs and benefits.	Scaling and weighting of impacts.
Typical techniques: discounted cash flows and balance sheets; payback periods and internal rates of return.	Typical techniques: cost-benefit analysis, extended cost-benefit analysis and risk-benefit analysis.	Typical techniques: impact matrices, planning balance sheets, concordance analysis, networks and trade-off analysis.

◄────────────────────────────────────►

less comprehensive/less data intensive more comprehensive/more data intensive

Environmental evaluation methods, showing increasing complexity and scale of analysis.

Financial Analysis	Economic Cost-Benefit Analysis	Extended Cost-Benefit Analysis	Environmental Impact Assessment	Multi-Criteria Decision Methods
financial profitability criterion; private costs and revenues; monetary valuation	economic efficiency criterion; social costs and benefits; monetary valuation	sustainable development principles; economic efficiency and equity trade-off; environmental standards as constraints; opportunity costs analysis	quantification of a diverse set of effects on a common scale, but no evaluation; or misleading composite index scores	multiple decision criteria; monetary and non-monetary evaluation combinations

Source: Pearce and Turner (1992)

Multi-criteria analysis offers opportunities to present the trade-offs and ranking of different priorities and criteria in a systematic manner which does not specify an overall single value framework, but which allows the sensitivity of both social and physical data to be tested for robustness, and which makes explicit the trade-offs between competing impacts and stakeholders.

The decision process has been well defined in management texts as having three separate stages: problem identification, developing possible courses of action, and selecting a course of action from the choices available (Janssen, 1994). This means that multi-criteria analysis must: effectively generate information on the decision problem from available data and ideas, effectively generate solutions (alternatives) to a decision problem, and provide a good understanding of the structure and content of a decision problem.

When issues such as social implications, ecological and environmental conservation or bio-physical impacts of decisions are also important to decision-makers, then multi-criteria analysis can be an appropriate analysis tool. Proponents of multi-criteria analysis believe it to be superior to economic cost-benefit analysis, outlined in the case study in Section 5.2, as it allows 'soft criteria' that cannot be expressed in monetary terms to be included in the analysis (van Huylenbroeck and Coppens, 1995). Multi-criteria analysis is particularly useful as an analysis tool in projects where there are conflicting objectives or priorities of different stakeholders. Another benefit of multi-criteria analysis is that it provides decision-makers with a set of feasible solutions, rather than one economically efficient outcome.

Multi-criteria analysis has been widely applied to land-use planning (Makowski *et al.*, 1996; Joubert *et al.*, 1997; Malczewski *et al.*, 1997). The lessons from research on applying multi-criteria analysis, where the aim is to achieve outcomes which are broadly acceptable to the relevant user groups, can be summarised thus: while multi-criteria analysis is a valuable tool for achieving resolution of environmental conflicts, there are several constraints to this in practice. Critical elements which must be clearly identified to enable participation in decision-making include: the relevant interest groups, the interactions between the interest groups, and the socio-economic activities undertaken by the interest groups.

The first step in a multi-criteria analysis is to define as far as is possible the actual problem, such as overuse of resources and degradation of the resources, ideally in discrete measures of the environmental impact, i.e. size of area involved, volume of each type of natural resource contained therein. A set of possible suitable alternatives (henceforth referred to as *scenarios*) for improving site quality are identified and compiled.

The model then requires that the predicted effects of each scenario be described. Before this can be done a set of objectives of management (henceforth referred to as *criteria*) must be developed. The criteria should reflect the different aspects socio-economic drivers for the relevant area, and ideally should be grouped into sub-headings that involve different user groups.

Each scenario is then measured, or valued, in terms of the list of criteria (potential effects). Due to differences in the methods and scales of measure of the different effects, there are likely to be variations in the accuracy of measurement. One way to standardise these generated measures of effect is to apply a *value function*, which converts the values into scores that range between 1 and 100 (Janssen, 1994). Some multi-criteria analysis computer software packages can perform this task automatically for the user.

To determine a rank ordering of alternative scenarios the relevant importance of the criteria must be distinguished. This can be achieved by *weighting* the criteria, both within each criterion (e.g. different types of economic impacts, net costs versus employment impact), and between criteria (e.g. economic impacts versus biological impacts). In many applications of multi-criteria analysis these weights are set by the analyst to reflect their judgement of the relative importance of the criteria, or are derived through the opinion of elite groups or experts, sometimes through a Delphi method.

Ultimately, application of the multi-criteria analysis should produce a best alternative scenario given weights determined by the decision-makers. This type of analysis is particularly useful where the criteria can be sub-grouped into two or three main criteria which offer conflicting solutions.

The appraisal of waste disposal options requiring a balancing of risks, costs benefits and their integration into the decision making process has in the past been analysed through the combination of CBA and multi-criteria analysis (Turner and Powell, 1993). In the waste disposal decision-making process, multi-criteria evaluation allows some insight into the relative importance of financial, resource and environmental considerations (social costs and benefits) (Maimone, 1985). The foreclosure of the North Sea sewage sludge disposal route, agreed at the 1987 North Sea Conference, was in accordance with the adoption of the precautionary approach but also served to highlight the possible drawbacks of such a strategy. The foreclosure decision involved significant social opportunity costs as land-based disposal operations will have to be utilised on a more extensive basis. It has been estimated that the ban could lead to additional capital expenditure in England and Wales of around £100 million and increased water company operating costs of £0.4 million per annum (WRc, 1990).

According to WRc (1990), in England and Wales approximately 1.22 x 10^6 tonnes dry solids of sewage sludge are generated annually by more than 6000 sewage treatment works. Half of all sludge is stabilised, principally by anaerobic digestion. In total, some 37% of the sludge is disposed of to agricultural land, 19% to landfill, 25% to sea and 6% to incineration. The sewage arising from about 13% of the population of England and Wales remains untreated and is discharged directly to the sea via outfall pipes. If all this sewage was subjected to treatment this would involve between £0.5 billion and £1.3 billion of capital expenditure and increased annual operating costs of between £15 million and £33 million, depending on the level of treatment that was installed.

If the sludge cannot be disposed of at sea then it must either be incinerated or deposited on land (via agricultural land or in landfills). Each of these alternative disposal options carry with them a set of environmental effects and related social costs and benefits. The economic cost-benefit approach would require that the net social benefits (expressed in monetary terms and discounted to present value) of the current disposal option be compared with the net social benefits generated by each of the feasible alternative options.

A preliminary look at these different costs and benefits indicates that for England and Wales no one option is clearly dominant. The three options need to be seen from a long-run perspective and with other background factors. Switching sewage sludge from the marine environment to land-based locations would generate a complicated set of social costs and benefits, many of which are difficult to evaluate. The influence of intervention failure (uncoordinated policies) from both current and future policy initiatives is also clearly apparent. It is also far from clear that the banning of sea disposal is likely to lead to the promotion of the 'best practicable environmental option'. Although pollution of coastal waters and delayed emissions of some CO_2 (due to the oxidation of organic material) to the atmosphere are risks associated with sea disposal, various official studies of the disposal grounds used by the UK in the North Sea indicate only minimal environmental impact (see MAFF Aquatic Environment Monitoring Report No. 20).

A recent UK multi-criteria analysis of sewage sludge disposal options investigated four feasible disposal routes:

a) sludge consolidation followed by incineration and landfill of residual ash;
b) sludge consolidation followed by soil injection;
c) sludge consolidation followed by anaerobic digestion with combined heat and power, mechanical de-watering and final surface spreading to agricultural land;
d) sludge consolidation followed by mechanical de-watering and landfill (WRc, 1990).

All the options were evaluated on the basis of three categories of criteria - discounted financial costs (over a 20-year period at a discount rate of 5%); operational security (a mix of operational and management risk factors plus longer-term general trend changes in land use policy, social acceptance etc.); and environmental impacts. On this basis the study identifies option a) incineration of sludge, as the preferred option - see Table 3.2.

Table 3.2 Multi-criteria analysis of sludge disposal options

Options	Financial Cost (discounted over 20 years)	Operational Security (rank order)	Environmental Impact
(a)	£16.9M (4)	(1)	(1)
(b)	£13.45M (1)	(2)	(2)
(c)	£15.7M (2)	(4)	(3)
(d)	£14.6M (3)	(3)	(4)

Source: WRc (1990)

If the majority of the sewage sludge currently disposed of to the marine environment was switched to incineration sites the main environmental implications would be:

i) a redistribution of heavy metals, inorganic material and possibly dioxins to landfill sites, agricultural land and to the atmosphere;
ii) a reduction in the direct transmission of dioxins, PCBs, pesticides, organics, and nutrients to the ambient environment;
iii) increased emissions of SO_2, HCl and NOx to the atmosphere;
iv) increased direct emission of CO_2 to the atmosphere;
v) small increase in volume of road traffic and related emissions.

Option b) agricultural land application was the second ranked option in the UK study but topped the ranking list in a recent US study (EPA, 1990) - see Table 3.3. In the US, land use pressures are relatively less intense than they are in the UK and landfill management has long been based on a 'concentrate and contain' basis. Nimbyism associated with incinerator facilities close to sewage works and population centres is as intense as, or perhaps more intense than in the UK. However, stocks of incinerators in both the USA and most of Europe are relatively larger and more modern than in the UK.

Table 3.3 US multi-criteria analysis of disposal options (Source: EPA, 1990)

Evaluation Factors	Incineration with Ash Landfilling		Land Application*	In-Vessel Composting	Drying & Product Use	Landfilling	Ocean Disposal
	6 units	4 units					
<u>Economic Analysis</u> (Total Equivalent Annual Costs)	$21,298,000	$19,053,000	$20,218,000	$28,735,000	$15,130,000	$24,704,000	$8,164,000
<u>Operability</u> (includes reliability, flexibility, and maintainability)	Moderate		Moderate	Moderate	Low	Low	Low
<u>Implementability</u> (includes public acceptability and management requirements)	High		High	Moderate	Moderate	Low	Low
<u>Potential Adverse Environmental Impacts</u>							
<u>Air Impacts</u>							
o Stack Emissions	x	x			x		
o Odor Emissions			x	x		x	
<u>Water Impacts</u>							
o Surface Water	x[1]	x[1]	x[2]			x	x
o Groundwater	x[1]	x[1]	x[2]			x	
<u>Land Impacts</u>							
o Transportation[4]	x	x	x	x	x	x	x
o Land Use Conflicts			x				
o Nutrients Overloading			x[3]	x	x	x	
o Landfilling Capacity	x	x					
o Aesthetics	x	x					

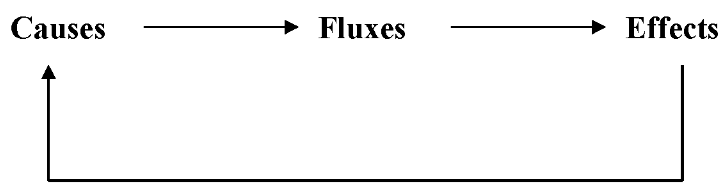
<u>Other Environmental Considerations</u>

[1] Potential impact at landfill; leachate generation from ash residue; [2] Impacts are possible but extremely low because of guidelines and regulatory controls; [3] Potential for nutrient overloadings are remote if state guidelines are followed; [4] Every alternative will require some type of hauling including ash from the incinerators; * Preferred option

3.2 Disaggregating the P-S-I-R Framework

Within the overall modelling framework provided by the P-S-I-R approach, LOICZ's central concern with the fluxes of nutrients, sediments and water across drainage networks and into coastal waters can be conceptualised as one set of components and their interrelationships among several sets that in aggregate represent the global environmental change process - see Figure 3.3.

Figure 3.3 General framework for integrated assessment[1]



[1] Parallels the P-S-I-R framework developed in the LOICZ Implementation Plan.

It is then possible to conceive of a suite of nested models related to different stages in the 'causes to fluxes and fluxes to effects' relationship - see Figures 3.4, 3.5 and 3.6.

In the 'causes to fluxes' relationship (i.e. pressure to state relationship) the drivers are land use and water use, as well as industrial development and population change, causing flux changes and hence changes in the environmental state (Figure 3.4). The social science modelling techniques at this stage are also shown in Table 3.8.

Figure 3.4 Drivers and modelling techniques inherent in the pressure to state relationship.

Cause → Flux relationship
Pressures → State relationship

Drivers	**Modelling technique**
Land use change	Food supply and demand models and nutrient flow models
Water regulation management	Physical run-off models, etc.
Industrial development	Input-output models, etc.
Population change	Housing supply and demand models, infrastructure impacts

Some of the drivers shown in Figure 3.4 have spatial elements while others do not. In the flux to impact relationship, the drivers and modelling techniques are outlined in Figure 3.5.

Figure 3.5 Drivers and modelling techniques inherent in the state to impacts relationship.

Flux → Impact relationship
State → Impact relationship

Drivers	**Modelling technique**
Changes to water quality/quantity	health impacts (dose/response models, health impact models); recreational demand models
Changes to nutrient loading and primary productivity	fisheries stock models coupled to fishing effort models
Changes to coastal geomorphology	recreational demand models physical risk and hazard assessments

Finally the impact to response (cause feedback) relationship is characterised by the drivers and modelling techniques outlined in Figure 3.6. These are primarily social science models (often normative or policy driven models). It is extremely difficult to control any of the drivers in the impact to cause feedback (response) relationship.

Figure 3.6 Drivers and modelling techniques inherent in the impact **to response feedback relationship.**

Impact → Cause feedback
Impact → Response relationship

Drivers	**Modelling technique**
Demand and taste changes	Demand for water quality, productive and recreational use of coastal resources
Policy process (political lobbying/decision-making)	Stakeholder assessments/participatory planning, political economy approaches
Economic constraints	Cost/benefit analysis

3.3 Pressure to State Sub-Models

This section provides an overview on building and running pressure to state models. Although space does not allow for all the details to be included, important considerations are discussed which will allow a choice to be made on the most appropriate model for the task on hand. Readers are referred to the relevant literature for further information wherever necessary. Armstrong (1978) provides a good introduction to modelling which the reader might refer to, and Appendix 2 provides some illustrative examples.

3.3.1 Drainage networks modelling

Billen *et al.* (1995) and Howarth *et al.* (1996) have completed calculations which show the impact of different degrees of climate and anthropogenic pressures in 14 defined regions within the North Atlantic catchment. They take drainage networks as the appropriate geographical scale (see Table 3.4) to scale up estimates of emissions of N to the global level.

Table 3.4 Area specific riverine export of N and P from the 14 regions considered within the North Atlantic catchment area (after Howarth *et al.*, 1996; Billen *et al.*, 1996)

Regions	**Population density inhab/km^2**	**Specific runoff (mm)**	**N export kgN/km^2/y**	**P export kgP/km^2/y**	**N/P ratio (molar)**
North Canadian rivers	3	316	76	4.5	37
St Lawrence basin	24	500	413	12	73
NE coast US	114	433	1070	139	17
SE coast US	44	168	676	32	47
Eastern Gulf of Mexico	66	303	675	32	46
Mississippi basin	20	170	566	33	38
Western Gulf of Mexico	31	200	601	5	271
Total North America	**22**	**286**	**404**	**21**	**42**
Caribbean Is. & Central America	33	908	476	62	17
Amazon & Tocantins	1.5	1080	505	236	4.8
Total Central & South America	**10**	**1034**	**498**	**190**	**5.8**
Baltic Sea	47	316	495	48	23
North Sea	186	452	1450	117	28
NW coast Europe	90	1111	1300	82	36
SW coast Europe	92	200	367	101	8.3
Total Western Europe	**95**	**415**	**805**	**78**	**23**
NW Africa	41	118	420	25	**38**
TOTAL	**29**	**524**	**486**	**83**	**13**

Other researchers have also calculated the effects of some economic activities on the nitrogen cycle of terrestrial systems - see Table 3.5.

Table 3.5 Area specific anthropogenic inputs of nitrogen to the 14 regions considered within the North Atlantic catchment area (after Howarth *et al.*, 1996).

all values in kgN/km^2/yr

Regions	**Anthropic atmospheric deposition**	**Fertiliser**	**Legumes N fixation**	**Net import food & feed**	**Total inputs**
North Canadian watersheds	72	161	33	-50	216
St Lawrence basin	542	331	256	-31	1100
NE coast US	826	600	748	998	3170
SE coast US	869	1170	369	454	2860
Eastern Gulf of Mexico	791	1260	248	576	2880
Mississippi basin	494	1840	1060	-1300	2090
Western Gulf of Mexico	234	1254	?	?	1490
Total North America	**356**	**878**	**397**	**-317**	**1310**
Caribbean Is & Central America	196	342	?	?	538
Amazon & Tocantins	139	63	?	?	202
Total Central & South America	**154**	**136**	**?**	**?**	**289**
Baltic Sea	451	1730	27	21	2230
North Sea	763	5960	5	-4	6720
NW coast Europe	765	2870	55	-324	3370
SW coast Europe	322	3370	15	-64	3640
Total Western Europe	**544**	**3230**	**22**	**-36**	**3760**
NW Africa	?				

Sources of anthropogenic nitrogen from terrestrial systems to drainage networks include leaching of agricultural and non-agricultural soils and direct discharge of sewage - see Table 3.6. The presence of lakes and dams/reservoirs also plays a vital part in controlling the N budget of whole river systems.

Table 3.6 Input of nitrogen to surface water by leaching of agricultural soils in the temperate watershed regions of the North Atlantic

Region	% forest	% crop land	% grass land	Fertiliser application kgN/ha	N conc cropland mgN/l (1)	N conc grassland mgN/l (1)	Runoff l/m²/yr	N emission TgN/yr
Baltic Sea	90	9	1	175	25	10	320	1.11
North Sea	38	33	29	96	10	2	450	1.45
NW Europe	84	6	10	185	18	10	1110	0.76
SW Europe	35	40	25	52	5	2	200	0.26
Total Europe		**20**	**13**	**97**			**420**	**3.58**
N. Canada	90	-	-				320	-
St Lawrence	77	20	3	14	4	1	500	0.66
NE US coast	85	11	4	40	6	1.5	430	0.15
SE US coast	81	13	6	62	7	2	170	0.06
E Gulf of Mexico	80	9	6	62	7	2	300	0.09
Mississippi	38	32	30	29	5	1	170	1.04
W Gulf of Mexico	48	8	44	24	5	1	200	0.24
Total N America		**14**	**15**	**30**			**290**	**2.24**

Direct point inputs of sewage into rivers can be calculated from population figures and data on the percentage of the population in each region that is sewered (OECD, 1991; WRI/UNEP 1988). Following Meybeck *et al* (1989) quoted in Billen *et al* (1995), a *per capita* nitrogen load in sewage of 3.3 KgN/inh/yr (9gN/inh/day) was assumed - see Table 3.7.

Table 3.7 Nitrogen input to the North Atlantic from sewage (after Howarth *et al.*, 1996).

Regions	Sewered population 10^3 inhab	Sewage input Tg/yr
North Canadian watersheds	6 670	0.022
St Lawrence basin	21 820	0.072
NE coast US	39 390	0.13
SE coast US	10 300	0.034
Eastern Gulf of Mexico	14 850	0.049
Mississippi basin	48 480	0.16
Western Gulf of Mexico	24 850	0.082
Total North America	**166 360**	**0.55**
Caribbean Is & Central America	42 420	0.140
Amazon & Tocantins	90	0.0003
Total Central & South America	**42 510**	**0.14**
Baltic Sea	30 300	0.10
North Sea	127 270	0.42
NW European coast	22 120	0.073
SW European coast	14 240	0.047
Total Western Europe	**193 930**	**0.64**
NW Africa	29 090	0.096
TOTAL	**431 890**	**1.43**

These external forcing estimates have been included as illustrative of the procedures LOICZ might expand on. The Billen *et al.* (1995) and Howarth *et al.* (1996) work has also highlighted the fact that there are profound differences among local situations represented in the 14 watershed regions that were considered. Some comparative analysis has been undertaken which may link into the typology exercise that LOICZ is undertaking. Thus the Amazon and Tocantins area is characterised by the lowest anthropocentric inputs of nitrogen. Because of the normal functioning of tropical rainforest and for other reasons nitrogen fluxes from this region actually exceed anthropogenic inputs of nitrogen. The North Canadian region seems to be closest to what could have been the pristine nitrogen cycle in temperate regions. External anthropogenic nitrogen inputs and specific riverine delivery both represent less than one fifth of the mean value for overall North Atlantic when expressed on a per area basis. The North Sea watershed region however displays the most perturbed situation, with a largely open nitrogen cycle. Fertiliser inputs (6000 kgN/km^2/yr) dominate external inputs and represent more than one third of nitrogen uptake by the vegetation, leading to a strong nitrogen soil leaching effect. The high population density results in a significant sewage effluent discharge to surface waters.

The Mississippi Basin is an intensive agricultural region, with a moderate population density. Fertiliser inputs and nitrogen fixation by crop vegetation dominate the inputs, but as much as one third of these is exported as food and feed to other regions. The NE Coast US region is characterised by a high population density and limited agriculture. Imports of feed and food and anthropogenic nitrogen atmospheric deposition represent the two major external nitrogen sources. The nitrogen delivery to the coastal sea is nearly twice that of the Mississippi basin.

3.4 State to Impact and Response Models

Because policy decisions are required relating to a range of spatial and temporal scales and different socio-economic and political levels, several broad assessment categories need to be distinguished (Barbier, 1993). A given change in nutrient flux and land-use changes impose a particular impact on an individual coastal resource or set of resources, e.g. due to discharge from an industrial plant, oil spillage from platforms, storage facilities or during transport, sewage disposal from urban areas. Thus in this **impact analysis** category, a specific environmental impact is assessed via the valuation of the environmental state changes in the coastal resource(s) connected to the impact. The valuation requires an estimate of the consequent net coastal resources production and environmental benefits effects. The total cost of the impact (P_c) in social welfare terms is the foregone net benefits (NB_{fe}); so $P_c = NB_{fe}$.

The foregone net environmental benefits related to a pollution impact, for example, can then be compared with a range of alternative pollution abatement options and their cost (e.g. product and process design modifications for waste minimisation, end-of-pipe treatment and 'safe' disposal etc.). Table 3.8 summarises some relevant environmental state changes and related economic valuation

Table 3.8 Coastal environmental impacts and valuation methods (Adapted from Turner and Adger, 1996).

Effects Categories	Valuation Method Options
PRODUCTIVITY:	
e.g. Fisheries, agriculture, tourism, water resources, industrial production, marine transport, storm buffering and coastal protection.	Market valuation via prices or surrogates Preventive expenditure Replacement cost/shadow projects/cost-effectiveness analysis Defensive expenditure
HEALTH	
	Human capital or cost of illness Contingent valuation Preventive expenditure Defensive expenditure
AMENITY	
Coastal ecosystems, wetlands, dunes, beaches, etc., and some landscapes, including cultural assets and structures.	Contingent valuation/ranking Travel cost Hedonic property method
EXISTENCE VALUES	
Ecosystems; cultural assets	Contingent valuation

A second assessment category, **partial valuation**, encompasses situations which require the evaluation of alternative resource allocations or project options. A planned large scale project (or extension of an existing project) such as a residential/recreation housing complex, or port and harbour facilities, might require the conversion of coastal wetlands and mudflats with significant biodiversity and other functional values. The net benefits (NB_c) of the wetland conversion then would be the direct benefits of the project (B_D), minus the direct costs of the project (C_D = capital and operating costs), minus the foregone net production and environmental benefits of the conserved wetland:

$$NB_C = B_D - C_D - NB_{fe} > O.$$

In some cases the estimation of only some elements of the valuation expression above is necessary to prove that the development project is uneconomical, provided that the on-going utilisation of the natural system is at a sustainable level. An analysis of the **opportunity cost of wetland conservation** (i.e. foregone project direct net benefits), for example, might show that $B_D - C_D$ is only marginally positive (some past agricultural conversion schemes have actually been shown to be negative).

As long as the conserved wetland yields a flow of functional benefits e.g. storm buffering capacity, fish and other product outputs, the positive valuation of only some of these outputs and services will be enough to tip the economic balance against the large-scale project.

On the other hand, the development project may generate significant employment and regional income benefits and be seen as part of a regional development policy strategy. Increasing employment and reducing regional income disparities may therefore be interpreted as pre-emptive constraints on the cost-benefit analysis and such benefits may be heavily weighted by policy makers.

A third assessment category covers the evaluation of protected areas schemes involving restricted or controlled resource use. Such marine park or coastal nature reserve schemes, for example, might be a required compensating shadow project element in the approval process for a large-scale project; or alternatively might preclude the existence of a given project altogether. The precise circumstances will depend on how 'weakly' or 'strongly' sustainability standards/constraints are interpreted and imposed by planning/management authorities. The on-going loss of coastal wetlands might have reached such a stage that regulatory authorities were seeking to impose a "no net wetland loss" rule on all future development activity in the coastal zone (a pre-emptive environmental policy constraint on Cost Benefit Analysis - CBA).

In situations where there is a direct choice between a development project and a marine park or similar conservation scheme, or where compensating environmental shadow project possibilities are not available, it may be necessary to use the **total valuation approach** (Figure 3.7). The analysis would seek to determine whether the total net benefits of the protected area kept in a sustainable 'natural' state (Nb_p) exceeded the direct costs (C_p) of establishing the protected zone and necessary buffer zone, plus the net benefits foregone (NB_{fd}) of alternative development uses of the protected area. The conservation zone plus buffer zone set-up costs may include costs of relocating or compensating existing users:

$$NB_p - C_p - NB_{fd} > O$$

More detailed information on resource valuation methods, techniques and literature can be found in Turner and Adger (1996), LOICZ Studies Report No. 4 and in Appendix 3 of this report.

Figure 3.7 Methods for valuing coastal zone benefits.

Mitigation Benefits
Enhancement Benefits
Preservation Benefits
Option Benefits
Existence Benefits

Total Use Values (TUV)

Direct Use Values

e.g. fish
recreation
transport/
navigation

Valuation Methods
Market analysis:
Other non-market
valuation methods
(TCM, CVM, HP)
(IOC); (IS)

Indirect Use Values
(Functional Values)

e.g. flood control
storm protection
nutrient cycling
waste assimilation
sedimentation
habitat loss reduction

Valuation Methods
Damage costs avoided
Preventive expenditures;
Value of changes in
productivity;
(Relocation costs)
(Replacement costs)

Option Values

insurance value
of preserving
options for use

Quasi-option

value of
increased
information in
the future
"natural
laboratory"

Valuation Methods
CVM

Total Nonuse Values (TNUV)

Existence Bequest

- value derived from just knowing a species or system is conserved
- value of passing on natural assets 'intact' to future generations
- 'moral resource' value motivations

Valuation Methods
CVM

Notes: **Market Analysis**: based on market prices; **HP =** hedonic pricing, based on land/property value data; **CVM** = contingent valuation method based on social surveys designed to elicit willingness to pay values; **TCM** = travel cost method, based on recreationalist expenditure data; **IOC** = indirect opportunity cost approach, based on options foregone; **IS** = indirect substitute approach.
The benefits categories illustrated do not include the **"indirect"** or **"secondary benefits"** provided by the coastal zone to the regional economy, i.e. the regional income multiplier effects.
Source: Adapted from Turner (1988), Barbier (1989) and Bower and Turner (1998)

4 SCALING UP PROCEDURES AND PROBLEMS

4.1 Limits on Scaling Relative Economic Values

The first issue in scaling up the 'state to impacts' relationship concerns scaling economic impacts, which may be quantified in monetary terms, to other sites and to other scales. The underlying issue here is that all values are relative, reflecting relative resource scarcity at one point in space and time. Thus estimation of the use values and non-use values of one hectare of mangrove are specific to the site, not only in terms of the physical characteristics of the site, but also in relation to its location with respect to markets, demand for its services and many other factors unrelated to the site. Thus that value is not straightforwardly transferable.

If a single environmental function of an ecosystem is converted into monetary value, and that function can be extrapolated and its value is *independent of location,* then scaling up economic values can be partially valid. The carbon sequestration function of coastal or terrestrial ecosystems, for example, has equal social value wherever it occurs. This social value of the function is related to the postponement of the impacts of increased atmospheric concentrations of carbon, namely the future impacts of climate changes. Brown and Pearce (1994) for example, summarise and characterise the carbon storage and fluxes associated with tropical forest ecosystems and their conversion to other uses. On the basis of this information, via the application of an economic value of carbon associated with climate change impacts avoided, they derive a generic value of the carbon sequestration function of tropical forests. This results in an estimate of the value of the function that is in the order of several hundred dollars per hectare. This value, however controversial the valuation technique, is potentially widely applicable because of the non-site specific nature of the function (carbon storage). But even in this case the value is contingent on current attitudes to risk and risk aversion, as well as prevailing time preferences in society. The spatial scaling of the carbon sequestration and other values for tropical forests using GIS has been demonstrated, for example, by Eade and Moran (1996), and in the context of total economic value (TEV) of tropical forests by Adger *et al.* (1995).

By contrast, the extrapolation of site and demand specific use values of tropical forests cannot be validly scaled up across large areas. Peters *et al.* (1989) estimated that the direct use value of sustainably harvested timber and non-timber products from sample forest plots was of the order of $US600 per ha, and hence greater than the economic returns from conversion to cattle ranching. But this result, although appealing, does not hold for each hectare of Amazonian rainforest because the 'value' in market terms is not the potential revenue but the potential revenue less the costs of extraction, so the value depends on demand factors as well as extraction costs such as distance to markets. Some studies have shown the overall 'value' of Amazonia's forests by extrapolating Peters *et al.*'s (1989) value across the basin (Pearce, 1991; Southgate and Clark, 1993), but this scaling is illustrative only and not valid in strict economic terms.

The position adopted by LOICZ is that, while there are limits to the economic calculus i.e. not everything is amenable to meaningful monetary valuation, economic valuation methods and techniques can and should play a significant role in the project, programme and policy appraisal process which leads to the setting of relative values (including environmental assets values). Costanza *et al.* (1997) controversially estimated the current economic value of seventeen ecosystem services on the biosphere-wide basis at between $US16-54 trillion ($10^{12}$) per year, giving an average annual value some 1.8 times the current global Gross National Product. Table 4.1 summarises the functional values for two typical coastal zone assets, coral reefs and mangroves.

Table 4.1 Composition of value elements for selected ecosystems

Coral Reefs $US per hectare per year		**Mangroves** $US per hectare per year	
coastal protection	2750	coastal protection	1839
waste treatment	58	nutrient cycling	6696
food production/ biological control	259	food production/ biological control	797
recreation	3008	recreation	658
TOTAL	**6075**		**9990**

Source: derived from Costanza *et al.,* 1997.

The rationale behind this valuation exercise could be based on a number or arguments:

- Due to a lack of adequate market price data (or absence of data), together with inadequate (or absent) property rights regimes which ensure that resource values can be practicably appropriated, ecosystem services are assigned too little or zero value and weight in policy decisions;

- Some important environmental science research and debate, together with related policymaking (i.e. international agreements and conventions) necessarily takes place at the global scale. There is a need therefore for social science research to 'engage' science and policy at this scale. But such an engagement must, in our view, encompass analysis which will show clearly why globally aggregated social science estimates are often not meaningful, if the objective is to move beyond mere dialogue towards a more rational policy process;

- It is important to prove how valuable ecosystem services really are and to formulate mechanisms by which such function-based values can be realistically captured. Such capture must be relevant for everyday socio-economic and political activity and decision taking, through national income and resource accounting and project cost-benefit appraisal, down to the grass roots level in developed and developing countries.

Costanza *et al.*'s study goes some of the way towards meeting the arguments set out above. Their paper has engaged environmental scientists and policy makers, but the global, biome scale, economic value calculations risk ridicule from both scientists and economists. On the basis of the data and methods cited in the article and supporting inventory, the conclusion that the value of the biosphere services really is around $US33 trillion per year, is not supportable. Apart from raising policymaker, scientist and citizen awareness about the environment's economic value and the possible significance of the loss of that value over time, the global value calculations do not serve to advance meaningful policy debate in efficiency and equity terms, in practical conservation versus development contexts. Such calculations with their 'single number' outcomes shroud a number of fundamental 'scaling' problems to do with valuation contexts i.e. the temporal, spatial and cultural specificity of economic value estimates. Such values can also only meaningfully be assigned to relatively small ('marginal') changes in ecosystem capabilities (functions/services). The practical problem is that determining precisely what is and what is not a discrete and marginal change in complex ecological systems is not straightforward.

The issues of relative scarcity and basis of value are generic and serve to constrain the transfer of site-based function and system services economic values across time and geographical and cultural space. It is not being argued that all such benefits transfer is invalid, but we do believe that such procedures must be handled with extreme caution and have real limits. Many value estimates will not be amenable to legitimate aggregation beyond local to 'regional' (defined biogeographically and including trans-national boundaries where necessary) scales. Further research to more precisely define these limits and to formulate a robust validity and reliability testing protocol is an urgent requirement.

At the core of the Costanza *et al.* (1997) valuation approach is a conceptual model which links ecosystem processes and functions with outputs of goods and services, which can then be assigned monetary economic values, see section 2.2. In principle, both economic use and non-use value estimates can be derived from sustainable or 'healthy' ecosystems. This model does provide a sound basis for future multi-disciplinary research on ecosystem services valuation expressed as ranges not point estimates. But the function-based approach must be undertaken on the basis of procedural rules which ensure scientific and economic validity and reliability. Its validity is conditioned by the existence of full knowledge about the relevant ecosystem structure, as well as temporal and spatial scale effects. Thus the raw empirical data inventory on environmental values utilised by Costanza *et al.* (1997), is not amenable to simple translation and aggregation, see Table 4.2.

The foundation of economic valuation based on a functional approach is an appreciation of the links between ecosystem structure, the characteristics of which provide society directly with extractive and non-extractive benefits (goods and services) and ecosystem processes providing indirect benefits.

Table 4.2 Aggregation and scaling problems

	Ecological Systems Perspectives	Economics Perspective	Ethical Perspective
STAGE I Identification, estimation and aggregation of individual function service value estimates; and compilation of ecosystem values	Systems behaviour is complex and characterised by interrelationships and feedback effects, not all of which are obviously related to human welfare concerns on the basis of existing science; total system value > total economic value.	Stock and flow concepts need to be distinguished; value estimates are not all strictly welfare changes and are not additive; other double counting mistakes need to be avoided.	Non-anthropocentric intrinsic value in nature can exist and is not commensurate with the other value dimensions.
STAGE II Individual function value estimates and/or ecosystem value estimates scaled up to global level	Area basis of world's ecosystems not the appropriate unit-e.g. overlap of ecosystem classification	Limits to scaling up (on a per hectare basis) in terms of temporal, spatial and cultural specificity of value estimates.	

Source: Turner *et al.* (1998)

This appreciation of the system's underlying dynamics is essential in order to sort out the stocks and flows involved and to ensure against possible double counting. Both stock and flow value estimates have, for example, been calculated and reported in forest services valuation studies, and researchers seeking to obtain aggregate figures have incorrectly summed both types of numbers.

Aggregation across different functions provided by a given ecosystem should be constrained by the danger of double counting. Exploitation of one function may preclude another, while some keystone processes and related functions may underpin others and the system infrastructure. There may also be possible incompatibilities between different valuation measures (such as opportunity costs, consumer surplus, market prices) as they are applied to different functions. Within a wetland, the exploitation of a particular function service such as wastewater cycling, for example, will preclude, or limit, the provision of other services, such as recreation. The global value of mangroves cited in Costanza *et al.* (1997) of $US9990 per ha, for example, is in fact made up of substitution cost estimates for the coastal protection function, replacement cost estimates for the nutrient cycling service, market price value of food production and a travel cost estimate of the recreation service.

Scaling up values of single functions of an ecosystem, or even more ambitiously, aggregate systems value to biogeographical and global scales, on a per hectare basis, increases the difficulties by an order of magnitude. If a single ecosystem service, in economic welfare terms, is relatively independent of location and its socio-cultural context, then the scaling up procedure can be regarded as more or less valid. Thus the economic value of carbon sequestration provided by coastal or terrestrial ecosystems has equal global social value (postponement of possible global climate change and its impacts) wherever it is provided. Or, in the case of coral reefs, the recreation value of reef visits and diving may be conditioned by a reasonably common set of site characteristics, regardless of precise location, and excluding totally inaccessible sites.

On the other hand, the values of many services are primarily determined by the locational factors. The recreational value of mangroves, for example is based on a very small number of published studies. A study in Trinidad and Tobago used the zonal travel cost method to estimate the recreational use value of Caroni Swamp. This site is of national significance as the only nesting site for the national emblem bird, the scarlet ibis. The large visitation rate at this site can be explained by this symbolic significance value and by its proximity to Port of Spain, the island's largest population centre and magnet for cruise ship visitors. Clearly, similar site and demand characteristics are not present in all other mangrove forest locations and therefore the scaling up value for more sites is very problematic (see Adger, 1997). Although some criteria for 'benefits transfer' have been discussed in the literature a comprehensive testing protocol is not in prospect. More research is required on these scaling possibilities and limits. It seems likely that the biogeographical regional scale will be the limit for a large number of service values.

Finally from a systems perspective, the aggregation and scaling up of individual ecosystem services value estimates does not lead to the quantification of a total ecosystem value. An evolving 'healthy' ecosystem presumably requires some, currently unknown, minimum configuration of structure, processes and functions to retain its resilience property. Economic valuation studies estimate the value of the flow of services assuming the sustainability of ecosystems and their interrelationships between themselves and the abiotic environment. The policy implications are that there is a role for the precautionary principle and the safe minimum standards notion (combined with social opportunity cost assessments).

4.2 Transboundary issues and scaling issues

A further issue in scaling up the economic impact and response aspects of integrated analysis arises in transboundary contexts (in the sense of across national boundaries). The modelling of policy options and response mechanisms is fundamentally different when considering a single regulator of coastal resources, compared to more than one national government. Thus if the normative technique of cost-benefit analysis is used to demonstrate the optimal policy response to the impacts of environmental change in the coastal zone (defined in economic terms), this assumes that there is a single overarching regulating institution and a single decision to be taken.

But reference to real world examples of coastal resource use, from water extraction to fisheries, demonstrates that unsustainable use is often associated with non-co-operative behaviour between different actors, for example between neighbouring countries. In such cases, each country or stakeholder group must be considered separately in terms of its different demands for scarce resources, affected by numerous factors such as income levels, cultural preferences and spatial distribution of population. In addition, the preference function of different stakeholder groups are directly impacted by their perceptions of the other stakeholder groups, for example observed in their willingness to cooperate in coordinated action.

An example of the transboundary approach is shown in the Baltic Sea case study (section 5.5) where the impacts of eutrophication of the Baltic and the benefits from 'response' are differentiated between countries. To determine whether coordinated action and the realisation of the gains from trade will take place (not undertaken in the case study presented) requires a fundamentally different model from that of observation of the policy responses of a coastal resource use issue within one administrative boundary.

5. CASE STUDY EXAMPLES

5.1 Introduction

The empirical case studies have been selected and ordered in terms of their spatial coverage and degree of functional complexity. They range from interdisciplinary analysis which seeks to quantify and evaluate individual ecosystem functions, through to whole drainage basins and coastal waters and their management. The case studies also serve to illustrate one or several steps in the P-S-I-R framework and the whole cycle, in the case of Tokyo Bay and the Baltic basin. Those two studies include the use of scenario simulation and analysis.

5.2 Impact and Response Evaluation through Cost-Benefit Analysis: Mangrove Planting in Coastal Vietnam

Introduction

This case study calculates the multiple economic costs and benefits of mangrove rehabilitation within a cost benefit framework using data and examples from Vietnam. It therefore provides an example of the state-to-impact analysis: it directly quantifies changes in environmental 'state' in terms of their impact on human well-being, or 'impact' referred to in the P-S-I-R framework. It further uses this information for two purposes: to appraise the economic desirability of rehabilitation (hence moving into the 'state-to-response' part of the framework) and to make recommendations on the cost-effectiveness of mangrove rehabilitation at this national scale.

The examples of mangrove rehabilitation reported here are distilled from research produced under University of East Anglia and Vietnam National University collaboration (funded by the UK Economic and Social Research Council) (Tri *et al.*, 1998; Adger *et al.*, 1997) and from research developed under the SARCS/WOTRO/LOICZ programme in South East Asia (e.g. Tri *et al.*, 1997). The results of the analysis show that mangrove rehabilitation can be desirable from an economic perspective based solely on the direct use benefits of local communities even when multiple objectives are present. The schemes have higher benefit/cost ratios with the inclusion of indirect benefits related to the avoided maintenance cost for the sea dike system which the mangrove stands protect from coastal storm surges. The following sections argue that impact analysis can be carried out through standard economic tools, and show how these calculations can be made.

Valuation and Cost Benefit Analysis

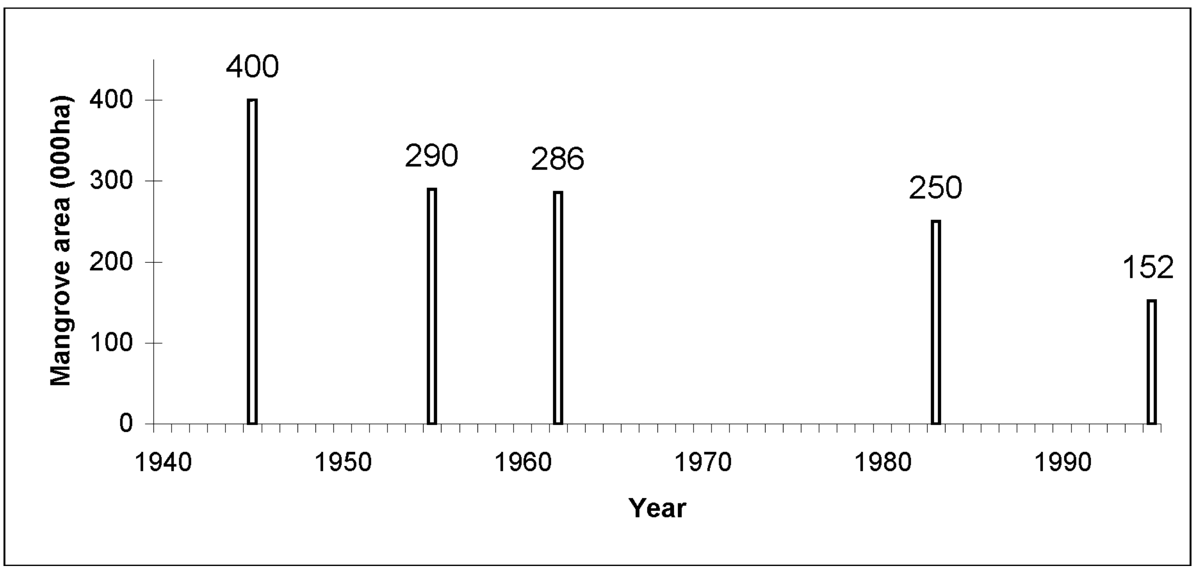
Carrying out an analysis of the impacts of resource management decisions requires a clear distinction and delineation of the limits of appraisal. Such appraisal often has economic dimensions. Where all factors, both economic and non-economic, are brought into an economic framework, this is known as extended cost benefit analysis, a standard economic tool which has been applied to many public sector decisions (see Pearce, 1983; Brent, 1996). The particular issues to do with human welfare changes and their valuation in situations of environmental change have been discussed in Section 4 (see also Pearce, 1993; Hanley and Spash, 1993; Turner and Adger, 1996). The range of available valuation methods is further outlined in Appendix 4 of this report. This section uses cost benefit analysis as an example of how impacts are identified and appraised and response options are determined.

In general the steps within the analysis (see for example those outlined in Hanley and Spash, 1993) are:

Step 1 Definition of proposed environmental impact or change in resource utilisation
Step 2 Delineation of the relevant costs and benefits
Step 3 Quantification of the environmental and physical impacts of the proposed or observed change
Step 4 Valuation of the relevant effects
Step 5 Discounting of the temporal cost and benefit flows
Step 6 Deciding on the desirability of the proposed change and policy recommendations
Step 7 Sensitivity analysis and scaling activities.

The first step in this appraisal framework is therefore to delineate the resource issue of potential environmental change to be examined. In this case study there is a need to examine the resource efficiency of using land, labour and capital inputs to rehabilitate or restore mangroves in the coastal areas of Vietnam. The total mangrove area of Vietnam has been in decline during the second half of this century, according to contemporary historical estimates of this total area (see Figure 5.1). The restoration of mangroves has therefore been undertaken along the coast of Vietnam where mangroves previously have been converted to agricultural or other uses, or have been degraded because of herbicides used during wartime.

Figure 5.1 Total mangrove area in Vietnam 1945-1995



Source: Tri *et al.* (1998, in prep.)

The various functions and services provided by mangrove areas in general have been documented and appraised by ecologists (e.g. Lugo and Snedaker, 1974; Mitsch and Gosselink, 1993). It has also been recognised in economic analysis that the functions and services provided by mangroves, and wetlands in general, have positive economic value and that these are often ignored in the ongoing process of mangrove conversion (e.g. Barbier, 1993; Farber and Costanza, 1987; Ruitenbeek, 1994; Swallow, 1994; Barbier and Strand, 1998). Mangrove wetlands display the features of public good in that their use is non-exclusive, and they are converted to other uses because these functions are undervalued. Identification of the functions and services (see Section 3.7), the incorporation of these values into policy and the encouragement of appropriate property rights are therefore necessary steps in promoting sustainable utilisation of such resources.

The second step in this appraisal is the identification of costs and benefits, in this case the environmental costs of conversion of mangroves, or their rehabilitation. The allocation of effects into costs and benefits involves determining what is the current situation, and focusing, using partial analysis, on the values of the marginal changes. When the issue to be investigated has been identified (conversion of mangroves, or rehabilitation of mangroves), the costs and benefits, which occur at different times, are assessed together. The development action is considered to be desirable from an economic perspective if its Net Present Value (NPV) is greater than zero:

$$NPV = \sum_t \frac{B_t - C_t}{(1+r)^t}$$

where B_t is benefit at time t, C is cost at time t, and r is the discount rate.

A partial valuation has been undertaken for restoring mangrove forests, in areas where they have been lost in coastal Vietnam. The crucial aspects of value for local decision-making, and for the differential impacts of global change, are the direct and indirect use benefits. Option and existence values may also exist for mangroves (see Section 3.4 and Turner and Adger, 1996 for explanation of these terms) but often accrue at the global scale to those not associated with management decisions. It should be noted that some economic benefits of the mangrove resource will increase in value over time, while others will remain constant or decline. For example, as agricultural development intensifies, the potential for economic loss from storm surges increases, so the value of the coastal protection function of the mangroves will rise accordingly. In addition, exogenous environmental change associated with global climate change may increase the frequency and intensity of storm surges, and hence the value of the mangrove protection function will rise in these circumstances.

The economic cost benefit analysis of mangrove rehabilitation schemes in this case is of the form:

$$NPV = \sum_{t=1}^{\gamma} \frac{B_t^T + B_t^{NT} + B_t^P - C_t}{(1+r)^t}$$

where

NPV	=	net present value (VND per ha)
B_t^T	=	net value of the timber products in year t (VND per ha)
B_t^{NT}	=	net value of the non-timber products in year t (VND per ha)
B_t^P	=	value of the protection of the sea defences in year t (VND per ha)
C_t	=	costs of planting, maintenance and thinning of mangrove stand in year t (VND per ha)
r	=	rate of discount
γ	=	time horizon (20 year rotation).

Estimates of the data sources and methods for carrying out the quantification and valuation (steps 3 and 4) of costs and benefits in establishing the rehabilitated mangrove stands are presented in Table 5.1. These estimate costs of establishment primarily based on the cost of labour for the activities described. The survey research to determine these costs estimates the cost for a work day in 1994 being typically 2.5 kg of rice or VND5,500. The estimates are averaged across three districts, with variations in costs dependent on where the propagules and seedlings were obtained. The planting and handling fees for seedlings obtained from forests in the area under rehabilitation are not significant compared to costs for collecting, handling and transportation for other areas. These costs increase depending on the distance from the seedling source site to the planting site.

For some mangrove species, such as *Sonneratia* spp., *Avicennia* spp., *Aegiceras* spp. and others, planting directly onto mud flats is unsuccessful due to the exposure to strong wind and wave forces which wash away the seedlings. The cost of raising such species in a nursery and transplanting them at eight months old is relatively high, with fees for maintaining the nursery, care, protection and transportation adding to overall expenditure. The costs of establishing a stand, including planting, gapping and protection, occur mainly in the first year. Maintenance, from the second year on, incurs

Table 5.1 Benefits and costs of mangrove rehabilitation in Vietnam and their valuation.

Impact or asset valued	Method and assumptions for valuation	Timing of costs and benefits
Benefits		
Timber benefits	Market data: Thinning (VND180 per tree); extraction of mature trees (VND 5000).	Thinning and extraction from year 6 with 3-year rotation
Fish	Market data: Mean price VND12,500 per kg; yield 50 kg per ha.	Fishing benefits from year 2 after planting
Honey	Market data: Potential yield estimated at 0.21 kg per ha.	Honey collected from year 5 after planting
Sea dike maintenance costs avoided	Morphological model: Costs avoided = f (stand width, age, mean wavelength).	Benefits rising from year 1.
Costs		
Planting, capital and recurrent costs	Market and labour allocation data: Costs of seedlings and capital (VND 440,000 per ha); Workdays valued at local wage in rice equivalent (VND 5,500 per day).	Planting costs at year 1; thinning from year 6 on 3-year rotation

Note: $US1 = VND11,000.

an estimated annual expenditure of VND82,500 per hectare. The cost of thinning occurs in years 6, 9, 12, 15 and harvesting is in year 20. This is shown in Table 5.2 where the next stage of calculating the discounted net present value are presented.

The direct benefits of rehabilitation include the value of the timber, as it becomes available through the first rotation of the mangrove stands; the other locally used products within the stands, including shellfish and crabs, and honey from bee-keeping. The major indirect benefit, and the principal reason for planting the stands, is the role of mangroves in protecting the extensive sea-dike systems present along much of the low-lying deltaic coast of northern Vietnam. This indirect benefit is estimated through a model where the major parameters determining the value of the protection (valued at replacement cost through work days saved) are the width and age of the stand, and the local hydrological features. The deterministic model is calibrated for the area, and gives plausible results for regular maintenance costs. However, a further set of models would need to be developed to examine the impact of global change, such as change in the incidence of severe storms, or of mean sea level rise, for the area (see Tri *et al.*, 1998).

All of these costs and benefits are shown in Table 5.2, demonstrating how the cost benefit calculations are made. Discounting these to the present day (step 5) takes place by giving lower weight to future costs and benefits. This is carried out to reflect observed economic behaviour (the future is discounted in capital markets and in investment and consumption decisions more generally). It also reflects the necessity to appraise decisions on behalf of society - society may wish to give greater weight to the future, particularly where the environment is concerned, and hence would adopt a low positive discount rate. A simple discussion on the rationale of discounting and its application is given in Hanley and Spash (1993, chapter 8) and Pearce (1993) (see also Price, 1993 and Markandya and Pearce, 1991). In Table 5.2 the discount factors for a 5 percent discount rate (calculated as $[1/(1+r)^t]$) are shown: the present value at each year represents the net benefits (total benefits - total costs * discount factor).

Table 5.2 Illustrative table of cost benefit calculations for mangrove rehabilitation over 20-year time horizon

	Benefits	**(VND per ha)**			**Costs**			**Undiscounted**		
Year	Fuelwood	Bee-keeping	Sea dike protection	Fisheries	Maintenance	Planting and thinning	Total Benefits	Total Costs	% discount	Discounted Net benefits
0			205447		82500	438000	205447	520500	1.000	-315053
1			209133	632500	82500		841633	82500	0.952	722984
2			212931	632500	82500		845431	82500	0.907	692001
3			216681	632500	82500		849181	82500	0.864	662288
4			220287	632500	82500		852787	82500	0.823	633717
5		6300	223694	632500	82500		862494	82500	0.784	611146
6	293220	6300	226880	632500	82500	179190	1158900	261690	0.746	669512
7		6300	229839	632500	82500		868639	82500	0.711	558694
8		6300	232575	632500	82500		871375	82500	0.677	533942
9	1075900	6300	235100	632500	82500	510125	1949800	592625	0.645	874847
10		6300	237459	632500	82500		876259	82500	0.614	487299
11		6300	237459	632500	82500		876259	82500	0.585	464094
12	153600	6300	237459	632500	82500	88000	1029859	170500	0.557	478523
13		6300	237459	632500	82500		876259	82500	0.530	420947
14		6300	237459	632500	82500		876259	82500	0.505	400902
15	510300	6300	237459	632500	82500	259875	1386559	342375	0.481	502270
16		6300	237459	632500	82500		876259	82500	0.458	363630
17		6300	237459	632500	82500		876259	82500	0.436	346314
18		6300	237459	632500	82500		876259	82500	0.416	329823
19		6300	237459	632500	82500		876259	82500	0.396	314117
20	10535000	6300	237459	632500	82500	1448562	11411259	1531062	0.377	3723742
								NPV = Σ PV	=	13475743

Net present value represents the comparable value at the present time and is the sum of these discounted net benefits.

The results of the cost benefit analysis are presented in Table 5.3. The calculations compare only establishment and extraction costs, with the direct benefits from extracted marketable products, and with the indirect benefits of avoided maintenance of the sea dike system. Thus it does not include valuation of biodiversity or of the links to offshore fisheries, as undertaken, for example, by Ruitenbeek (1994), Swallow (1994) and Barbier and Strand (1998).

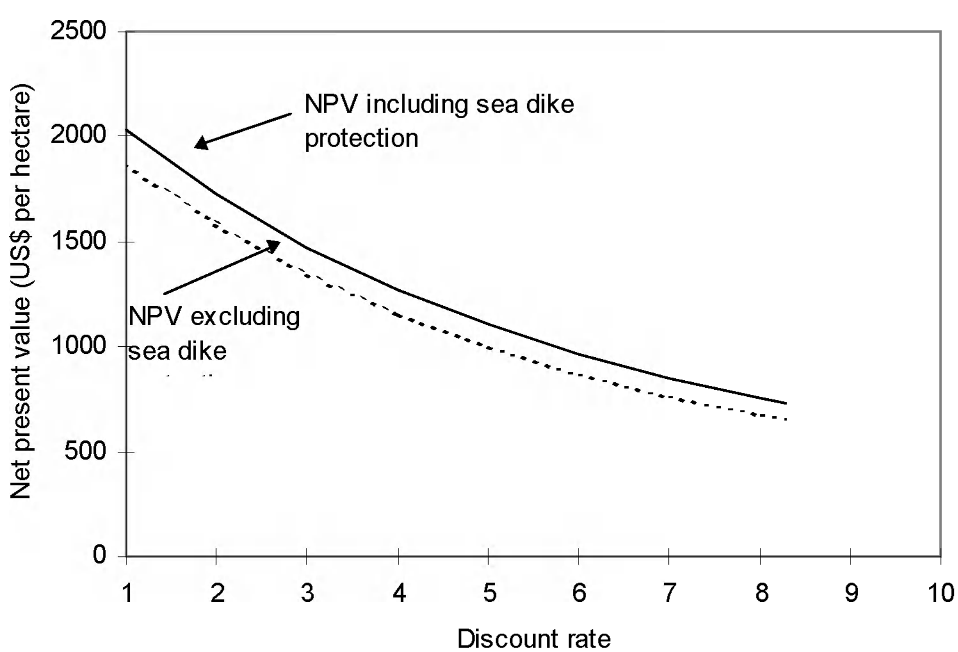
Table 5.3 Costs and benefits of direct and indirect use values of mangrove restoration compared.

Discount rate	Direct benefits (PV million VND per ha)	Indirect benefits (PV million VND per ha)	Costs (PV million VND per ha)	Overall B/C ratio
3	18.26	1.40	3.45	5.69
6	12.08	1.04	2.51	5.22
10	7.72	0.75	1.82	4.65

Note: US$1 = VND 11,000. B/C ratio = NPV Total Benefits / NPV Costs.
Source: Tri *et al.* (1998).

Is mangrove planting desirable? This is step 6 of the cost benefit analysis and is taken with reference to net present value or the benefit to cost ratio. A benefit to cost ratio is the ratio of the present value of benefits to the present value of costs, and is an alternative and equivalent indicator to NPV. A project is desirable if the B-C ratio is greater than one. The results show a benefit to cost ratio in the range of 4 to 5 for a range of discount rates, which means mangrove rehabilitation can be justified on economic grounds for all the discount rates analysed.

Figure 5.2 Net present value of mangrove rehabilitation including value of sea dike protection by discount rate.



Source: Tri *et al.* (1998)

The final step (7) is sensitivity analysis and scaling, where appropriate. This is undertaken in this case by examining sensitivity of the results to discount rates and to the inclusion or exclusion of the value of particular functions. Figure 5.2 illustrates that the direct benefits from mangrove rehabilitation are more significant in economic terms than the indirect benefits associated with sea dike protection. The sea dike protection estimates do not include the benefits of reduced repair of storm damage, or the potential losses of agricultural produce when flooding occurs. Flooding associated with severe tropical storms can lead to large economic losses, as well as to loss of life, and a reduced probability of flooding associated with the protection from the mangrove itself would be another indirect benefit. It is clear from the figure that, even if no indirect benefits were available, the direct benefits from mangrove rehabilitation justify this activity as economically desirable and hence the prescriptive response based on this cost benefit analysis is not sensitive to the inclusion of the indirect benefit. This is shown by the positive Net Present Values at all discount rates considered, demonstrating *inter alia* that mangrove planting in these circumstances is also not sensitive to the discount rate chosen. Choosing the rate of discount is considered by many economists to be somewhat arbitrary and dependent on whether the project to be appraised is being undertaken in the public or private domains. A range of real discount rates from 1 to 20 have been used in many circumstances, but as discussed above, rates at the lower end of this range tend to reflect the time preferences implicitly applied by governments in investments on behalf of society (see Markandya and Pearce, 1991).

The case study therefore illustrates each of the steps in this appraisal of costs and benefits. It is argued that this represents one way to operationalise the impacts and response parts of the P-S-I-R framework: when faced by environmental changes, the impacts of these on society (in economic terms where appropriate) can be estimated and the responses examined within the range of economic alternatives offered. In the mangrove case presented here, investment in mangrove planting in Vietnam's coastal zone has various impacts on coastal residents, with the benefits outweighing the costs. The estimation of these requires both socio-economic surveys and modelling of the sea dike maintenance function. It would appear that planting is desirable and this result is robust to various scenarios of discount rates and changes in the benefits stream. Economic analysis is not the only means by which this element of P-S-I-R can be operationalised, as discussed extensively in the main sections of the report. The distribution of costs and benefits in coastal zone changes (Adger *et al.*, 1997 on distributional consequences of mangrove conversion) and the incorporation of stakeholder preferences into decision-making (Brown *et al.*, 1998 on multi-criteria analysis) require that a range of flexible socio-economic approaches are adopted.

5.3 The Integration of Systems Analysis for Analysing Pressure, State and Response to Environmental Change: a Model of South-West Sulawesi, Indonesia

The approach and case study site

For the effective management of coastal resources, information is desirable on both the spatial and temporal effects of the analysed policies on the coastal zone system. To this aim a dynamic system model for the biophysical and socio-economic coastal-zone interactions has been integrated with a spatial model based on the technique of Constrained Cellular Automata (Uljee *et al.*, 1996; White and Engelen, 1994).

The central aim of this case study is to demonstrate a method, developed as a computer-based decision-support system, by which coastal decision-makers can gain insight into the short and long term consequences of their decisions. The research is funded by WOTRO (Netherlands) and involves multidisciplinary co-operation in the fields of economics, geography, anthropology, fisheries science, oceanography, marine biology and management science, from the Netherlands and Indonesia. A detailed discussion of the project and the theoretical framework of analysis and documentation accompanying the model can be found in de Kok (1996), de Kok and Wind (1996), de Kok *et al.* (1997). Two Internet sites provide a detailed description of the Sulawesi case study and the model (including downloadable demo version and user manual). Addresses are: **http://www.minvenw.nl/projects/netcoast/ramco/** and **http://www.zod.wau.nl/wotroczm/**). A brief summary of the study area and of the prototype decision-support system based on the case study, are given here.

The case study application is based on a coastal zone of South-West Sulawesi, Indonesia. The aim of the ongoing research program is to obtain the scientific knowledge required to support the sustainable management of coastal resources in tropical countries. This requires information on the main biophysical, socio-economic and socio-cultural processes affecting the sustainable use of coastal resources such as mangroves, seagrass beds, coral reefs and associated fish species. In order to facilitate communication with end-users and provide a modelling tool a prototype version of a **Ra**pid Assessment **M**odule for **Co**astal-zone management **(RaMCo)** has been developed .

The study area is located in the southern part of the province of South Sulawesi and consists of five rural regions and the urban district of Ujung Pandang. The area comprises a mainland shore and a 40 km broad shelf, the Spermonde archipelago, in which a multitude of coral reefs can be found. Four ecological coral zones parallel to the coast have been identified. The outer shelf reefs are most exposed to storm-generated waves, whereas the reefs on the inner shelf are dominantly influenced by land-based processes. The main city in the region, Ujung Pandang, has a fast-growing population of over 1.2 million inhabitants in 1995, which is expected to double in twenty years. A clear stratification of resource use can be observed in the research area. Fisheries and reef exploitation are the main source of income on the islands of the Spermonde archipelago. Fish and other marine animals are caught around reefs or in the open sea. Near the coast brackish-water shrimp ponds are used to cultivate fish, prawns and seaweed. The river delta is dominated by irrigated rice-fields or sawah's. Most of the alluvial soil of the river delta is used to cultivate rice on irrigated fields or paddies (sawah culture). More upstream the soil is used for dry-field agriculture based on crops such as corn, sweet potatoes and cassava.

The urban region of Ujung Pandang provides the main source of non-rural employment in the region and exerts a major pull force on the working population of the rural areas. Major projects are ongoing or planned to develop the urban area including the Makassar harbour, the nearby Hasanuddin airport and regional tourism. A large dam is being built near BiliBili, about 20 km upstream from the Jeneberang river mouth, to cope with the anticipated future increase in the municipal and agricultural demand for water.

During the wet season the Jeneberang river is a major outlet of terrigenous sediments. The resulting change in the coastal water turbidity threatens nearshore seagrass beds and coral reefs. Soil-eroding land-use practices in the catchment area contribute to the sediment level. The use of fertilisers and pesticides in irrigated rice culture enhances the level of nutrients in the coastal waters which can lead to the eutrophication of coastal waters. The construction of a dam in the river will strongly affect the discharge of water and sediment. The exact consequences for the coastal morphology are not yet known. On the other hand, human-induced soil erosion in the hinterland may affect the functioning of the dam reservoir. Along the coast most of the mangrove stands, which provided a natural coastal protection, have been cut to provide room for brackish-water ponds in which highly profitable prawns are cultivated. Waste water discharge from the shrimp ponds also forms a major source of coastal waters pollution due to the intensive utilisation of antibiotics and overfeeding of the shrimps. Marine fisheries and reef exploitation are mainly small-scale but intensive in the Spermonde archipelago. Overexploitation and destructive fishing methods such as the use of dynamite and cyanide poison result in declined catches from the area and destruction of the associated coral reef habitat. Several problems arise with regard to the environment and water management. The agricultural runoff of fertilisers and pesticides may affect the quality of water used for nearby tambaks.

The productivity of the rice fields almost solely depends on the amount of irrigation. The question is whether the dam under construction near BiliBili will be able to meet the future demand for irrigation water. At present, less than 50 percent of the demand is met by the municipal water supply. It is unlikely that the water supply from the designed reservoir will be able to keep up with the rapid urbanisation of the city. The catchment area upstream of the dam is characterised by considerable soil erosion, increased by intensive land-use for horticulture products and large-scale forestry. Without proper counter-measures the resulting runoff of sediments can reduce the life of the reservoir. In a less direct way agricultural activities in the hinterland may similarly affect the coastal waters. The increased cultivation and subsequent erosion of the upland areas has resulted in an increase of the input of suspended sediments in the coastal waters. Domestic waste water is a major source of organic and bacteriological pollution as the sewage is discharged directly into the sea without sufficient treatment. A further increase of the total pollution load can be expected with the ongoing urbanisation.

Methodology for developing a decision-support system

The simulation model has been developed to support the preparatory and planning stage of coastal-zone management. In policy preparation three steps can be distinguished, which differ as far as the objectives and the time and costs are concerned (de Kok and Wind, 1996). The aim of *problem analysis and diagnosis*, which is of qualitative nature, is to outline the policy problem in the study area. *Rapid assessment* is an engineering approach towards selecting promising solutions, finding weak spots in the analysis and determining the elements in the analysis for which the solutions are most sensitive. This modelling phase is quantitative but global. In order to be able to carry out a rapid assessment in a relatively short time, the rapid assessment is particularly based on readily available knowledge, data and information. In design terms, a rapid assessment is a preliminary design and paves the road toward the detailed design: the *comprehensive analysis*. The result of the comprehensive analysis is a detailed analysis of policy alternatives and their impacts. The difference between rapid assessment and comprehensive assessment concerns primarily the resolution in space and time and the corresponding level of detail of the required information. The RaMCo prototype model developed for South-West Sulawesi falls in the category of rapid assessment.

Before a coastal zone manager can decide on the course of action to take, including non-intervention, not only the objectives must be clear but also the constraints within which the coastal zone manager must act. Furthermore the coastal zone manager must have a clear indication of the consequences of the alternative actions. This requires models to predict what will happen if the coastal zone manager chooses one alternative, given the existing state of the coastal zone and exogenous conditions. There are many strategies or plans of action for a policy analysis. Miser and Quade (1985) have presented a general methodology for policy analysis, which serves as the guideline for management research. The steps to be taken include:

- problem formulation
- identification, design, and screening of alternatives
- building of models to predict consequences

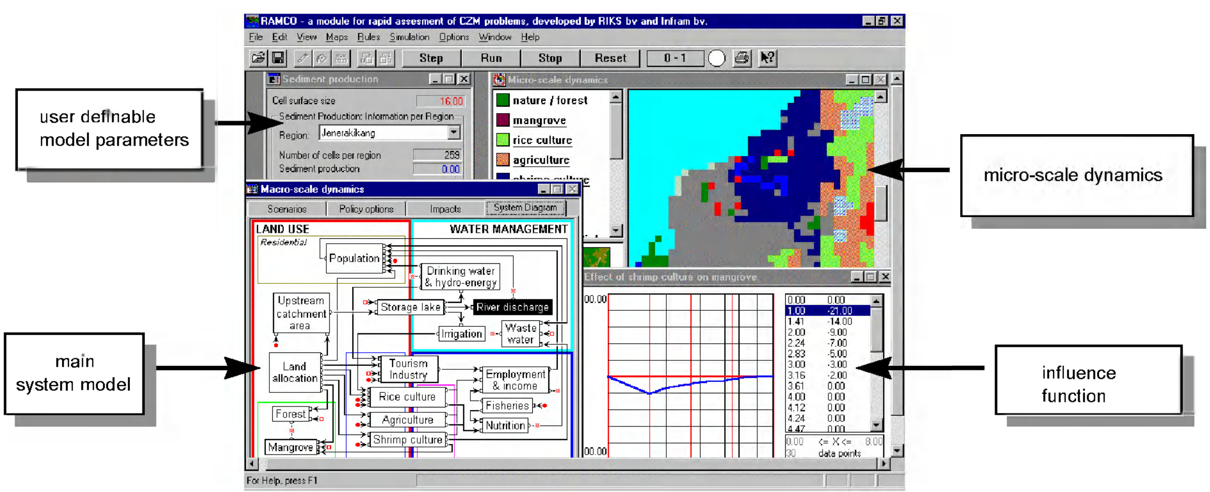
- forecasting future contexts
- ranking and comparison of alternatives

The simulation model

The RaMCo model combines a Geographical Information System with a dynamic system model for the biophysical and socio-economic coastal-zone interactions. For a given set of demographic and economic conditions, the development of the coastal zone over a period of twenty-five years can be simulated. The model allows the user to study the impacts of different *management interventions*, such as the construction of a storage lake or investments in local tourism, on the coastal zone system. Different *scenarios* for the demographic, economic, and hydrological conditions can be introduced.

The user interface of the prototype developed for South-West Sulawesi comprises four windows (Figure 5.3): a *system window* which allows access to the model structure and models used (including references), a *scenario window* to introduce different climatological, demographic, and economic external conditions, a *policy option window* which enables the implementation of a number of policy options such as the construction of a storage lake and the rehabilitation of mangroves, and an *impact window* with the socio-economic and physical consequences of the policy alternatives. Different scenarios for the demographic, economic, and hydrological conditions can be introduced. During the simulation, tables and maps are provided to allow the user to study the socio-economic and ecological consequences of the different policy options. In the model a distinction is made between the *macro-scale* model and the *micro-scale* model.

Figure 5.3 Main screen of RaMCo showing the macro-scale and micro-scale models, the user interface and some of the dialogue boxes



Source: model based on Uljee *et al.* (1996).

At the macro-scale level the temporal dynamics of the coastal-zone system are described. Examples of processes included in the macro-scale model are the discharge of sediments from the river catchment, the market-driven expansion of various agricultural sectors such rice and shrimp culture, and the urbanisation of the city of Ujung Pandang. The driving mechanisms for these processes consist of population growth, price developments and the hydrological conditions, and can be defined in the scenario screen of the model. Tourism and industrial development can be stimulated through short- and long-term investments.

At the micro-scale level the spatial consequences of the dynamic changes area dealt with. The spatial distribution of the sector is represented at the micro-level, using Constrained Cellular Automata (White and Engelen, 1994). The research area is described by a grid of square cells, each representing an area of 16 ha. The state of a cell represents a specific type of land use, such as mangrove forest, industry, or shrimp culture. During the simulation, state changes of the cellular automata are the result of the consistency with the macro-scale model and micro-scale dynamics. The latter include geographical suitability for the different types of land use in the location at hand and the interaction of each cell with the surrounding cells.

Simulation models as integrative tools

The RaMCo prototype is a first step towards an interactive and integrated modelling tool for analysing and comparing coastal policies by combining temporal and spatial dynamics. As such the model builds a bridge between existing dynamic simulation models and Geographical Information Systems. At the present stage the simulation results should not be interpreted as predictions, but qualitative indications of the response of the coastal zone to certain types of management intervention. The RaMCo prototype not only serves as a demonstration tool for coastal decision makers. The flexibility of the model also allows for the incorporation of new data or theoretical concepts by coastal analysts. The problem-based design methodology contributes to the effectiveness of the envisaged decision-support system. Communication between the decision makers and the modellers, especially during the initial phases of the design, is an essential precondition for the development of an appropriate tool for analysing coastal measures.

5.4 Evaluating the Economic and Physical Impact of Scenarios for Tokyo Bay, Japan

Introduction

Tokyo Bay is located approximately at the centre of the archipelago of Japan, with its mouth opening to the Pacific Ocean. It is relatively small, slightly less than 1000 km^2 in its inner bay area and is surrounded by a very densely populated metropolitan area. The coast around the bay is one of the most heavily industrialised areas of the world.

Between 1945 and the late 1980s, pressures intensified in and around the Bay region (Bower and Takao, 1993). As a result of the urban and industrial development, vast quantities of wastewater and liquid wastes were discharged into the bay; water quality deterioration culminated during the early 1970s. A decade later, the water quality of the bay began to recover, due to severe measures taken by central and local governments. Multiple and conflicting resource use demands were identified in the Bay by the 1980s, including waste disposal (liquid and solid waste), land reclamation for port and industrial activities, marine transport, commercial fishing and recreation (Bower and Takao, 1996).

The study reported in this section was undertaken in the mid-1980s, focusing on aspects concerning the use of Tokyo Bay. The main pressures were identified alongside the evolution of uses of the Bay and their mixes over time, in relation to management policies. To illustrate the net benefits to society accruing from different management strategies, this section analyses alternative future scenarios affecting the Bay, bearing in mind that different governmental actions or ‘policies’ can lead to alternative mixes of uses of Tokyo Bay.

Net benefits criterion

To calculate the maximum net social benefits to society, the analysis required that the costs and benefits of each mix be estimated; scenario analysis was utilised, based on the following conditions:

- focus on two points in time: 1980 (the base year) and 2000,
- benefits and costs expressed in 1980 price levels,
- conversion of capital costs (investment) to annualised costs based on a capital recovery factor of 0.1, equivalent to an interest rate of 7.75 percent for 20 years. Costs and benefits were calculated as annual figures. The only criterion adopted in the evaluation was net annual benefits (= benefits:costs),
- major projects already in progress were assumed to be completed by 2000.

Any combination of policies will lead to a management strategy for the area; given that several combinations are possible, only a few are considered. Benefits and costs that would be achieved if each strategy were implemented were estimated across a number of possible combinations of different policies or 'mixes of outputs'. A series of cases were identified to explore the implications of alternative mixes of uses in terms of net benefits to society. A case was defined as a combination of values of scenario variables (S) and policy variables (P).

S variables relate to variations in socio-economic conditions and are composed of population and industrial activities in the Tokyo Bay region. P variables correspond to government actions which directly affected the utilisation of Tokyo Bay, e.g. waste disposal regulations, recreational site provision.

Environmental change scenarios

Three S scenarios were defined, each one a combination of level of population and level of industrial activity (see Table 5.4):

S-0 = no change from base year 1980 ('base case');
S-1 = moderate growth in both population and industrial activity;
S-2 = rapid growth in population and industrial activity.

Table 5.4. Estimates of population and industrial activity by the year 2000 for each scenario, based on government research for four prefectures of the Bay.

Prefecture	Item	Base (S-0)	S-1	S-2
Tokyo	Population	11.5	11.5	11.6
	Industrial output	14.5	17.0	20.4
Kanagawa	Population	6.9	7.2	7.4
	Industrial output	16.0	22.4	25.1
Chiba	Population	4.7	5.4	5.6
	Industrial output	7.7	14.0	14.3
Saitama	Population	5.4	5.8	6.1
	Industrial output	7.2	9.0	11.4
Total	Population	28.5	29.9	30.7
	Industrial output	45.5	62.4	71.2

Source: Takao and Bower (forthcoming).
Note: Base case (S-0) represents conditions in 1980; population is expressed in millions of inhabitants; industrial output unit is in billion yen.

The policy variables (P) used in the analysis refer to:

A. liquid waste disposal policies. Three cases were devised based on waste disposal volumes entering the Bay (LW-0; LW-1 and LW-2), taking into account: i) extent of sewerage network coverage in the Tokyo Bay watershed; ii) degree of waste removal from the liquid discharges emitted by municipal sewage treatment plants and industrial activities (Table 5.5);

B. water-oriented recreation. An increase in the demand for water-based recreational activities was estimated for the period of analysis. It was estimated that the feasibility of this increase would depend on: i) maintenance of water quality; ii) access to recreational activities. Policies to achieve these goals (R) would therefore depend on the conversion and additional development of facilities for recreation by the government. Three options were examined:
   - R-0 = no change from situation or level of facilities in 1980;
   - R-1 = conversion of old port facilities to recreational use and development of man-made beaches as of 1980;
   - R-2 = as R-1 and including additional conversion to recreational use of low-handling piers and wharves and of other areas where plans for recreational use do not exist.

Table 5.5. Values of liquid waste disposal policies in the year 2000 for Tokyo Bay.

Policy elements		Liquid waste disposal policy		
		LW-0	LW-1	LW-2
Extent of connection to sewerage systems (%)		42	75	82
Reduction in discharges				
Additional reduction in discharges from municipal STPs (%)	COD	0	0	50
	DIN	0	0	70
Additional reduction in discharges from direct discharging industrial activities (%)	COD	0	20	50
	DIN	0	0	50

Source: Takao and Bower (forthcoming).
Note: STP = sewage treatment plant.

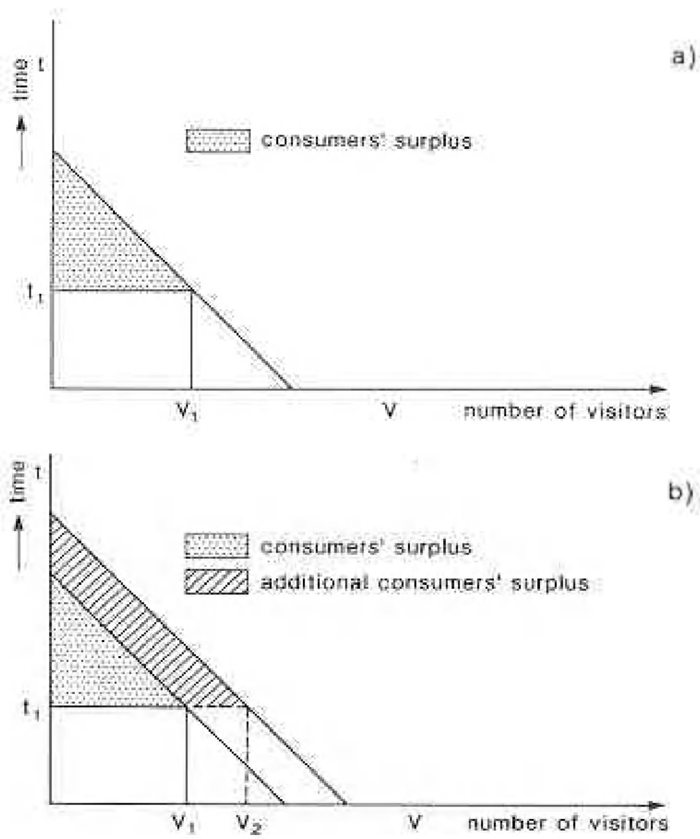
The study assumed that the demands for various types of water-oriented recreational activities would increase substantially over the period of the analysis (1980 - 2000), as a function of leisure time and increased per capita income. Therefore the two scenarios in addition to the base case (S-0 to S-2), three liquid waste disposal policies (LW-0 to LW-2) and three alternative recreational policies (R-0 to R-2) yielded eighteen cases for analysis.

Consumer surplus and travel cost method

It was established that, within the prescribed time horizon, no adverse impacts to the bay would arise from other policies enhancing fisheries, marine transport and solid waste disposal. Thus, the key policy change in the management of Tokyo Bay relates to recreation and the benefits that could accrue from future increases in water-based recreational demand. Any benefits accruing from the management of the bay would therefore devolve into benefits related to water-based recreation.

These benefits were estimated using the concept of consumers` surplus, valued via a travel cost model, illustrated below and with the method outlined in detail in Appendix 3. The extent to which individuals from a given zone in a region will visit a given recreation site for day recreation in a year is assumed to be a function of the time-distance, t, to the site and the quality, Q, of the site. So visits, V, = f(t,Q). As distance from the Bay increases so, other things being equal, the number of visitors decline (see Figure 5.4.a).

Figure 5.4 a and b Relationship between time and distance to recreation site and number of visitors: a) for a given quality at site and b) for improved quality at site.



Given a distance t_1, the number of visitors is V_1. As the diagram illustrates, all visitors up to V_1 are willing to travel for longer time to the site (t). Translating this time t into travel cost (travel expenditure + opportunity cost) provides a willingness-to-pay measure of individual visitors for the recreational experience (expressed on the ordinate of the graph). However, all individuals to the left of V_1 have to pay less than they are willing to pay, so that there is a `consumer surplus` (represented by the hatched area in Figure 5.4a).

By improving the quality of the site, circumstances change (see Figure 5.4.b). If distance remains the same (t_1) more visitors (V_2) and an increase in consumer surplus value can be expected. This increase in consumer surplus represents the benefits arising from improvement in quality. However, the increase in consumer surplus may or may not be larger than the costs of quality improvement measures.

Monetary estimates of consumer surplus can be derived by combining a physical demand function of a recreational activity with estimates of unit travel cost and of the population from which the recreationists originate (see Takao and Bower, forthcoming).

Using the series of policies outlined above in conjunction with a water quality model, the quality of effluents in terms of chemical oxygen demand (COD) and dissolved inorganic nitrogen (DIN) in Bay waters are obtained. Modelling the aquatic ecosystem in fact allows outcomes of different policies to be related to changes in ambient water quality, which in turn influence the type and extent of water-based recreation activity. A steady-state, two-layer, two-dimensional model was used (calibrated with data for 1978-82); it included parameters on primary production, decomposition, inputs and outputs of COD and DIN, tidal influences. The model revealed that substantial reductions in load were required before any significant improvement in ambient water quality could be achieved.

Estimating costs

Costs were estimated for the liquid waste disposal policies (LW) and for R policies. The liquid waste disposal costs at sewage treatment plants, capital costs and operation and maintenance costs were derived from cost functions available in documentation provided by the Japanese government.

With respect to the costs of liquid waste disposal from industrial activities, reduction of discharges was assumed to be by treatment using the activated sludge process. To achieve an additional 20 percent reduction in COD discharges with the LW-1 policy required the addition of a flocculation-filtration system to the activated sludge process. The cost of this system was estimated to be 20 percent of the cost of the activated sludge system.

Table 5.6 Incremental costs to meet scenario LW-policy combinations in 2000.

LW policy	Scenario S-1		Scenario S-2	
	LW-1 billion yen	LW-2 billion yen	LW-1 billion yen	LW-2 billion yen
Capital costs				
STPs	0	265	0	272
Industrial activities	107	292	107	292
Total capital costs	107	557	107	564
Annualised capital costs	11	56	11	56
Operation and maintenance costs				
STPs	0	7	0	7
Industrial activities	5	14	5	14
Total operation and maintenance costs	5	21	5	21
Total annualised costs	16	77	16	77

Source: Takao and Bower, forthcoming.
Note: All values are expressed in 10^9 1980 yen; annualised capital costs are calculated with a recovery factor of 0.1 (ca. 7.75 percent at 20 years).

The LW-2 policy requires industrial operations to achieve a 50 percent reduction in discharge of COD and a 50 percent reduction in discharge of DIN. The incremental cost to achieve an additional 30 percent reduction in COD discharge was estimated to be 1.5 times the LW-1 increment. It was estimated, based on a study by Tokyo Metropolitan government, that capital and operation and maintenance costs for reduction of DIN discharge for the industrial discharges would be 25 billion yen and 1.2 billion yen respectively. The costs to sewage treatment plants (STPs) and industrial activities to meet the various scenario-policy combinations are shown in Table 5.6.

In relation to recreation policy (R), capital costs consist of: a) converting the old-fashioned and inefficient port facilities to recreation uses; b) developing man-made beaches; c) installing recreational facilities, plus annual operation and maintenance costs associated with these investments (Table 5.7).

Table 5.7. Costs (estimated) for beaches, piers and related facilities (development and maintenance) for water-based recreation, under R-1 and R-2 policies, in year 2000.

Item	Recreation policy	
	R-1	R-2
Length of shore (km)	2.2	8.6
Length of piers (km)	2	99
Total length (km)	4.2	107.6
Capital costs (10^9 yen)	9	74
Annualised capital costs (10^9 yen)	0.9	7.4
Operation and maintenance costs (10^9 yen)	0.4	10.8
Total annualised costs	**1.3**	**18.2**

Source: Takao and Bower, forthcoming.

Note: Total length = increment to R-0 conditions
Annualised capital costs calculated at recovery factor = 0.1 (ca. 7.75 percent at 20 years)
Operation and maintenance costs based on 0.1 x 10^9 yen/km/year

Estimation of benefits

Based on observed recreation behaviour, the Tokyo Bay region was divided into eleven residential zones, from which the recreationists originate. These recreationists were assumed to visit one of five recreation zones along the coastal area of the Bay. The estimated travel cost per head of travel was 1210 yen, a combination of expenditure and the opportunity cost in terms of wage income for zone. This estimate was inflated (because of expected labour productivity growth in the future) to 1440 yen in the S-1 scenario, and 1570 yen in the S-2 scenario. It was also necessary to calculate carrying capacity limits for the recreation zones and different recreation activities. Table 5.8 illustrates the estimated gross benefits in relation to water-based recreation.

Table 5.8 Estimated gross benefits of recreation under alternative scenario-policy combinations (expressed in 10^9 1980 yen).

Scenario	Policy	Coastal Bathing	Shell-gathering	On-shore fishing	Total gross benefit
S-1 (moderate growth)	LW-0; R-0 (no change in efficient disposal; no extra recreational facilities)	1.1	1.4	-0.7	1.8
	(extensive effluent treatment; extra recreational facilities)	11.3	19.0	60.2	90.5
S-2 (high growth)	LW-0; R-0	1.6	1.7	-1.5	1.8
	LW-2; R-2	12.7	21.3	64.5	98.5

Source: adapted from Bower and Takao (1993).

Net benefits are estimated by combining gross benefits and costs, as mentioned above. Table 5.9 summarises the net benefit outcomes for selected management cases. In almost all cases, the monetary value of the potential recreation demand is larger than the realised demand, due to the limits of the operational capacities of the recreational facilities.

Table 5.9 Costs and benefits for selected cases, management of Tokyo Bay.

Scenario	Policy		Benefits: Gross Benefits	Costs: Liquid Wastes Disposal	Costs: Recreational Facilities	Costs: TOTAL	Net Benefits
S-1	LW-0	R-0	1.8	0	0	0	1.8
		R-1	2.8	0	1.3	1.3	1.5
		R-2	14.6	0	18.2	18.2	-3.6
	LW-1	R-0	-0.2	16	0	16	-15.8
		R-1	1.0	16	1.3	17.3	-16.3
		R-2	11.3	16	18.2	34.2	-22.9
	LW-2	R-0	44.2	77	0	77	-32.8
		R-1	47.7	77	1.3	78.3	-30.6
		R-2	90.5	77	18.2	95.2	-4.7
S-2	LW-0	R-0	1.8	0	0	0	1.8
		R-1	2.8	0	1.3	1.4	1.5
		R-2	12.9	0	18.2	18.2	-5.3
	LW-1	R-0	-0.2	16	0	16	-16.2
		R-1	1.0	16	1.3	17.3	-16.3
		R-2	10.9	16	18.2	34.2	-23.2
	LW-2	R-0	52.5	77	0	77	-24.5
		R-1	56.4	77	1.3	78.3	-21.9
		R-2	98.5	77	18.2	95.2	3.3

Source: adapted from Bower and Takao (1993) and Takao and Bower, forthcoming.
Note: Benefits relate only to recreation and represent increment in consumer surplus. Costs represent capital costs x capital recovery factor of 0.1 + operation and maintenance costs. Benefits and costs relate to conditions in the terminal year of the period 2000. The net benefits do not represent the value of the time streams of costs and benefits.

Additional net benefits could be obtained by developing more recreational facilities, up to the level where the marginal annualised cost of the facilities is equal to the marginal benefit, assuming the same LW policies and that the recreational activities themselves do not result in deterioration of ambient water quality.

Except for the LW-0 policy cases, the estimated gross benefits from recreation exceed the estimated costs of the recreational facilities alone for the S-1/LW-2 and S-2/LW-2 cases. Substantial gross benefits from recreation are achieved only with policy LW-2, irrespective of the recreation policy. This suggests that reduction in DIN discharges, more than reduction in COD discharges, is critical in achieving ambient quality sufficient to induce more recreational activity.

Positive net benefits are achieved for LW-0/R-0 and LW-0/R-1 combinations for scenarios S-1 and S-2. This reflects the increased demand as a result of increased population, household income, leisure time, even with ambient water quality at the 1980 level. The only other positive net benefits outcome is S-2/LW-2/R-2.

The results indicate that two conditions are required for achieving significant net benefits from management of Tokyo Bay: 1) improved and maintained water quality; and 2) increased provision of recreational facilities. Once water quality has been improved to an adequate level (LW-2), additional recreational benefits can only be achieved by more inputs to provide recreational opportunities.

The values of the scenario variables had relatively little effect on results, compared with the values of the LW and the R variables. Finally, no attempt was made to assess the distribution of benefits and costs among groups in the Tokyo Bay region, although the distribution effects would be important considerations in choosing a strategy in reality.

5.5. Managing Nutrient Fluxes and Pollution in the Baltic: An Interdisciplinary Simulation Study.

Introduction

All countries with a coastline have an interest in the sustainable management of the coastal resource systems. The task of sustainable management is likely to be made more difficult because of the consequences of global environmental change. A particular characteristic of global environmental change is that it has led to, among other things, the progressive opening of biogenic nutrient cycles, e.g. increased mobility of nutrients which has meant increased exchanges between land and surface water and consequent impacts on the ecological functioning of aquatic systems.

Understanding the interactions between the coastal zone and global changes cannot be achieved by observational studies alone. Modelling of key environmental processes also has an important role to play. In particular, modelling work on the dynamics of carbon (C), nitrogen (N) and phosphorus (P) in the coastal ocean needs to be combined with socio-economic analysis of the drivers of C, N & P fluxes and the human welfare consequences of changes in these fluxes across the coastal zone over time. This study reports the overall results of interdisciplinary research which focused on N and P fluxes on a drainage basins-wide scale in the Baltic region (Turner *et al.*, 1995).

The aims of the study were: 1) to provide a comprehensive picture of the land use and ecological carrying capacity of the region, in relation to the patterns of human activity in the region; the P-S-I-R framework was adopted to facilitate the analysis; 2) to develop a model of different nutrient loading scenarios and their consequences on the ecological state of the Baltic Sea and its sub-systems; 3) to estimate the costs of various strategies designed to reduce the nutrient loading of the Baltic Sea, and the identification of the most cost-effective nutrient abatement options; 4) to estimate the economic valuation of eutrophication damage to the Baltic Sea; 5) to increase the understanding of the institutional issues involved in the management of the Baltic Sea.

P-S-I-R Framework applied to the Baltic study

It is possible to identify a group of interrelated socio-economic trends and pressures which contribute significantly to the Baltic's environmental change impacts and environmental risk to the marine ecosystem and the drainage basins' biophysical and socio-economic systems. The P-S-I-R framework is well suited as a conceptual model, to aid in the analysis of the Baltic Sea.

An increasing degree of environmental pressure has been felt in the Baltic region as a result of a range of socio-economic drivers. The outcome has been that the Baltic Sea and coastal zone resources have been subject to a range of usage demands. The economic and the environmental systems are now sufficiently interrelated as to be jointly determined. They are now in a process of co-evolution.

The Baltic Sea region catchment area covers around 1,670,000 km^2 and contains a population of about 85 million people. The Baltic Sea itself has a total surface area of 415,000 km^2; it is the largest brackish body of water in the world and a naturally very sensitive area. It depends on short and long term variations in climate and because of its semi-enclosed character, it has a very slow water exchange, the mean residence time for the entire water mass being of the order of 25-30 years (Folke *et al.*, 1991). This combination of biophysical and socio-economic characteristics has important implications for the environmental vulnerability of the Baltic Sea and its resource system.

The natural vulnerability of the area has been magnified by the magnitude and extent of socio-economic activities, impacts and interventions that have become commonplace since the 1950s. A significant proportion of the world's industrial production comes from this area, but up until around forty years ago there was little recorded environmental damage in the Baltic Sea. However, since 1960, the environmental condition of the Baltic Sea has increasingly become a cause of public concern and is currently perceived to be in an unacceptably polluted state; eutrophication is a major problem facing policymakers and the public.

Because of the sheer scale of economic activity the pollution generated is a pervasive problem across the drainage basin and beyond (e.g., municipalities, industries and agriculture). Evaluating the importance (in human welfare terms) of the various environmental impacts requires that their effects be measured in biophysical and then in monetary terms, providing some measure of the state of the Baltic environment and the importance of the environmental degradation that has taken place.

Market and policy intervention 'failures'

It is important therefore to know what the principal causes of the resource degradation and pollution problems in the Baltic Sea are. One way of analysing these problems is to identify a set of interrelated 'failures' phenomena which seem to underlie the degradation and quality decline trends. Two main related 'failures' can be distinguished, market failure and policy intervention failure, which when combined with scientific and social uncertainties (information failure) can account for the environmental damage process.

Of the various 'failures' relevant to the Baltic context, the most widespread type of market failure is that of pollution externalities. External costs result from waste generators (municipalities, industry and farms) who over-utilise the waste assimilative capacity of the ambient environment, e.g., rivers and the Baltic Sea, because this environmental function is perceived to be virtually free of charge (absence of market prices). Some waste generators also have had almost open access to the marine waste repository over time.

Government interventions have also been partly responsible for the environmental degradation process in the Baltic. There is a general absence of properly integrated coastal resource management policies and water catchment management and planning. This has resulted in intersectoral policy inconsistencies and resource depletion and degradation, with the loss of wetland ecosystems being an important damage impact.

Although these 'failures' phenomena are or were pervasive across the entire drainage basin, they tend to be focused in greater numbers and with greater severity in Poland, Russia, the Baltic Republics and the Slovak and Czech Republics, partly as a result of the historical legacy left by a central planning system based on input-intensive, inefficient heavy industry complexes. In the Nordic countries and the western parts of Germany, municipal and industrial pollution loads have been significantly reduced over the last few decades. Nevertheless, the agricultural sector poses problems due to the intensive nature of the farming regimes.

The dire message for Baltic policymakers in the future is clear: if the agricultural sectors in Poland etc., develop intensive methods similar to those fertiliser/pesticide dominated regimes commonplace in Denmark and Sweden, the outlook for the reduction of eutrophication pollution is poor.

Land use, nutrient loads and damage in the Baltic Sea

The current status of the Baltic Sea is determined by the set of activities present in the entire drainage basin. The load of nutrients to the various sub-drainage basins is determined by several factors such as land use, population density, climate, hydrology, and air transportation of nitrogen oxides and ammonium. A set of Geographic Information System (GIS) map layers were created and used to generate information on the current landscape characteristics and population distribution patterns in the drainage basin. A description of the technical procedures and the primary data sources used to create each layer, as well as an assessment of data quality, is presented in Folke and Langaas (1995), Sweitzer and Langaas (1994), Sweitzer *et al.* (1996).

The map layers were combined to generate new results, basic statistics on land use and population in the drainage basin, and characteristics of the drainage basin as they relate to distance from the coast. The further away from the coast or from rivers that eutrophying substances are released, the more likely they are to be absorbed through ecosystem processes and prevented from entering the Baltic Sea. High population concentrations, agricultural land, and urbanised land are all important nutrient generation sources. Wetlands, forests and inland water bodies can act as natural filters and sinks for nutrients as well as other pollutants. The information on the location of various land uses and population within the drainage basin provides a useful basis for the estimation of nutrient load discharged directly into the Baltic Sea or transported by surface water (Table 5.10).

In 1993, the total load of nitrogen and phosphorus to the Baltic Sea amounted to approximately 1.022 million tonnes of N and 39 thousand tonnes of P. The largest basin of the Baltic Sea, the Baltic proper, receives about 85 percent of the total load of both nitrogen and phosphorus.

In principle, there are two major sources of waterborne nutrient loads: arable land and sewage treatment plants. In addition atmospheric transports of nitrogen are also deposited directly on the Baltic Sea; these originate not only from countries within the drainage basin, but from other external countries.

The agricultural sector, excluding the emissions of ammonium, accounts for one fifth of the total load of nitrogen. Other water transports of nutrients include flows from sewage treatment plants and air

Table 5.10 Landscape characteristics and population distribution in the Baltic drainage basin.

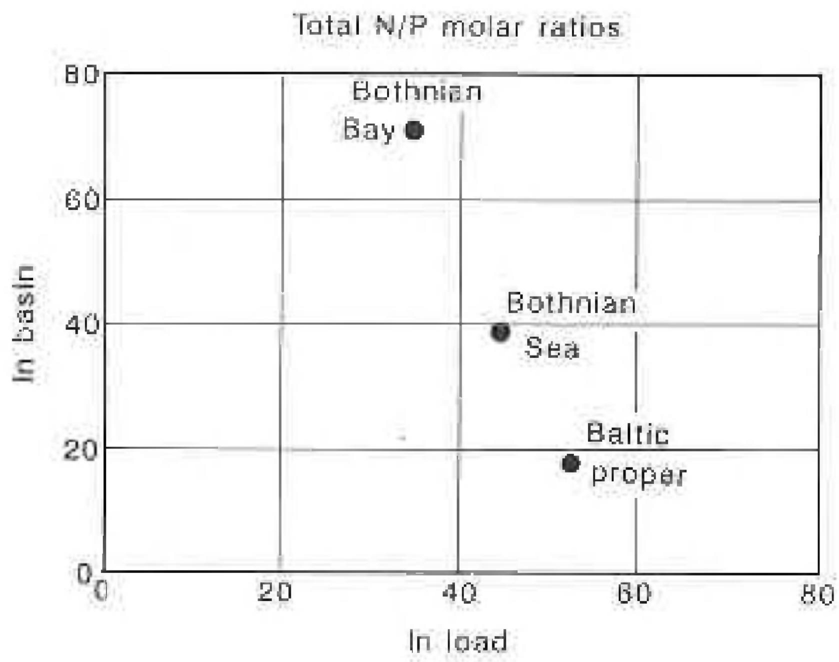
	Whole drainage basin	Within 10 km of the coast	Within 50 km from the coast
Landscape characteristics			
Forests as % of total	48%	5%	20%
Arable land as % of total	20%	8%	23%
Open lands (low or non-productive)	17%	5% pasture land	17% pasture land
Wetlands	8% mostly in the northern regions	2% inland water bodies	10% inland lakes
Population distribution			
% of populated area	100%	27%	43%
Total population within whole drainage basin	85 million in 14 countries	19% of total population	31% of total population
Within metropolitan areas (>250,000)	26% of total population	90% all urban	83% all urban
Within small towns or cities (250,000 >pop >200,000)	45% of total population		
Rural settlements (pop < 200,000)	29%	n/a	n/a
Baltic proper drainage areas	64% of total population	n/a	n/a

emissions deposited on land within the drainage basin (nearly 50 percent of the total N load). The direct discharges, mainly sewage treatment plants located at the coast, correspond to approximately 10 percent of the total load. Poland is the largest discharging country with respect to total nitrogen loading (28.5 percent), followed by Sweden (10.4 percent) and Germany (10.2 percent).

Poland is also the country providing the largest load of phosphorus to the Baltic Sea, approximately 50 percent of total load. The phosphorus load from the agricultural sector accounts for about one third of the total load and the direct discharges, mainly from sewage treatment plants, correspond to one quarter of the total load.

Although the impacts of greater nutrient input are well documented, the quantitative relationships between variations in loads of nutrients and concentration are poorly understood. The increased nutrient flux entering the Baltic Sea implies a higher concentration of a given nutrient which in turn may lead to an overabundance of phytoplankton production. Oxygen deficits occur and reduce the spatial extent of regions available for successful cod reproduction. On the other hand, the abundance of phytoplankton could lead to increases in zooplankton, contributing to augment the stocks of other fish species, providing enough oxygen is available to maintain these populations. N/P ratios in each sub-basin are significantly altered by the characteristics of the environment: water and nutrient residence times, load received, and internal biogeochemical processes (Figure 5.5). There is a gradient from north to south where the production in the Bothnian Bay is P-limited and the Baltic proper is N-limited (i.e. phosphorus rich). Using regional datasets and an expanded empirical budget model of the Baltic (Wulff, 1995) consisting of three coupled basins with advective water and nutrient transports among these and the Kattegat, the relationships between concentrations and nutrient sinks were derived.

Figure 5.5 Variations in N/P ratios



To select abatement measures geared towards improving the conditions of the Baltic Sea based on minimum cost, it is necessary to obtain information on the flow of nutrients from the land to the sea and their impact on the Baltic. Initially, however, the 'filter' and sink capacities of the wetlands in the drainage basin need to be estimated.

The natural wetlands in the drainage basin account for about 8 percent of the total area. Their nitrogen retention and elimination capacity was estimated to be close to 65,000 tonnes per year, taking only atmospheric downfall of nitrogen into account. Adding direct emissions per capita in terms of excretory release, in relation to the location of the wetland to human population densities, the nitrogen retention and elimination capacity was estimated at about 100,000 tonnes per year (Jansson *et al.*, in press).

The GIS-database was used to assess the spatial relationship between nutrient sources and sinks, creating maps to show the location of wetlands in relation to population centres in the Baltic drainage basin. The assumption is that wetlands will function more effectively as nutrient traps if they are in close proximity to nutrient sources. A visual assessment of the maps shows that areas with high concentrations of wetlands in the drainage basin are distant from the densely populated regions. Areas with moderate or low concentrations of wetlands tend to have low or moderate population densities. These results suggest that development and restoration of wetlands in highly populated and also intensively cultivated areas could be an effective and practicable means to reduce nutrient flows into the Baltic Sea.

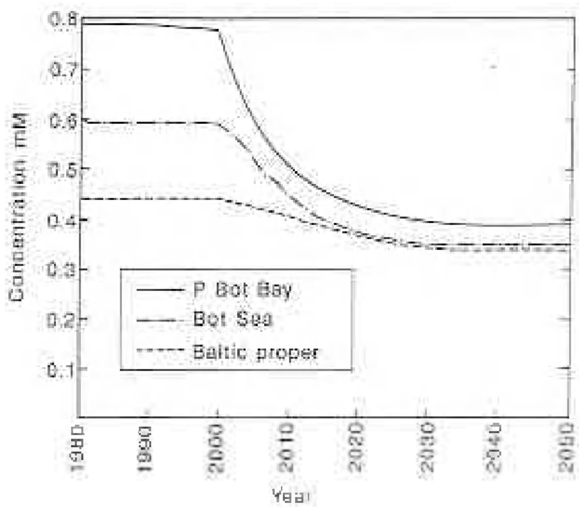
Therefore, the potential nitrogen retention and elimination capacity was estimated in a scenario where drained wetlands in the drainage basin would be restored. The capacity of wetlands to retain or eliminate nitrogen in such a scenario was estimated at about 180,000 tonnes per year (Jansson *et al.*, in press). Additional analyses on the nitrogen filtering capacity will be reported on below.

Nutrient reduction simulations

Two nutrient reduction simulations were modelled:

i) both N and P loads are reduced, but only to the Baltic proper.
Since the effect of eutrophication is most clearly seen in the Baltic proper, this scenario would be the most obvious choice for a future abatement strategy. In this scenario it is assumed that the nutrient reduction occurs instantaneously in year 2000 and the changes in concentrations follow on until a new steady state occurs. As can be seen from Figure 5.6 the nitrogen concentrations reach this new steady state within 10 years while it takes about 25 years for phosphorus, due to the inherently different behaviour of these nutrients in the Baltic as in most other marine systems. Denitrification represents an efficient internal nutrient sink for N while P reduction is less efficient in this brackish system. Simulations show final concentrations of P and N at about 50 percent and 70 percent of the current levels in the Baltic proper.

Figure 5.6 Reduction from current levels of both N and P load with 50 percent to Baltic proper



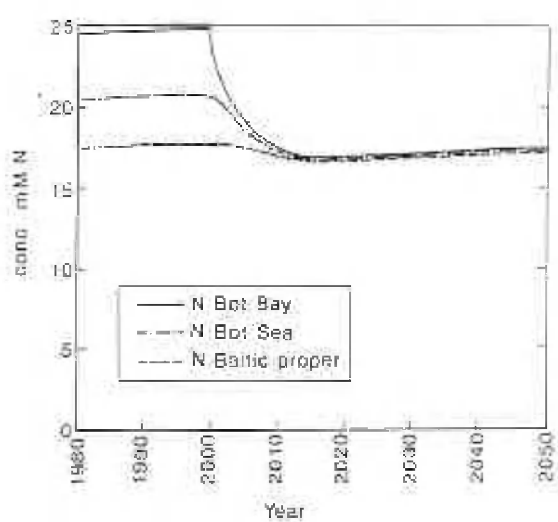
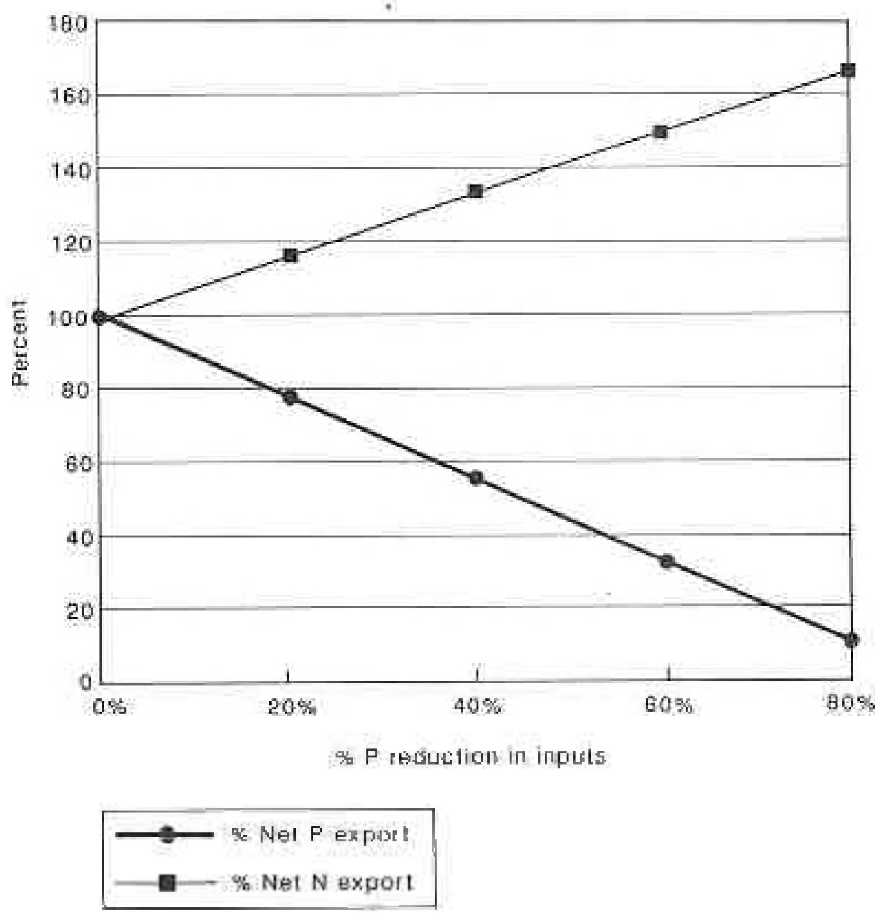


Figure 5.7 Percentage change in nitrogen and phosphorus export from the Gulf of Riga at different levels of reduction in P load



It is difficult to estimate the ecological consequences from the output of this model alone - the results have to be related to empirical knowledge of 'the state of the Baltic' with different concentrations of nutrients. The 'new' nutrient concentration corresponds roughly to levels found during the 1960s, before the drastic deterioration of the Baltic environment occurred; less primary production of organic matter and thus less frequent periods of oxygen deficiency in the deep basins would be expected. It is also likely that the decrease in P concentrations will reduce the frequency of cyanobacterial bloom during the late summer. These are now favoured by the high P concentrations (and low N/P ratio) found during summer and may cause accumulations of toxic algal mats on the surface of the Baltic proper.

ii) only the P load is reduced to the Baltic proper.

This scenario was considered since it is likely that the inputs of P are more easily reduced than those of N, as the sources are mostly municipal and agricultural (Wulff and Niemi, 1992). The model simulations show corresponding reductions of P concentrations on N (50 percent for the Baltic proper and 80 percent for the Bothnian Bay). A decrease of input and concentration of P means that less N will be utilised in the biogeochemical cycles, since these nutrients are utilised in fixed stochiometric Redfield ratios (16 moles of N for each mole P). Thus in this scenario, less N will be incorporated into organic matter and subsequently mineralised and denitrified (lost). The Baltic proper and Bothnian Sea would change from N-limited (i.e. phosphorus rich) to P-limited systems.

These simulations illustrate the consequences of nutrient reductions on a basin-wide scale. However, decisions about abatement policies are often made because of concerns at the local or regional level, rather than on the basis of large-scale environmental concerns. Possible conflicts arising from this situation are explored below in a regional study of the Gulf of Riga. This is a region of the Baltic where lack of sewage treatment contributes to very large inputs of P to the sea. A model was therefore built to explore the consequences of different N and P reduction schemes on the Gulf and the Baltic Sea. One model run is illustrated in Figure 5.7. The net exports of nitrogen and phosphorus to the Baltic are shown in relation to different levels of P reduction in the inputs of the Gulf. Naturally, a P reduction in inputs will result in reduced exports of P; however, more nitrogen will be exported to the (N-limited) Baltic proper.

The overall model clearly demonstrates that reduction of inputs to the Baltic proper is most efficient in reducing concentrations in this basin. A strategy where all inputs are uniformly reduced is not optimal, since the situation in the two northern basins is not critical in terms of eutrophication (only small amounts of nutrients are exported southwards). The simulations also demonstrate that both nitrogen and phosphorus inputs have to be reduced as emphasised in the regional Gulf of Riga study where it was shown that P removal might actually increase the net export to the off-shore Baltic proper. It has also been demonstrated that it will take several decades before the nutrient levels are returned to an acceptable level, particularly for phosphorus.

To understand the institutional implications of this result, consider a problem in which there are two basins, each controlled by a different country acting unilaterally. Basin A is phosphorus-limited (like the Gulf of Riga) while Basin B is nitrogen-limited (like the Baltic proper). Country A controls discharges into Basin A and has preferences only over the quality of this Basin. Similarly, Country B controls discharges into Basin B and cares only about eutrophication in this Basin. As Basin A is phosphorus-limited, A can enhance its welfare by reducing its phosphorus discharges and improving the state of Basin A. However, the reduction of phosphorus in A will also release nitrogen, and this released nitrogen will be exported to Basin B. As B is nitrogen-limited (i.e. phosphorus-rich), eutrophication will increase in Basin B as a consequence of the actions undertaken by A. Similarly, if Country B reduces its nitrogen discharges in Basin B, phosphorus will flow into Basin A, exacerbating A's eutrophication problem.

As abatement of phosphorus by A increases eutrophication in B, Country B's best response is to reduce its nitrogen discharges further. But in doing so eutrophication is made worse in A, and A will therefore respond by reducing its phosphorus discharges even further. The process will continue until neither country can improve its welfare by abating discharges any further. This state defines the equilibrium in unilateral policies. As neither A nor B take into account the effect of their actions on the welfare of the other country, each is driven to abate its own Basin's limiting pollutant too much and the other Basin's limiting pollutant too little. As is typical of all equilibrium in unilateral policies, pollution of both Basins is excessive compared with the full co-operative outcome. However, it appears that the abatement of each Basin's limiting pollutant is also excessive in the equilibrium in unilateral policies (for more detail see Barrett, 1995) in contrast to every paper so far published in the literature. The policy implication is that full co-operation is the optional strategy, but one in which abatement effort is redistributed rather than merely increased overall. Since marginal costs increase with abatement effort this means that a small redistribution in abatement will lower total costs as well as total environmental damages.

Cost-effective abatement strategies

'Cost-effectiveness' is defined as achievement of one or several environmental targets at minimum costs. A condition for cost-effectiveness is that the marginal costs of all possible measures are equal. Marginal cost is defined as the increase in costs when, in this case, nutrient load to the Baltic Sea is decreased by 1 kg N or P. As long as the marginal costs are not equal it is always possible to obtain the same level of nutrient reductions at a lower cost by reducing the load via measures with relatively low costs and increasing the load by the same amount via measures with relatively high costs. To calculate cost-effective nutrient reductions to the Baltic Sea it is necessary to i) identify all possible measures, ii) quantify their impact on the Baltic Sea iii) calculate marginal costs for all measures.

The environmental impact of a certain reduction of nutrient load at the source is, *ceteris paribus*, determined by the location of the source. If the source is located some distance away from the coastal waters of the Baltic Sea, only a fraction of any reduction at the source is finally felt at the coast. The share of the source reduction that reaches the coast depends on the retention of the nutrient that may occur at various points between the source and the coast. This implies that, for a given marginal cost at the source, the marginal cost of coastal load reduction is higher than for remote sources with low impact on the coast. To calculate impacts of source-related measures, information on source location as well as on transportation of N and P is needed. No water and soil transport models exist for the drainage basin and so very simplified retention numbers are used.

Table 5.11 Marginal costs of different measures reducing the nitrogen load to the coast

Region	Agriculture SEK per kg per N reduction	Sewage treatment plants SEK per kg per N reduction	Atmospheric deposition SEK per kg per N reduction	Wetlands SEK per kg per N reduction
Sweden	20-242	24-72	135-9500	23
Finland	57-220	24-60	874-6187	66
Germany	20-122	24-60	210-3576	27
Denmark	23-200	24-60	544-3576	12
Poland	12-101	7-35	523-3412	10
Latvia	59-196	7-35	183-1195	20
Lithuania	72-208	7-35	254-1723	15
Estonia	55-192	7-35	153-1999	36
St Petersburg	43-236	7-35	353-1884	51
Kaliningrad	28-210	7-35	273-1593	43
Belgium			742-4184	
France			1507-9045	
Netherlands			562-7184	
Norway			475-3460	
UK			785-4855	

Note: SEK = Swedish Kronor

The abatement measures can be divided into three different classes: (i) reductions in the deposition of nutrients into the Baltic Sea and on land within the drainage basin; (ii) changes in land uses; (iii) creation of nutrient sinks to reduce the transports of nutrients to the Baltic Sea.

In principle, the cost of an abatement measure includes the cost at the emission source and the cost impacts on other sectors of the economy. In the following analysis only the abatement costs at the source are included, calculated by means of engineering methods and econometric techniques (see Gren *et al.,* 1995). The calculated marginal costs at the source for different abatement measures aimed at reducing nitrogen load in different regions (Table 5.11) indicate that increased nitrogen cleaning capacity at sewage treatment plants is a low-cost measure in all countries. Further low-cost measures include the reduction in use of nitrogen fertilisers, cultivation of catch crops and the construction of wetlands; whereas measures reducing air emissions are relatively expensive in all countries. The marginal costs of phosphorus reductions tend to be much higher than those for nitrogen (Table 5.12).

Table 5.12 Marginal costs of phosphorus reductions.

Region	Agriculture SEK per kg per P reduction	Sewage treatment plants SEK per kg per P reduction	Wetlands SEK per kg per P reduction
Sweden	155-6604	41-52	18232
Finland	225-6080	41-52	1748
Denmark	144-2610	41-68	1202
Germany	188-2964	41-68	899
Poland	114-2033	20-100	611
Estonia	282-5622	20-100	6090
Latvia	234-5662	20-100	1234
Lithuania	186-6696	20-100	964
St Petersburg	230-4314	20-100	823
Kaliningrad	338-4290	20-100	545

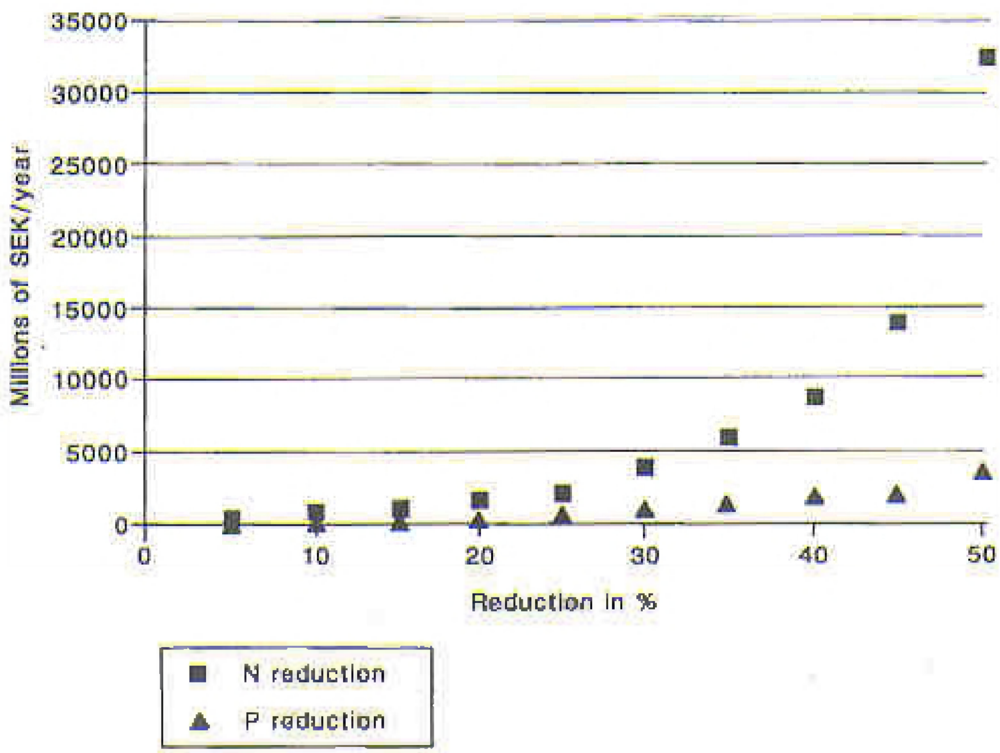
Minimum costs of nutrient reductions

Since reductions in nutrient loads to the Baltic Proper have the main impact on the ratio N/P, only minimum costs for load reductions to this basin are estimated. These are shown for various reductions in either N or P in Figure 5.8; the costs of reducing the load of nitrogen are much higher than the costs of corresponding decreases in phosphorus loads.

Several of the measures mentioned earlier, such as change of manure-spreading time and wetlands restoration, imply reductions in both nitrogen and phosphorus. When one of these measures is implemented with the aim of reducing the load of one nutrient, reductions are obtained in the other nutrient load ‘free of charge’. These joint impacts on several nutrients imply that abatement measures are relatively less costly if simultaneous reductions in N and P are undertaken. Total costs for various reduction levels are then lower for simultaneous decisions on N and P than for separate decisions, especially for abatement levels in excess of 40 percent reductions (see Gren, 1995 for more details).

Note that P is more 'mobile' than N and therefore requires more abatement effort, thus P can be the 'keystone' pollutant, i.e., if P is managed then so is N, but not usually vice versa.

Figure 5.8 Cost effective N and P reductions



It is important to emphasise that the cost estimates are based on various assumptions: i) biological: retention of nutrients, the leaching impact of agricultural measures and the nitrogen removal capacity of wetlands; ii) physical: feasibility limits of different measures such as nutrient cleaning capacity of sewage treatment plants; iii) economic: estimation of the costs of the various measures. Having performed a sensitivity analysis for an overall reduction of 50 percent in the load of both nutrients, the costs of both nitrogen and phosphorus reduction seem to be sensitive to assumptions of a biological character. Changes in the physical assumption about land available for agricultural measures often have a significant impact of the total costs. At other overall reduction levels, the costs may be sensitive to other types of assumptions.

In order to achieve a 50 percent reduction in nitrogen loading the most cost-effective mix of measures would be one in which agriculture, wetlands and sewage treatment plant-related measures account for 35 percent, 28 percent, and 31 percent respectively of the total nitrogen reduction. Measures involving air emissions account for 6 percent. The single most important country source in a cost-effective reduction strategy is Poland, which accounts for 40 percent of the total reduction (corresponding to about two thirds of the Polish load of nitrogen). Note that Poland, Russia and the Baltic states account for 72 percent of the total nitrogen reduction. The nitrogen reduction contribution of Swedish and Finnish regions amounts to only 8 percent and 7 percent respectively.

For phosphorus load reductions wetland measures can only play a minor role in coastal waters, and in fact it is only in Germany that such measures form part of a cost-effective abatement package. Instead, measures relating to sewage treatment plants are of major importance, accounting for 66 percent of the total reduction. This is a reflection of the relatively large load of phosphorus from households and industries and the availability of low cost abatement options. Again the single most important country source in a cost-effective reduction strategy is Poland, which accounts for 67 percent of the total reduction. The Baltic states, Poland and Russia together account for approximately 90 percent of the total phosphorus reduction.

Benefits valuation

The process of measuring the economic value of eutrophication damage in the Baltic involves three basic stages. Firstly, discharges of nutrients into the Baltic lead to eutrophication as outlined earlier and this leads to reductions in the various measures of environmental quality. Second, these changes in environmental quality lead to changes in the stream of services (use and non-use values) provided by the Baltic region. Third, the change in the stream of services will affect individual well-being and the economic proxy for well-being - monetary income, such that willingness to pay for the stream of services will change.

A concerted attempt was made to estimate the economic benefits of environmental improvements in the Baltic. Presented here are the results of two studies carried out in Poland and Sweden investigating the use and non-use value of reducing eutrophication to a sustainable level (a total of fourteen empirical valuation studies in three countries - Poland, Sweden and Lithuania - were carried out to look at benefit estimation issues; full details are presented in Georgiou *et al.*, 1995). The Polish and Swedish studies were used to estimate basin-wide benefits.

The first study was a contingent valuation study (CVM) focusing on Baltic Sea use and non-use values in Sweden. This study was designed as a mail survey; a questionnaire was sent to 600 randomly selected adult Swedes. The response rate was about 60 per cent, similar to other CVM mail questionnaire surveys that have been undertaken in Sweden (details including the questionnaire are presented in Söderqvist, 1995). It contained, *inter alia*, summary information on the causes and effects of eutrophication of the Baltic Sea. In the valuation scenario, the respondents were asked to assume that an action plan against eutrophication had been suggested, and that this action plan would imply that the eutrophication in 20 years would decrease to a level that the Baltic Sea could sustain. The types of action that this plan would involve were briefly described. It was also explained that the way to finance the actions would be to introduce an extra environmental tax in all countries around the Baltic Sea.

The respondents were then asked: 'If there were a referendum in Sweden about whether to launch the action plan or not, would you vote for or against the action plan if your environmental tax would amount to SEK X per year during 20 years?' Seven different amounts of money, X, were randomly used for the question. The answers to the question give an estimate of mean annual Willingness To Pay (WTP) of about 5900 SEK per person (or 3300 SEK assuming non-respondents to the survey have a zero willingness to pay).

It is likely that the respondents considered use values as well as non-use values when they answered the WTP question. This means that the WTP reflects perceived total benefits. However, note that there may be important differences between perceived benefits and real benefits. One reason for this is that the information communicated to the respondents about the eutrophication and its effects was far from complete. Moreover, the results from this CVM study may be influenced by embedding phenomena, i.e. that the respondents have also considered their WTP for other environmental improvements, and not only for a reduction of eutrophication. Embedding is a recognised problem in CVM studies. Note also that it is not easy to relate the outcome in the valuation scenario - a reduction of the eutrophication to a level that the Baltic Sea can sustain

- to a specific reduction of the nutrient load (though such an outcome is probably consistent with the 50 percent nutrient reduction target adopted by the Helsinki Commission). A time horizon of 20 years is reasonable in the sense that even if considerable action is taken today, any results will be evident only in many years' time. The description of the outcome as a 'sustainable' level reflects the fact emphasised by ecologists that actions against eutrophication will probably result in neither the complete disappearance of eutrophication, nor a return to the same ecological situation that characterised the Baltic Sea some decades ago, but rather to some new equilibrium.

The second study was almost identical to the first except that it was carried out in Poland, thus providing a direct international comparison to be made between the benefit estimates found in both countries. Again a mail questionnaire was used and 600 questionnaires were sent out to a random sample of Polish adults. The response rate was just above 50 percent, which was considered reasonable for this context and location. It was found that the level of support for the environmental tax was 54.9 percent. Mean annual WTP per person for the action plan was 840 SEK (or 426 SEK assuming non-respondents to the survey have a zero WTP).

To calculate basin-wide benefit estimates, the values for the different activities need to be added up, taking care not to double count, and using the relevant correct populations. Since there are benefit estimates available for the same valuation scenario in only two of the fourteen countries that are included in the Baltic drainage basin, any aggregation to the whole basin has to rely on strong assumptions. The aggregate benefit estimates presented below should thus not be taken too literally; but they provide useful information on the order of magnitude of basin-wide benefit estimates.

Table 5.13 shows estimates of aggregate benefits for the total economic value of a Baltic Sea nutrient reduction strategy. Data from the Polish and Swedish mail surveys are used since they are both concerned with total economic value (use and non-use value), and they contain the same valuation scenario. Given an adjustment for the difference in Gross Domestic Product (GDP) per capita levels between the countries, the Polish mean WTP estimate of 840 SEK (426 SEK) will be regarded as representative for the transition economies around the Baltic Sea, i.e., Estonia, Latvia, Lithuania, Poland, Russia; and the Swedish mean WTP estimate of 5900 SEK (3300 SEK) is taken as representative of the market economies of Finland, Germany, Norway and Sweden (Table 5.13). The possible WTP of the population in the other countries included in the Baltic drainage basin (Belarus, Czech Republic, Norway, Slovakia and Ukraine) will be ignored in this analysis.

In order to calculate national WTP estimates, the estimate per person was multiplied by the (adult) population in the Baltic drainage basin part of each country. According to Table 5.13, the basin-wide estimate for total economic value is MSEK69,310 per year (MSEK37,892 per year). This is a highly uncertain figure, but it indicates that the benefits from a Baltic Sea cleanup of eutrophication may be considerable.

Table 5.13 Basin-wide benefit estimates

Country	GDP *per capita* at PPP (US$)	Annual WTP per person[a] (SEK)	National WTP, year 1[b] (MSEK)	National WTP, present value[c] (MSEK)	National WTP, present value per year (MSEK)
<u>Transition</u>					
<u>economies</u>	3823	**700** (355)[d]	**790**	**8369** (4248)	**418** (212)
Estonia	3058	**569** (284)	(401)	**11653** (5816)	**583** (291)
Latvia	3632	**665** (337)	**1100**	**18465** (9355)	**923** (468)
Lithuania	4588	**840** (426)	(549)	**232623** (117974)	**11631** (5899)
Poland	4970	**909** (461)	**1743**	**69761** (35384)	**3488** (1769)
Russia			(883)		
			21958		
<u>Market Economies</u>	19306	**6770** (3790)	(11136)	**247529** (138570)	**12376** (6929)
Denmark	15483	**5430** (3040)	**6585**	**215980** (120920)	**10799** (6046)
Finland	18541	**6500** (3640)	(3340)	**167385** (93736)	**8369** (4687)
Germany	16821	**5900** (3300)		**414458** (231818)	**20723** (11591)
Sweden					
			23365		
			(13080)		
			20387		
			(11414)		
			15800		
			(8848)		
			39122		
			(21882)		
TOTAL			**130850** (71533)	**1386223** (757821)	**69310** (37892)

Notes:

a) For the transition economies, the Polish mean WTP estimate of SEK 840 (SEK 426) was multiplied by the ratio between each country's GDP *per capita* (at purchasing power parity) and Poland's GDP *per capita* at PPP. For the market economies, the Swedish mean WTP estimate of SEK 5900 (SEK 3300) and Sweden's GDP *per capita* at PPP were used correspondingly. Source of GDP data: OECD.

b) The annual mean WTP estimates per person multiplies by the (adult) population in the Baltic drainage basin part of the country (Sweitzer *et al.*, 1995, Statistical Yearbook, 1995). MSEK = millions of SEK.

c) Time horizon: 20 years. Discount rate: 7 percent (this rate was also used in the estimation of nutrient reduction costs.

d) Note: Figures in brackets are for benefit figures which assume zero WTP of non-respondents.

Table 5.14 brings together both the costs of pollution abatement and related economic benefit estimates in a cost-benefit analysis framework. It is clear that there are considerable net benefits available to a number of Baltic countries, sufficient for them to pay their own clean-up costs and subsidise the Baltic republics' abatement programme, while still gaining increased economic welfare benefits. While the economic benefit calculations are not precise point estimates they are indicative of the range or order of magnitude of clean-up benefits in the Baltic. Poland faces the largest cost burden because of its relatively high pollution loading contribution and the modest levels of effluent treatment that it currently has in place.

Table 5.14 Costs and benefits from reducing the nutrient load to the Baltic Sea by 50 percent, millions of SEK/year.

Country	% Reduction	Costs MSEK/year	Benefits MSEK/year	Net benefits MSEK/year
Sweden	42	5,300	20,723 (11,591)	15,423 (6291)
Finland	52	2,838	10,799 (6046)	7,961 (3208)
Denmark	51	2,962	12,376 (6929)	9,414 (3967)
Germany	39	4,010	8,369 (4687)	4,359 (677)
Poland	63	9,600	11,631 (5899)	1,761 (-3701)
Russia	44	586	3,488 (1769)	2,902 (1183)
Estonia	55	1,529	418 (212)	-1,111 (-1317)
Latvia	56	1,799	583 (291)	-1,216 (-1508)
Lithuania	55	2,446	923 (468)	-1,523 (-1978)
TOTAL	50	31,070	69,310 (37892)	38,240 (6,822)

Note: Figures in brackets are for benefit figures which assume zero WTP of non-respondents.

The costs in Table 5.14 refer to the allocation of nitrogen reductions that minimises total costs. The reductions, measured in percentages of original loads, vary between 39 percent (Germany) and 63 percent (Poland). If the abatement cost strategy was based not on a cost-effectiveness criterion linked to an overall ambient quality target, but on some 'political' solution based, for example, on uniform national load reductions then aggregate costs would be increased significantly (Table 5.15). This cost increase is due to the expensive measures that have to be implemented in Germany and Sweden. However, several countries with reduction levels exceeding 50 percent (Table 5.14) will gain from a country restriction as compared to a restriction of the total load of nitrogen. The costs presented in Table 5.14 may also be overestimates as they do not include other environmental improvements associated with these nutrient reductions (e.g. improved ground water quality and less acidification related to nitrogen oxides emissions) and land use changes which also yield other ecological services, such as wetlands providing food, biodiversity and flood-water buffering. If all these other positive aspects were included, some measures might imply internal net benefits instead of net costs.

The simulation results derived from modelling of nutrient transports in the Baltic Sea provide a proxy for the missing dose-response scientific data. The model simulates the impacts of nutrient reduction on the concentration ratios of N and P but does not provide any detailed information on the impacts on the biological conditions and production of ecological services. The available model does, however, predict that a 50 percent reduction in the loads of nitrogen and phosphorus to the Baltic Sea may correspond to the levels found during 1960s before the major deterioration in the Baltic environment occurred. This scenario is likely to be consistent with the one used in the CVM studies. Therefore, a crucial assumption when comparing costs and benefits is that 50 percent reductions in the loads of both nitrogen and phosphorus imply reaching ecological conditions which resemble those of the Baltic Sea prior to 1960s. Another important assumption concerns the nutrient filtering capacity of different Baltic Sea coasts, which is likely to vary greatly. Given that there are no appropriate data on the coasts' filtering capacity, distinction between different coastlines was not feasible. With all these qualifying assumptions, estimated costs and benefits of an overall reduction in the nutrient loads by 50 percent for different countries were calculated (Table 5.15).

Table 5.15 Cost change of a move from a 50 percent reduction in total load to 50 percent reduction in the load of each country, in percent.

Region	% Nitrogen reduction	% Phosphorus reduction
Sweden	-57.8	361.8
Finland	-51.9	718.1
Denmark	-48.8	29.1
Germany	543.6	32.1
Poland	-80.4	-57.9
Latvia	18.6	195.0
Lithuania	-17.5	289.2
Estonia	-13.4	277.1
St. Petersburg	981.8	-80.0
Kaliningrad	779.4	-81.6

Note: A negative sign implies cost savings when country restrictions are imposed as compared to reduction by 50 percent in the total load of nutrient.

Policy implications

There is considerable merit in the adoption of a basin-wide approach to pollution abatement policy in the Baltic and therefore in the implementation of an integrated coastal zone management strategy. It is clear that the ambient quality of the Baltic Sea is controlled by the co-evolution of both biophysical and socio-economic systems throughout the macro-scale drainage basin.

Despite the pioneering nature in the 'transition' economies of some of the economic benefits research, there seems to be little doubt that a cost-effective pollution abatement strategy roughly equivalent to the 50 percent nutrients reduction target adopted by the Helsinki Commission would generate significant positive net economic benefits. Monetary valuation of environmental benefits also indicated that the public's and experts' perception of environmental quality and quality decline are not necessarily synonymous.

A policy of uniform pollution reduction targets is neither environmentally nor economically optimal. Rather, what is required is a differentiated approach with abatement measures being concentrated on nutrient loads entering the Baltic proper from surrounding southern sub-drainage basins (the northern sub-drainage basins possess quite effective nutrient traps and contribute a much smaller proportionate impact on the Baltic's environmental quality state). The countries within whose national jurisdiction these southern sub-basins lie are also the biggest net economic gainers from the abatement strategy. This research indicates that the simultaneous reduction of both N and P loadings into the Baltic is more environmentally effective as well as cost-effective. The increased deployment of N-reduction and P-reduction measures within existing sewage effluent treatment works, combined with coastal wetland creation and restoration schemes and changes in agricultural practice, would seem to be a particularly cost-effective option set.

The marginal costs of nutrient reduction measures increase sharply towards the full works treatment end of the spectrum. This suggests that the greatest environmental and economic net benefits are to be gained by an abatement policy that is targeted on areas which lack treatment works of an acceptable standard, rather than on making further improvements to treatment facilities that already provide a relatively high standard of effluent treatment. Relating this to the importance of the spatial location of nutrient loading, suggests that nutrient reduction measures in the Polish and Russian coastal zone areas would be disproportionately effective. The financing of such measures remains problematic if only 'local' sources of finance are to be deployed. Non-commercial funding and bilateral agreements could play a vital role in the enabling process for an effective and economic Baltic clean-up programme.

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Appendix 1. LOICZ Typology

The current 'typology' plan of action involves the following conceptual structure:

1. The short-range objective is a system that makes it possible to globalise or extrapolate local and regional flux estimates derived from budget models or other sources by applying such results to other coastal reaches of the same or similar type.

2. The products should be as rigorous, versatile, and comprehensive as possible, but the twin constraints of time schedule and applicability to biogeochemical flux globalisation are paramount in the early stages of typology evolution.

3. The objective is the development and continued expansion of a number of data sets which could be used for testing typologies for coastal zone processes and compartments, and especially the estuarine and inner shelf portions of the coastal zone. Data sets directed towards typologies for the outer shelf and exchanges with the open ocean should be handled separately; extensive development - primarily terrestrial data sets - should be coordinated with other relevant programmes.

4. The objectives of identifying fluxes through, and transformations of, materials in the estuarine zone may be effectively achieved through the identification of data sets which might be used for three component sets of typologies, each of which would have the potential for including multiple typologic approaches:

   a) an '**input**' typology class representing primarily natural and anthropogenic fluxes from land and atmosphere into the estuarine zone;
   b) a '**transformation**' typology class characterising the biogeochemical reactions within the zone (e.g. net primary production, biomineralisation); and,
   c) an '**exchange**' typology class representing the exchange of material with the inner shelf - primarily the outer shelf and ocean.

5. The formulation of data sets for 'transformation' and 'exchange' typologies was relatively straightforward and depended on a reasonable number of primarily marine components. In contrast, the data for the 'input' typology deal primarily with terrestrial variables and their interactions, making its development more challenging.

There are three central issues in the typology process. One is the nature, appropriate scale, and potential problems with readily available digital databases of relevant environmental variables. A second point of concern is that the variables should to be sampled at the same spatial and temporal scales, so that there would inevitably be certain types of desirable statistical or modelling manipulations that should not be used with certain typology data sets. Third, the problem of defining the landward portion of the coastal zone in a practical fashion depends on, and is constrained by, both of the other issues, as well as by operational considerations.

One key question is whether to use a definition of the landward extreme of the coastal zone based on a simple topographic criterion, such as elevation, or to use a definition that incorporates some aspects of drainage basins. Although a drainage-basin approach is favoured in principle, the whole-basin approach would expand the definition of the coastal zone to full continental coverage and greatly increase data needs and processing requirements. The possible use of topographically defined coastal basins, or of coastline assignment to basins defined on the basis of divides between major river watersheds is a longer-term objective of the typology development process.

The initial data collection for the typology initiative has been undertaken on the basis of a coastal strip defined by the 50 m depth and elevation contours and the coastline. Although relatively arbitrary, this is considered conservative in terms of including the inner shelf, or ‘estuarine zone’, and the most relevant portion of the terrestrial coastal zone. This definition is also relatively quick and easy to implement for trial applications and to test against alternative topographic definitions. When combined with classified, as opposed to continuous, numerical values of environmental data, a reasonable number of the basic goals of typologic extrapolation or globalisation could be met without violating calculational principles.

The initial LOICZ Typology Data Set has been compiled for coastal cells on a 1° x 1° grid between the -50 m to +50 m global elevation generated from TerrainBase (NOAA, 1995).

Data for Input typology. The LOICZ Typology data set is under development and can be viewed and downloaded at **http://www.nioz.nl/loicz/projects/core/typo/**. The following list indicates both the desired data sets and those available (*) as of September 1998 (refer also to the full Typology Data Set breakdown available on page 96 from the LOICZ url).

- * Vegetation class (3.17);
- Land cover;
- * Soil type (3.11; 3.13);
- * Soil carbon content (3.13);
- * Soil texture (3.12);
- * Soil moisture (3.33);
- * Monthly precipitation and evaporation - mean and extremes;
- * Vegetation Index (NDVI);
- * Monthly temperature - mean and extremes (3.28, 3.29);
- Fertiliser (N and P) use;
- * Population density (CIESIN gridded data - http://www.ciesin.org/) (3.22); and,
- River discharge of fresh water (3.23), sediments and nutrients.

Data required for Transformation typology:
- * Coastal Zone Color Scanner (CZCS) data (SeaWifs - Feldman *et al.*, 1989) (3.34);
- * Sea Surface Temperature - mean and extremes w/months of occurrence (SeaWifs); and,
- * Monthly irradiance - mean and extremes w/months of occurrence (ISLSCP, 1996) (3.27).

Data required for Exchange typology:
- * Freshwater flow - monthly mean and extremes;
- * Tidal frequency (3.7);
- * Tidal magnitude (3.8);
- * Wind speed and direction (3.24, 3.25, 3.30);
- Coastal sinuosity; and,
- Areal extent.

From the integrated modelling/assessment perspective, a number of key issues now emerge:
- what other demographic and socio-economic data sets are available or could be constructed in order to improve the utility of the ‘input’ typology, by increasing its comprehensiveness to cover all significant environmental change pressures?;

- is it desirable and feasible to formulate a fourth component typology ‘human welfare’ characterising the initial spatial location and density of populations and their economic activities in juxtaposition to changing C, N & P flux situations and other climatic etc. change factors (perhaps along the lines of vulnerability indexes)?

Typology Data Set (Compiled and edited by M. van der Zijp).

This can be found at: **http://www.nioz.nl/loicz/projects/core/typo/frame1b.htm**

META DATA

3.1. Variable: Grid cell ID

3.2. Variable: Longitude and latitude

3.3. Variable: Country name, region and continent

3.4. Variable: Basin ID

3.5. Variable: Cell location ID

3.6. Variable: Wave height

3.7. Variable: Tidal type

3.8. Variable: Tidal range

3.9. Variable: Cultivation intensity

3.10. Variable: Methane

3.11. Variable: Soil type

3.12. Variable: Dominant soil texture

3.13. Variable: Soil carbon content

3.14. Variable: DSRF, Dunes, swamps and glaciers

3.15. Variable: Ecosystem

3.16. Variable: Coral

3.17. Variable: Vegetation class

3.18. Variable: Tropical forest destruction

3.19. Variable: Morphologic and tectonic classification

3.20. Variable: LGP

3.21. Variable: GNP

3.22. Variable: Population density

3.23. Variable: Runoff

3.24. Variable: Tropical storms

3.25. Variable: Winter gales

3.26. Variable: Precipitation

3.27. Variable: PAR

3.28. Variable: Dew point temperature

3.29. Variable: Mean air temperature

3.30. Variable: U-wind

3.31. Variable: NDVI

3.32. Variable: Surface temperature

3.33. Variable: Soil moisture

3.34. Variable: CZCS

3.35. Variable: Salinity

3.36. Variable: Ocean current

Appendix 2. The use of Input Output economic modelling for integration of environmental impacts

What is an IO model?

Input Output (IO) models are a representation of all the economic activity which takes place in a national economy based on flows of economic value (e.g. in dollars) between sectors. They are widely used by national economic planners to estimate the impacts of exogenous changes in the economic system on particular sectors, such as on the agricultural sector or the household sector, or on final demand and employment. The models are based on matrix tables where the non-leading diagonal elements make up the inter-sectoral flows. The data for these models are normally held by government statistical services with the major coefficients being re-estimated periodically through sectoral surveys, but perhaps only every decade. A major text on the IO approach is that of Miller and Blair (1985).

IO models can be extended in various ways such that they can potentially contribute to integrated modelling in coastal areas. Firstly, regional models can be calibrated such that the flow of goods and services is specified for a specific region of a country. Secondly, and most importantly for this purpose, a set of environmental coefficients can be developed such that flows of economic activity between sectors can be represented as flows of materials or pollutants. Early examples of this analysis include Ayres and Kneese (1969) who demonstrated that the production of environmental impacts from all sectors of the economy is pervasive and that increasing the overall scale of economic activity increases the sector-specific production of pollutants and other waste products, known in economics as externalities. The major limitations of the IO approach in general are in relating even a regional IO matrix to a particular coastal zone since the models themselves are not spatial in nature; and in the availability of data, particularly of the pollution coefficients from each sector. On this latter issue, the approach signifies the periodic flows of pollutants, where the coefficients can be estimated, but does not distinguish between those pollutants which are cumulative in the environment, such as many heavy metals and other substances which accumulate in coastal marine life, from those which are non-persistent. This issue highlights one of the general limitations of the IO modelling framework: that it is in general static, and has difficulty in handling both materials flows and technological change across time.

Despite these limitations, the IO modelling framework can be used in integrated modelling of coastal change by demonstrating the impact of scenarios of driving forces or pressures on the coastal zone on the state of these resources through loading of pollutants and other materials. The example given below comes from a regional IO model in the Philippines (Mendoza, 1994; Orbeta *et al.*, 1996) which is being utilised in conjunction with other models to examine, among other things, the impact of land-based environmental changes on the coastal environment of the Lingayen Gulf, under the SARCS/WOTRO/LOICZ project in the Philippines (contact Liana MacManus and Doug McGlone).

What are the basics of computation of IO models?

As outlined above, IO models are made up of matrices representing flows of goods in the economy. Matrix algebra forms a convenient shorthand for outlining how the computations are made, while the tables themselves are presented through this section, drawing on the regional IO developed in the Philippines.

An Input Output relationship for an economy can be expressed in matrix form as:

1) $\mathbf{X} = \mathbf{AX} + \mathbf{Y}$,

where $\mathbf{X}$ = an n x 1 vector of gross output, $[X_i]$, with X_i being the gross output from each production sector,
$\mathbf{A}$ = an n x n technical coefficient matrix, $[a_{ij}]$, with a_{ij} as defined above,
$\mathbf{Y}$ = an n x 1 vector of final demands, $[Y_i]$, with Y_i being total final demand for sector i.

Equation 1 relates supply ($\mathbf{X}$) to demand ($\mathbf{AX} + \mathbf{Y}$), where intermediate demand is now represented by the matrix $\mathbf{AX}$. Matrix manipulation of equation 1 yields:

2) $$\mathbf{X} = (\mathbf{I} - \mathbf{A})^{-1} \mathbf{Y},$$

where $\mathbf{I}$ is an identity matrix, and
$(\mathbf{I} - \mathbf{A})^{-1}$ is referred to as the Leontif inverse.

The elements of the Leontif inverse are known as output multipliers. Each row element indicates the value of the change of a sector's output due to a unit change in final demand for the sector's output. A low column sum reveals a weak sectoral interlinkage; otherwise, it shows a sector's strong dependence on the other sectors' output to meet a unit increase in final demand for its output. The sector with the largest multiplier provides the largest total impact on the economy.

One common use of the IO framework is to examine the effects of an exogenous change in final demands (for example, an increase in population that causes an increase in household demand). These effects are determined from the following:

3) $$d\mathbf{X} = (\mathbf{I} - \mathbf{A})^{-1} d\mathbf{Y},$$ where $d\mathbf{X}$ = changes in sectoral gross outputs,
$d\mathbf{Y}$ = projected changes in final demands.

Thus, if an increase in population were to double the final demand from the household sector, equation 3 could be used to determine the changes in production ($d\mathbf{X}$) necessary to supply this extra demand.

Changes in sectoral gross output may not be the only item of interest to policy-makers. For example, there are certain production outputs (such as residuals, or pollution) that are not included in typical demand categories. Consider the adjustment of the basic model for the inclusion of residuals generation. This adjustment may be captured in a vector of impact variables.

Start with a matrix of residual or direct impact coefficients $\mathbf{v} = [v_{kj}]$, where v_{kj} is the amount of pollution of type k generated per (currency unit)'s worth of industry j 's output. Thus, the total pollution associated with a given level of output ($\mathbf{V}$) is given by:

4) $$\mathbf{V} = \mathbf{v}\,\mathbf{X},$$

or total pollution = pollution per unit output times output. This approach assumes that each sector produces pollution in fixed proportion to its output.

Inserting equation 2 into equation 4 gives:

5) $$\mathbf{V} = [\,\mathbf{v}\,(\mathbf{I} - \mathbf{A})^{-1}\,]\,\mathbf{Y},$$

where the bracketed quantity is a matrix of total impact (residual) coefficients. An element of this matrix is the total residual production generated per (currency unit)'s worth of final demand.

Changes in impact variables due to a change in final demand may be found using:

6) $\mathbf{dV = v(I - A)^{-1} dY}$,

or, substituting from equation 6:

7) $\mathbf{dV = vdX}$

Equation 6 may be used to estimate changes in pollution emissions brought about by a projected change in final demands. Equation 7 may be used in the case of projected changes in gross outputs.

An application in the Philippines

Orbeta *et al.* (1996) applied the above methodology in a policy simulation study for the Philippines. This study was prepared for the Philippine Environmental and Natural Resource Accounting Project and applied the Environmental and Natural Resource Accounting Framework (Mendoza, 1994) to analyse the resource and environmental impacts of economic policies at a regional level. This framework included modification of an 11 sector by 11 sector inter-industry transactions table to account for income from non-marketed, nature-based household production and environmental variables. The 11 x 11 transactions table was also extended with the endogenisation of the household sector to account for the household consumption response to changes in labour income, which is dependent upon sectoral gross output. This procedure involves movement of the personal consumption expenditure (PCE) sector out of final demand and into the technically interrelated table. In addition, the compensation of employees component of the value added sector rows is moved into the technically interrelated table. Endogenisation of the household sector can be important, since a considerable amount of pollution is discharged by this sector (Mendoza, 1994).

For this regional analysis, a 1988 intraregional 11x11 inter-industry transactions table of the non-competitive type (i.e., inter-industry transactions are confined to the region and refer purely to regionally produced goods and services) was used to simulate the impacts of four alternative development scenarios and the regional growth targets espoused in the Medium-Term Development Plan of Southern Mindanao, Philippines, for the period 1993-1998. The intraregional inter-industry transactions table was derived using the national IO coefficients as a first approximation of the region's IO structure (Secretario, 1995). This procedure assumes that the production technology in the region is the same as that in the nation as a whole. The coefficients are made region-specific using the simple location quotient approach.

The regional inter-industry transactions table is provided in Table A2.1, which is an empirical example of **X** in equation 1, but in expanded form. Table A2.1 disaggregates the purchasing sector category into 'compensation of employees' (CE), 'other value added' (OVA), and 'imports' (M) sectors. Table A2.1 also disaggregates the final demand sector into personal consumption expenditures (PCE), government consumption expenditures (GCE), gross fixed capital formation (GFCF), changes of stocks (CS), exports, imports (M) and 'total productive input' (TPI).

Note that the Total Intermediate Demand column of Table A2.1 represents the vector of intermediate demands, **AX**, in equation 1. The column Total Output is the vector of gross outputs, **X**, in equation 1. The column Total Final Demand is the vector of final demand, **Y**, in equation 1.

To derive the '**A**' matrix of equation 1, each of the z_{ij} elements of Table A2.1 are divided by the appropriate column sums, X_j. The column sums X_j are provided in Table A2.1 by the Total Input (TI) row. It should be noted that the column sum X_j is the sum of all inputs; those of both the production and payments sectors. The resulting '**A**' matrix is provided in Table A2.2. Creation of the Leontif inverse matrix $\mathbf{(I - A)^{-1}}$ follows from derivation of the '**A**' matrix, and is given in Table A2.3.

The residual discharge coefficient matrix **v** is given in Table A2.4. This table provides discharge coefficients v_{kj} for air pollution (particulate matter, sulphur oxides, nitrogen oxides, volatile organic compounds, carbon monoxide), and water pollution (biological oxygen demand, suspended solids, total dissolved solids, oil, nitrogen, and phosphorus) for each production sector plus the endogenised household sector. Residual coefficients are measured in tonnes per thousand pesos of a sector's total output. These coefficients were derived from a variety of local sources in addition to the World Health Organisation's Rapid Assessment of Sources of Air, Water, and Land Pollution (WHO, 1993).

Estimates of water effluent and air emission discharges are presented in Table A2.5. This table represents the total pollution matrix **V** in equations 4 and 5. The discharges are determined by multiplying the total regional output for each sector by the corresponding residual coefficient, as in equation 4.

This basic framework can then be used to develop scenarios and estimate changes in the outputs of both monetary flows in the economy and environmental residuals. Among the policy simulations carried out in the ENRAP study, for example, was the evaluation of the projected impacts of four alternative development scenarios on gross output and the environment. These scenarios involve changes in final demand, and the resulting impacts on residual generations were therefore determined using equation 6. These changes in the state of the environment can therefore form a part of assessment under the Pressure State Impact Response framework. The IO analysis is therefore useful in integrated modelling in the coastal zone with the constraints, as outlined above, being primarily on the spatial downscaling and the availability of data, particularly relating to the environmental coefficients.

Table A2.1 Example of a modified 12 x 12 industry IO table (Region XI, The Philippines, 1988).

SECTOR	Modified (in '000 pesos) 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH	Total Intermediate Demand (+ Labour PCE)
1 agriculture	1,430,038	161	9,481	382	4,943,426	11,642	216	47	3,007	525	18,647	1,191,524	7,609,096
2 fisheries	2,589	301,646	396	1,075	426,468	4,138	36	4	1,285	3,496	17,383	433,634	1,192,150
3 forestry and hunting	197	0	382,811	8,875	1,028,264	2,666	8,723	930	113,037	1	13,478	166,380	1,725,362
4 mining and quarrying	4,624	1,776	901	11,226	15,605	331,407	98	79	92,551	3,306	18,391	41,821	521,785
5 manufacturing I	305,165	75,902	54,269	111,088	2,913,739	72,188	1,426	357	281,077	32,387	579,993	7,483,074	11,910,665
6 manufacturing II	230,259	65,551	56,590	228,250	232,868	347,868	6,358	1,521	307,598	189,475	208,922	167,214	2,024,474
7 electricity and gas	7,437	3,246	2,325	132	40,264	9,790	243	419	1,759	2,072	73,194	19,798	160,679
8 waterworks & supply	3,504	1,755	8	656	8,953	73	9	1	1,340	10,272	68,776	5,004	100,351
9 construction	10,397	10,454	297	31,616	20,785	3,178	934	20	498	4,703	126,082	4,643	213,607
10 transportation	158,642	41,181	23,816	40,230	238,190	43,460	108	271	65,607	116,353	583,696	646,094	1,957,648
11 other services	386,979	122,791	58,146	244,712	1,444,287	210,279	4,716	9,238	152,099	354,462	1,610,261	4,156,512	8,754,482
CE (HH)	5,839,675	480,807	932,063	305,350	1,128,466	174,401	20,613	20,338	443,887	307,364	4,662,733		14,315,697
TII Total Intermediate Inputs	2,539,831	624,463	589,040	678,242	11,312,849	1,036,689	22,867	12,887	1,019,858	717,052	3,318,823	14,315,697	36,188,298
M	1,532,830	588,241	316,697	1,155,305	3,227,324	1,524,355	33,931	11,918	1,108,113	986,231	3,037,189		13,522,134
CE	5,839,675	480,807	932,063	305,350	1,128,466	174,401	20,613	20,338	443,887	307,364	4,662,733		14,315,697
OVA	14,311,664	1,566,489	3,010,204	2,134,158	8,145,479	994,614	101,960	59,933	1,509,410	1,098,683	12,933,972		45,866,566
TPI	20,151,339	2,047,296	3,942,267	2,439,508	9,273,945	1,169,015	122,573	80,271	1,953,297	1,406,047	17,596,705		60,182,263
TI Total inputs	24,224,000	3,260,000	4,848,004	4,273,055	23,814,118	3,730,059	179,371	105,076	4,081,268	3,109,330	23,952,717	14,315,697	109,892,695

Table A2.1. Continued.

Final Demand Matrix						Total Final Demand (TFD)	Modified Total Output (TO)	Sector
PCE Original (Modified)	GCE	GFCF	CS	Exports	M			
2,316,478	0	123,001	(29,962)	15,396,911	0	17,806,428	24,222,726	1
843,041	0	5,092	(270)	1,653,621	0	2,501,484	3,253,416	2
323,464	0	8,781	(76,113)	3,032,890	0	3,289,022	4,848,004	3
81,305	0	0	(14,862)	3,726,648	0	3,793,091	4,273,055	4
14,548,074	0	112,255	(193,021)	4,919,219	0	19,386,527	23,814,118	5
325,085	0	469,505	(28,755)	1,088,964	0	1,854,799	3,730,059	6
38,490	0	0	0	0	0	38,490	179,371	7
9,729	0	0	0	0	0	9,729	105,076	8
9,026	0	1,816,774	0	2,046,504	0	3,872,304	4,081,268	9
1,256,091	0	48,720	0	492,965	0	1,797,776	3,109,330	10
8,080,803	765,658	512,542	0	9,995,744	0	19,354,747	23,952,717	11
							14,231,785	HH
27,831,586	765,658	3,096,670	(342,983)	42,353,466	(25,566,344)	73,704,397	109,800,926	Total
8,613,505	1,313,871	2,586,973	(470,139)	0	0	(13,522,134)	0	M
0	0	0	0	0	0	0	14,315,697	CE
0	0	0	0	0	0	0	45,866,566	OVA
0	0	0	0	0	0	0	60,182,263	TPI
36,445,091	2,079,529	5,683,643	(813,122)	42,353,466	(25,566,344)	60,182,263		TI

Note: 1. Modified TO is the value of total output adjusted for household production (forestry sector) and environmental damages (agriculture, fishery and household sector). Source: Orbeta *et al.* (1996)
TFD = PCE + GCE + GFCF + CS + E - M
TO = TID + TFD

Table A2.2 A' Matrix, ENRAP 12 x 12 industry IO table (Region XI, the Philippines, 1988).

SECTOR	Modified 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
1 agriculture	0.05903	0.00005	0.00196	0.00009	0.20758	0.00312	0.00120	0.00045	0.00074	0.00017	0.00078	0.08323
2 fishery	0.00011	0.09253	0.00008	0.00025	0.01791	0.00111	0.00020	0.00004	0.00031	0.00112	0.00073	0.03029
3 forestry and hunting	0.00001	0.00000	0.07896	0.00208	0.04318	0.00071	0.04863	0.00885	0.02770	0.00000	0.00056	0.01162
4 mining and quarrying	0.00019	0.00054	0.00019	0.00263	0.00066	0.08885	0.00055	0.00075	0.02268	0.00106	0.00077	0.00292
5 manufacturing I	0.01260	0.02328	0.01119	0.02600	0.12235	0.01935	0.00795	0.00340	0.06887	0.01042	0.02421	0.52272
6 manufacturing II	0.00951	0.02011	0.01167	0.05342	0.00978	0.09326	0.03545	0.01448	0.07537	0.06094	0.00872	0.01168
7 electricity and gas	0.00031	0.00100	0.00048	0.00003	0.00169	0.00262	0.00135	0.00399	0.00043	0.00067	0.00306	0.00138
8 waterworks and supply	0.00014	0.00054	0.00000	0.00015	0.00038	0.00002	0.00005	0.00001	0.00033	0.00330	0.00287	0.00035
9 construction	0.00043	0.00321	0.00006	0.00740	0.00087	0.00085	0.00521	0.00019	0.00012	0.00151	0.00526	0.00032
10 transportation	0.00655	0.01263	0.00491	0.00941	0.01000	0.01165	0.00060	0.00258	0.01608	0.03742	0.02437	0.04513
11 other services	0.01598	0.03767	0.01199	0.05727	0.06065	0.05637	0.02629	0.08792	0.03727	0.11400	0.06723	0.29035
CE (HH)	0.24107	0.14749	0.19226	0.07146	0.04739	0.04676	0.11492	0.19356	0.10876	0.09885	0.19466	0.00000

Source: Orbeta *et al.* (1996)

Table A2.3 Leontief Inverse matrix '(I-A)-1' for ENRAP 12 x 12 industry IO table (Region XI, the Philippines, 1988).

SECTOR	Modified 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
1 agriculture	1.13847	0.05642	0.06455	0.03396	0.29406	0.03265	0.04159	0.06168	0.05907	0.04195	0.06774	0.27307
2 fishery	0.01525	1.11276	0.01251	0.00644	0.03112	0.00662	0.00820	0.01247	0.01026	0.00951	0.01381	0.05594
3 forestry and hunting	0.01433	0.01095	1.09752	0.00911	0.06166	0.00694	0.06067	0.02140	0.04162	0.00818	0.01373	0.05108
4 mining and quarrying	0.00350	0.00466	0.00333	1.00923	0.00382	0.09982	0.00551	0.00423	0.03196	0.00886	0.00402	0.00817
5 manufacturing I	0.21334	0.16637	0.17600	0.10537	1.25864	0.09094	0.11427	0.16975	0.19152	0.12043	0.19493	0.74644
6 manufacturing II	0.02263	0.03414	0.02340	0.06591	0.02440	1.11459	0.04660	0.02643	0.09424	0.07786	0.02215	0.03920
7 electricity and gas	0.00169	0.00224	0.00165	0.00093	0.00308	0.00361	1.00226	0.00537	0.00169	0.00199	0.00445	0.00465
8 waterworks and supply	0.00086	0.00123	0.00058	0.00064	0.00113	0.00052	0.00048	1.00082	0.00094	0.00416	0.00372	0.00233
9 construction	0.00166	0.00463	0.00106	0.00832	0.00229	0.00249	0.00602	0.00163	1.00142	0.00298	0.00667	0.00395
10 transportation	0.02918	0.03093	0.02358	0.02057	0.02850	0.02334	0.01343	0.02288	0.03221	1.05444	0.04552	0.07971
11 other services	0.13958	0.13162	0.11366	0.11287	0.15086	0.11426	0.09442	0.19458	0.12078	0.19563	1.17773	0.44846
CE (HH)	0.32147	0.21931	0.26296	0.11622	0.18140	0.09907	0.16652	0.26505	0.17551	0.16674	0.26728	1.21828

Source: Orbeta *et al.* (1996)

Table A2.4 Matrix of residual coefficients for IO 12 x 12 matrix

Impact Variables	Sector 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
Residuals:												
PM	0.00001	0.00000	0.00008	0.00201	0.00045	0.00054	0.00044	0.00000	0.00053	0.00031	0.00002	0.00441
SO_x	0.00000	0.00008	0.00005	0.00092	0.00017	0.00025	0.00629	0.00000	0.00004	0.00019	0.00001	0.00002
NOx	0.00001	0.00015	0.00010	0.00056	0.00012	0.00017	0.00109	0.00000	0.00010	0.00028	0.00002	0.00016
VOC	0.00002	0.00005	0.00010	0.00045	0.00010	0.00011	0.00003	0.00001	0.00010	0.00047	0.00006	0.00690
CO	0.00011	0.00014	0.00058	0.00270	0.00060	0.00061	0.00011	0.00001	0.00053	0.00137	0.00013	0.03265
BOD5	0.00878	0.00000	0.07153	0.00000	0.00039	0.00007	0.00000	0.00000	0.00000	0.00000	0.00284	0.01196
SS	0.92180	0.00000	14.19494	1.86304	0.00038	0.00009	0.00541	0.00000	0.00000	0.00000	0.00300	0.00547
TDS	0.00000	0.00000	0.00000	0.00000	0.00281	0.00013	0.00002	0.00000	0.00000	0.00000	0.00000	0.00000
OIL	0.00000	0.00000	0.00000	0.00000	0.00003	0.00001	0.00000	0.00000	0.00000	0.00000	0.00014	0.00000
N	0.00482	0.00000	0.05502	0.00000	0.00001	0.00000	0.00000	0.00000	0.00000	0.00000	0.00010	0.00096
P	0.00005	0.00000	0.00087	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00003	0.00039

Source: Orbeta *et al.* (1996)

Table A2.5 Estimated matrix of residual discharges

Impact Variable	Sector 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
PM	314	0	400	8,606	10,725	2,021	78	0	2,167	970	505	63,095
SO_x	88	259	242	3,941	4,026	924	1,129	0	182	589	252	346
NO_x	243	479	505	2,407	2,890	641	195	0	393	857	591	2,291
VOC	404	171	464	1,942	2,326	392	5	1	395	1,473	1,522	98,747
CO	2,744	445	2,792	11,558	14,378	2,283	19	1	2,176	4,263	3,210	467,368
BOD5	212,593	0	346,766	0	9,274	244	0	0	0	0	68,038	171,220
SS	22,329,619	0	68,817,134	7,960,853	9,111	345	971	0	0	0	71,780	78,353
TDS	0	0	0	0	67,003	501	3	0	0	0	0	0
OIL	0	0	0	0	703	40	0	0	0	0	3,290	0
N	116,814	0	266,743	0	145	8	0	0	0	0	2,383	13,805
P	1,322	0	4,214	0	0	0	0	0	0	0	631	5,521

Source: Orbeta *et al.* (1996)

Regional Input-Output model linkages to regional flux budget: applications in Merbok, Malaysia.

The basis of the I-O model to be applied to the Merbok site in Malaysia is to bring all energy into a standard form - i.e., the common currency of carbon. But, unlike the traditional multiplier analysis using the Leontief inverse, we adapted Johansen's (1974) multisector model by introducing and environmental capital component into the equation system. Following the suggestion made by Pearce and Turner (1990, p.153), the usual Cobb-Douglas production equation also included in the system was modified into a dichotomy between market inputs (a combination of labour and machines) and environmental inputs rather than, as usually is the case, between labour and capital.

This Johansen framework was presented at our October 1996 meeting in Hanoi and further debate led to the development of carbon budgets by assessing the flux embedded in the respective ecosystems which constitute our individual study areas. Carbon has an energy equivalent measured in calories or joule which makes it appealing as we attempt to examine how energy is transformed from one form into another as we undertake economic production and whether the rate of this transformation *vis-à-vis* the carbon flux is sustainable over the long term.

The problem is therefore how best to incorporate carbon fluxes into an economic-ecological framework thereby bridging the gap between the ecology in energy equivalent terms with economic behaviour operating in the market system. Input-output systems have much potential for such an application. The critical issue is to select a suitable unit of measurement (a numeraire) with which to capture energy flows.

Energy equivalents and economics

The survey by Faucheux and Pillet (1994) indicated three main views on energy valuation. The first involves estimating the ratio of energy to money (see Odum and Odum, 1981, p. 44) so that we can measure money in energy terms or vice-versa. This view is a misconception because energy does not have the same properties that money has. It is a mistake to think that energy and money are convertible from one to the other. Money can be transformed from one form of asset into another and back again. Fluctuations in money values encountered in the conversion process are not due to transformation losses as happens for energy due to thermodynamic laws but according to changing market demand and supply conditions.

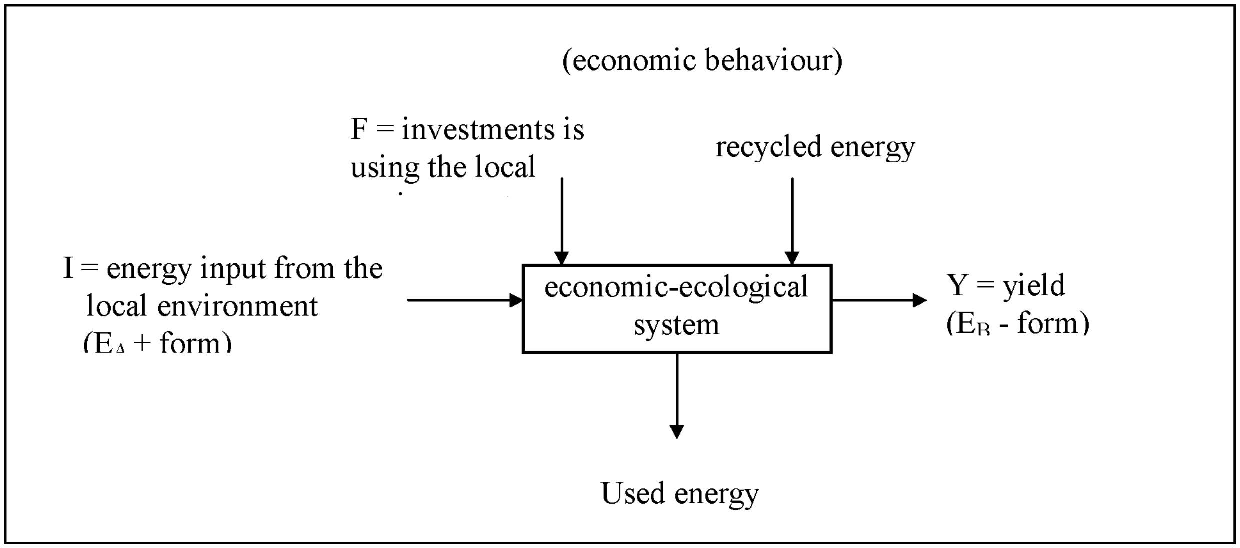
The second view concerns energy theories of value that attempt to attribute labour, materials, capital and all other production factors into energy terms. The limitation of this approach is that when we lose sight of the money values for these items we also lose sight of the price signals that affect how these items are brought into play within the production process. Thus, while an accounting of energy within the ecosystem is a useful inventory exercise, it will not help much when we wish to incorporate economic considerations that impact on the ecosystem.

The third view leaves energy and money as distinct entities and does not attempt to replace one by the other, but attempts to relate them. Economic activities are seen as a continuous transformation of low entropy energy sources into high entropy and in the process emits irreversible waste. Responding to this transformation of energy, composite indicators are developed that show to what extent a threshold is drawing near, beyond which the ecosystem will undergo a major change. The next section will discuss details of this view.

Energy metrics

The most effective way to make an assessment of the energy fluxes found in various forms within a local ecosystem is in terms of the solar energy that was used to produce them. All energy forms found are thus standardised in relation to solar energy, which is the embodied energy denoted as eMergy contained in the various forms of energy. The principle of this approach initiated by Odum (1983) is illustrated by Figure A2.1.

Figure A2.1. eMergy-energy relationship



As $E_A{}^{+}$, which is the embodied energy (or eMergy), is transformed through the process of economic activities into another form of energy $E_{B,}$ we obtain an eMergetic balance by the ration $E_B/E_A{}^{+}$ expressed in joules by solar joules or emjoules. This ratio defines the solar transformity of $E_B{}^{-}$ telling us the amount of $E_A{}^{+}$ incorporated in $E_B{}^{-}$. Both the first and second laws of thermodynamics are thus taken into account with respect to energy transformation and losses. The degree of solar transformity thus serves as a qualitative description of the ecosystem being assessed. The biomass of the local ecosystem expressed in $E_A{}^{+}$ emjoules indicate the amount of solar energy that had gone into generating this ecosystem.

To attempt bridging what we know of the energy state of the ecosystem, in terms of the degree of transformity, with economic production another term called monergy is introduced (see Pillet, 1994).

$$\text{monergy} = \frac{\text{eMergy}^{\text{nation}}\,(\text{emjoule})}{\text{GDP}\ (\$)}$$

This is a macroeconomic indicator which relates the total energy state of the country, that is eMergy in emjoules against the total economic production of the country in dollars.

Our intention is to estimate the unknown ecological price for a given hectare of land, say located within our study area, for a given year. If we assert that this price, $P_I(\$)$, in proportion to the country's total income, GDP(4), is exactly equal to the proportion of the energy inventory of that hectare of land to the total energy state of the country, that is:

$$\frac{P_I(\$)}{\text{GDP}(\$)} = \frac{\text{eMergy}^{\text{local}}(\text{emjoule})}{\text{eMergy}^{\text{nation}}(\text{emjoule})}$$

then, we can obtain an estimate of $P_I(\$)$ as follows:

$$P_I(\$) = \frac{GDP(\$).\ eMergy^{local}(emjoule)}{eMergy^{nation}(emjoule)}$$

$$P_I(\$) = eMergy^{local}(emjoule).\ \frac{GDP\ (\$)}{eMergy^{nation}}(emjoule)$$

$$P_I(\$) = eMergy^{\ local}(emjoule).\quad \frac{1}{monergy}$$

In other words if we can separately estimate the monergy of the country and if we perform an energy inventory of the local ecosystem in eMergy terms, we will be able to estimate the price of the local ecosystem, $P_I(\$)$.

The economy-cum-energy input-output model

The standard input-output model is established in the following way. Consider an n x n matrix $Z=\{z_{ij}\}$ of inter-industry flows expressed in millions of ringgit. Such flows only account for intermediate demands, i.e., purchases of industry outputs to be used as inputs into further production. Total output by the economy is an nx 1 vector $X=\{x_j\}$ obtained after adding final demands $Y=\{y_i\}$. In other words,

$$Z + Y = X$$

Analysis begins by calculating the intermediate inputs per dollar of output for each of the elements of Z to form a technical coefficients matrix $A = \{_{ij}\}$that is,

$$A = a_{ij} = \frac{Z_{ij}}{X_i}$$

Since Z = AX, we have

$$AX + Y = Y$$
$$Y = (I - A)\ X$$

and therefore,

$$X = (I - A)^{-1}Y$$

This equation is called the Leontief inverse. It forecasts the level of economic activity given by the direct, indirect and induced economic impacts, X, for the different industry sectors given some assumptions or scenarios of the pattern of final demand Y under a given set of technological structures fixed by the technical coefficients set $(I-A)^{-1}$.

There are many versions of economic-cum-energy variation of this model. The one which was built for analysis in this paper is based on Miller and Blair (1985). It used revised form so matrices we will call Z^*, Y^* and X^* in which are contained energy flows in energy units alongside industry flows in ringgit. In addition a diagonal matrix of total energy consumption, F^* is established.

The following are defined

$$\delta = F^* (X^*)^{-1} A^*$$

$$\delta = F^* (X^*)^{-1} (I - A)^{-1*}$$

Here , X^* is diagonalised matrix of the otherwise n x 1 vector containing both energy and non-energy sectors to facilitate matrix multiplication. The resulting matrix indicated by δ shows the direct energy intensities by sectors. The matrix α shows the total energy intensities which incorporate secondary impacts made up of indirect and induced effects.

The matrices α and δ contain values identical to the A^* and $(I-A^*)^{-1}$ matrices respectively except that pre-multiplication by F^* and $(X^*)^{-1}$ removes the inter-industry money flows. Such flows are irrelevant here because they should be analysed under standard input-output analysis.

Regional tables expand on the national table by recording flows between sectors and between regions (see Miller and Blair, 1985). To simplify the regional table, concern is only given to flows inside the region; flows with the rest of the country are considered as another composite region.

Incorporating the environment into the input-output framework is complicated by the need to introduce an elaborate set of environmental sectors which have indicated flows among themselves and among these environmental sectors with the various economic sectors. Furthermore there is the need to resolve the units of measurement for the environmental sectors.

Within the eMergy concept, the environmental component needed on the input-output table is reduced to one sector flowing out as eMergy, E_A^+ and becomes energy, E_B^-. A sketch of the input-output framework is shown in Figure A2.2.

Figure A2.2. Economic-eMergetic input-output table framework.

	local economic sectors	E_B - local	economic sectors in rest of country	E_B - rest
local economic sectors	A	B	C	D
E_A + local	E	F	G	H
economic sectors in rest of country	I	J	K	L
E_A + rest	M	N	O	P

In figure A2.2, the usual inter-industry flows within the locality are entered into A and the economic investments into the energy transformation process of the local environment (referred to as F in the figure) goes into B. In C and D economic inputs affecting the rest of the country are entered. E contains data on eMergy inputs into economic production in the locality while F records transformation losses involved from E^+_A to E^-_B. Again, G and H are meant for interactions from the locality to the rest of the country. The remaining parts of the table contain similar inputs but this time dealing either with flows within the rest of the country or from the rest of the country into the locality.

Input-Output Coefficients

The first step to input-output analysis is to transform the above table format into what is called a technical coefficients table by dividing the column entries by gross economic output in dollar terms. The resulting entries become input-output flows per dollar of gross output. Notice that wherever the nominators are in dollars, we obtain the usual input-output coefficients. Wherever the nominators are in eMergy terms, the coefficients become monergy values. Thus from equations described above, environmental prices of the local ecosystem can be expressed as its total eMergy divided by monergy values on the coefficients table.

Beyond such descriptive indicators, standard input-output analysis procedures can be introduced from which we obtain secondary and induced impacts based on the Leontief inverse and the interconnectedness between input and output sectors based on Rasmussen's power and sensitivity indices.

Appendix 3. Monetary Valuation Methods and Techniques

Alternative and appropriate methods

The state to impact and impact to policy response model linkages require that ecosystem changes with direct or indirect effects on human welfare (i.e. well-being in terms of income and wealth creation and quality of life, including health effects) be evaluated in order to determine their magnitude and significance. Monetary valuation methods and techniques provide one approach to the evaluation of impacts exercise. They can be deployed in any of the three resource assessment categories (impact analysis, partial valuation and total valuation) defined in section 3.4.

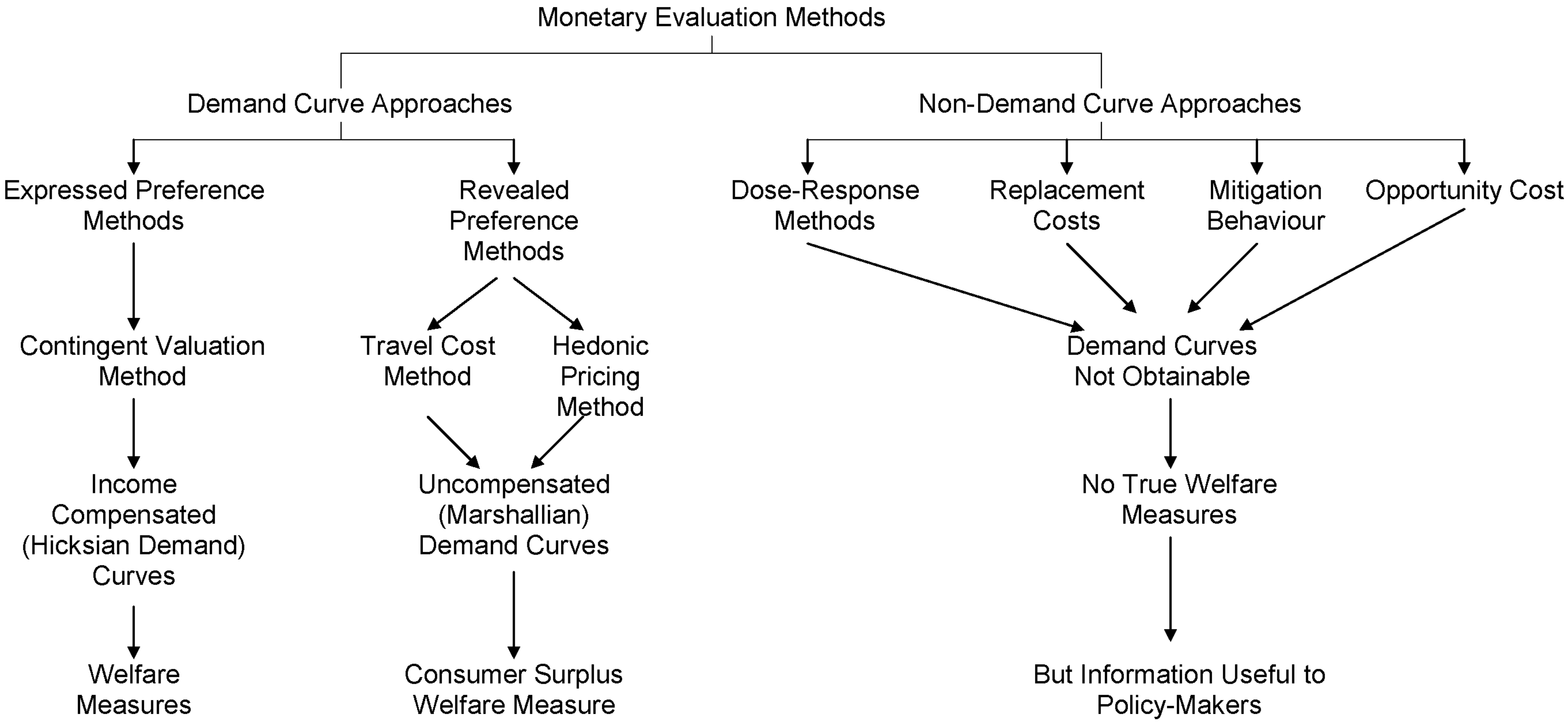
The environmental effects that require valuation can be classified into productivity changes, health effects, amenity gains and losses and assets existence value conservation or loss. Each of these effects is amenable to particular valuation methods as discussed in section 3.4 (see Table 3.8). A general survey of these valuation methods, plus some outline case study examples that have utilised such methods is provided in Turner and Adger (1996). In this appendix the methods themselves are reviewed in more detail and the text is supported by relevant references to empirical studies.

As reported in Turner and Adger (1996) it is possible to divide the monetary valuation methods into demand-curve approaches and non-demand curve approaches, as outlined in Figure A3.1. The former are more strictly valid in economic theory terms, but the latter are sometimes the only option because of data deficiencies and conceptual problems. While non-demand curve approaches are not capable of providing valid economic welfare estimates, they do provide useful monetised information on impact significance and are widely used in representing the relative importance of many environmental goods and services (see Dixon and Sherman, 1990 for a review). So, for example, the replacement cost of the loss of soil due to erosion from agricultural land may involve the market cost of fertilisers to replace the soil fertility so that productivity is maintained. But this replacement cost estimate does not reflect the demand for soil fertility by the farmer and is somewhat hypothetical. The cost to the farmer is best represented as the actual value of the loss of production of agricultural outputs from the less fertile land, rather than the hypothetical cost of replacement of soil. The following sections first outline the non-demand curve approaches and then concentrate on methods within the demand curve approaches to valuation, namely the travel cost method, the hedonic pricing method and the contingent valuation method. Further information on these can be found in Bateman (1993), Bateman and Turner (1993), Barbier (1994), Turner *et al.* (1994) and other texts and examples of applications of these in Adger *et al.* (1995), who attempt to aggregate total economic value for forest resources, and other examples highlighted in the text.

Non-demand curve approaches

The opportunity cost method quantifies what society has to give up if an environmental resource such as a wetland or a coral reef is to be conserved. An estimate of the monetary costs of the conservation option (the social opportunity cost) is made in terms of the alternative development option that is given up e.g. alternative uses for the wetland as drained farm land, or industrial, housing, or port facilities. The development option is assessed in economic terms in order to determine what net economic benefit (if any) society would have to give up when deciding to favour the environmental asset conservation option.

Figure A3.1 Demand curve and non-demand curve methods for the monetary evaluation of the environment



The replacement cost method examines the functions that a given environmental system provides, when it is operating in a 'normal' and 'healthy' state. It might then be possible to estimate what it would cost society if the system was lost or damaged, in terms of replacing some of the functions. A coastal wetland might, for example, be providing a storm buffering function and a nutrient sink function. If the wetland was converted to some industrial or other use, society would need to invest in a replacement sea defence system (or an augmented existing system) and perhaps a sewage treatment plant.

Another variant of this approach involves estimating the costs of so-called shadow projects. Thus it might be possible to re-create the threatened wetland elsewhere in the same general area, or to restore existing nearby but already degraded wetlands. The costs of these shadow project options would then need to be estimated and fed into the policy process.

Statistical techniques can sometimes be used to relate differing levels of pollution (the 'dose') to differing levels of damage (the 'response'). They are known as dose-response functions. Applications in the coastal zone context would include fisheries and coral reef damage from pollution and water quality-related human health damage effects. Many of these techniques are dealt with in Dixon and Sherman (1990) and Turner *et al.* (1994).

Travel Cost Method (TCM)

The TCM evaluates the recreational use value of resources, hence measures one aspect of indirect use values. The Travel Cost Method is a survey technique, whereby visitors to a site are asked a series of questions to ascertain their place of residence; necessary socio-economic information; frequency of visits to the particular and other similar substitute sites; means of travel; and cost information about the trip. From these data visit costs can be calculated and related to visit frequency so that a demand function can then be used to estimate the recreation value of the whole site.

The method was developed in the 1960s in the US for estimating the value of outdoor recreation, particularly as information for management of national parks and other assets. The method is somewhat restricted in the range of impacts and changes in which it can provide economic values, but it is of use in the estimation of value in coastal environments where recreational use of beaches and other resources represents a significant demand. Examples of travel cost estimates include those for beach resorts, where the quality of the beach affects demand, hence the environmental quality has a marginal value (Bell and Leeworthy, 1990). Various estimates of recreational value of forests and non-coastal resources exist including Tobias and Mendelsohn (1991), Maille and Mendelsohn (1993) and Mercer *et al.* (1995) which all investigate the recreational value of forest resources in the tropics.

The value for a specific recreation site is estimated under this method by relating demand for that site (measured as site visits) to its price (measured as the costs of a visit). A simple TCM model can be defined by a trip-generation function such as:

$$V = f(C, X)$$

where V = visits to a site
C = visit costs
X = other socio-economic variables which significantly explain V.

The literature can be divided into two basic variants of this model according to the particular definition of the dependent variable V. The 'Individual Travel Cost Method' (ITCM) simply defines the dependent variable as the number of site visits made by each visitor over a specific period, say one year. The Zonal

Travel Cost Method (ZTCM) on the other hand, partitions the entire area from which visitors originate into a set of visitor zones and then defines the dependent variable as the visitor rate (i.e., the number of visits made from a particular zone in a period divided by the population of that zone).

The ZTCM approach redefines the a trip-generation function as:

$$V_{hj}/N_h = f\,(C_h, X_h)$$

where V_{hj} = Visits from zone h to site j
N_h = Population of zone h
C_h = Visit costs from zone h to site j
X_h = Socio-economic explanatory variables in zone h

The visitor rate, V_{hj}/N_h, is often calculated as visits per 1,000 population in zone h.

The underlying theory of the TCM is presented with reference to the zonal variant, and discussion of the differences between this and the individual variant is presented subsequently before consideration of more general issues. Discussion of the ZTCM is illustrated by reference to a constructed example detailed in Table A3.1 which estimates the recreation value of a hypothetical site. The method proceeds in nine steps as follows:

Step 1 Data on the number of visits made by households in a period (say annually) and their origin is collected via on-site surveys.

Step 2 The area encompassing all visitor origins is subdivided into zones of increasing travel cost (column 1 of Table A3.1) and the total population (number of households) in each zone noted (column 2).

Step 3 Household visits per zone (column 3) is calculated by allocating sampled household visits to their relevant zone of origin.

Step 4 The household average visit rate in each zone (column 4) is calculated by dividing the number of household visits in each zone (column 3) by the zonal population (number of households; column 2). Note that this will often not be a whole number and commonly less than one.

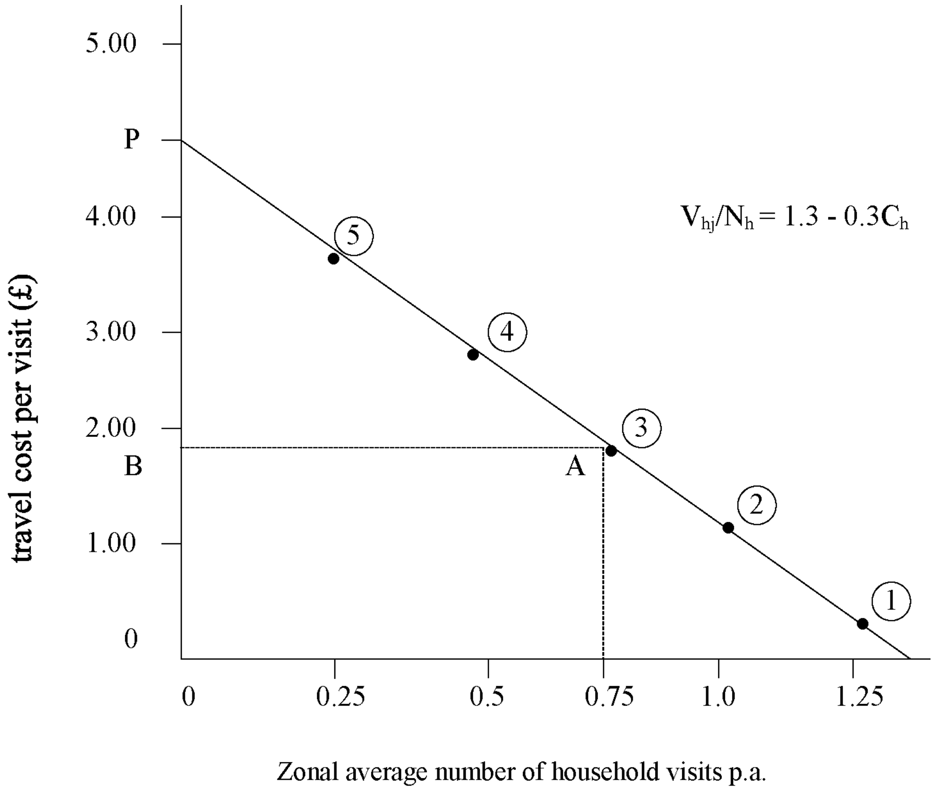
Step 5 The zonal average cost of a visit (column 5) is calculated with reference to the distance from the trip origin to the site.

Table A3.1 Worked example of consumer surplus estimates for recreation experience using zonal travel cost method

Column No.	1	2	3	4	5	6	7	8
	Zone No.	Zonal population (no. of households)[1] (N_h)	No. of household visits to site p.a.[2] (V_{nj})	Average no. of visits per household p.a.[3] (V_{nj}/N_h)	Average travel cost per household visit[4] (£) (C_h)	Consumer surplus per household all visits p.a. (£)	Consumer surplus per household per visit (£)	Total consumer surplus p.a. (£)
	1	10,000	12,500	1.25	0.16	2.60	2.08	26,040
	2	30,000	30,000	1.00	1.00	1.67	1.67	50,100
	3	10,000	7,500	0.75	1.83	0.94	1.25	9,400
	4	5,000	2,500	0.50	2.66	0.42	0.84	2,100
	5	10,000	2,500	0.25	3.50	0.10	0.40	1,000
				Total annual consumer surplus of the recreational experience			=	88,000

Notes: Trip generating function $V_{hj}/N_h = 1.3\text{-}0.3C_h$.
1. from census records.
2. from survey; annual totals derived by extrapolating from sample data according to available information regarding tourism rates.
3. column 4 = column 3/column 2.
4. either calculated with reference to zonal distance or via survey .

Figure A3.2 Demand curve for the whole recreation experience



Key: 1 = zone number 1.

Step 6 A demand curve is then fitted relating the zonal average price of a trip (travel cost) to the zonal average number of visits per household. This curve estimates demand for the 'whole recreation experience' rather than just the time spent on-site. In our hypothetical example this demand is explained purely by visit cost and the curve has the (unlikely) linear form given by:

$$V_{hj}/N_j = 1.3 - 0.3\ C_h$$

where V_{hj}/N_j = visit rate (average number of visits per household) from each zone
C_h = visit costs from each zone

Figure A3.2 illustrates this particular whole recreation experience demand curve. The estimation of this curve involves the implicit assumption that households in all distance zones react in a similar manner to visit costs. They would all make the same number of trips if faced with the same costs i.e. they are assumed to have identical tastes regarding the site.

Step 7 In each zone the household consumer surplus for all visits to the site (column 6) is calculated by integrating the demand curve between the price (cost) of visits actually made from each zone and that price at which the visitor rate would fall to zero (i.e. the vertical intercept of the demand curve at point P in Figure A3.2). Households in zone 3 for example would have a consumer surplus equal to area ABP for all their trips to the site:

$$\text{Consumer surplus for zone 3} = \int_{C_h=B}^{P} (1.3 - 0.3C_h).dC_h$$

Step 8 In order that annual total consumer surplus for the whole recreation experience can be estimated in each zone, total household consumer surplus must firstly be divided by the zonal average number of visits made by each household to obtain the zonal average consumer surplus per household visit (column 7). This can then be multiplied by the zonal average number of visits per annum (column 3) to obtain annual zonal consumer surplus (column 8).

Step 9 Cumulative annual zonal consumer surplus (column 8) across all zones gives our estimate of total consumer surplus per annum for the whole recreational experience of visiting the site.

These steps, leading to consumer surplus estimates, give a value of the recreational experience. There are various caveats to this value being used directly for any coastal resource. These caveats include whether there are substitute sites, whether the visitors are valuing particular attributes of the site; and whether the visitation rate and distance can be taken as an indication of recreational value (see Bateman, 1993). Given these caveats, the method is useful for determining recreational value of coastal resources. It is, however, ultimately limited in the scope of environmental impacts which can be addressed.

Hedonic Pricing Method

The Hedonic Pricing Method (HPM) relies upon the assumption that the local environmental quality (or lack of it) will determine the price of property and that differences in these prices can be taken as an indicator of marginal value of environmental change. The environmental factors, however, are only a subset of property price determinants which, for residential houses, may include amongst other factors the number of rooms and accessibility to shops and workplaces. The general specification of a hedonic price model is therefore:

HOUSE PRICE = f (ROOMS, ACCESS, ENVIRONMENT)

The equation states that house price is a function of (f) the number of rooms in the house (ROOMS), the distance in miles to local facilities from the house (ACCESS) and some measure of local environmental quality (ENVIRONMENT). If we were interested in valuing the environmental impact of local traffic noise then we could measure this in terms of decibels of traffic noise inside the houses in question.

We then need to measure each of the items HOUSE PRICE, ROOMS, ACCESS and ENVIRONMENT for a large number of houses so that we can begin to see how, on average, house price changes when each of the influencing factors change. We would expect house price to rise as the number of rooms increase; that house price would fall as the distance to local facilities rises, and finally, for house price to fall as the traffic noise increased, i.e. a typical demand curve relationship. This is indeed the results obtained in an US study of road noise. The following table (A3.2) shows the average percentage fall in house price which corresponded to a one unit increase in traffic noise in a number of US areas.

As an example, if a new road scheme was likely to raise traffic noise by one unit in Washington DC, then a monetary value for this increased noise pollution could be found by taking 88 percent of average house prices in the affected area.

The hedonic method has also been applied to the impact of water frontage, amenity, and other quality factors on house prices. This technique has been utilised in coastal areas to examine the impact of proximity to beach property, and hence derive a value for that environmental good (see Brown and Pollakowski, 1977). For a summary of the methods and applications see Brookshire *et al.* (1982) Garrod and Willis (1992), Freeman (1979) and Bateman (1993). Again this method, as with the travel cost method, is limited in its applicability to valuing the impacts of environmental change in coastal zones. The method is data-dependent and can only reasonably be applied where the environmental asset under consideration is well understood within the purchasing decisions of house or property owners. However, it can give some estimates of both the availability of recreational assets, as well as the impact of risk of inundation or flooding in coastal areas.

Table A3.2 The impact of traffic noise on house prices in the US

City	% fall in house price due to a one unit increase in noise
North Virginia	0.15
Tidewater	0.14
North Springfield	0.18 - 0.50
Towson	0.54
Washington DC	0.88
Kingsgate	0.48
North King Country	0.40
Spokane	0.08
Chicago	0.65

Source: Nelson (1978)

Note: Traffic noise measured as the equivalent continuous sound level (in decibels) which would have the same sound energy over a given period as the actual fluctuation sound level measured at houses in the study.

Contingent Valuation Method

The Contingent Valuation Method (CVM) is a method for placing monetary values upon assets and impacts which do not have market prices. It achieves this by constructing a hypothetical market and asking individuals, for example, what they are willing-to-pay (WTP) towards preservation of a particular environmental good. Therefore CVM relies upon individuals' expressed preferences (rather than the revealed preferences indicated by market prices).

The advantages include that the method allows us to ask questions about and estimate both use and non-use values (see Figure 3.7) and provides direct Hicksian welfare measures, which overcome some of the problems with non-demand curve approaches to valuation, as discussed above. The disadvantages of the method are that respondents may not believe in the credibility of the hypothetical markets; and that without an actual market place, stated WTP may not equate to what would actually be paid. The flexibility of these techniques in valuing many aspects of environmental quality have led to a voluminous literature on this subject (e.g. reviewed in Mitchell and Carson, 1989; Cropper and Oates, 1992), as well as voluminous critiques of the method and economic valuation more generally (e.g. Sagoff, 1998). Part of the controversy stems from the influence that this technique now holds in determining liability for damage assessment in the US legal system, to the extent that the estimation of damages from the oil spill of the Exxon Valdez in Prince William Sound, Alaska in the early 1990s was partially determined by using a CVM survey.

How to carry out a CVM

The steps in applying the contingent valuation method are set out in note form as follows:

Step 1 Preparation of survey and study

Set up the hypothetical market: individuals may be presented with two basic variants:
- How much are you willing to pay (WTP) for a welfare gain?
- How much are you willing to accept (WTA) in compensation for a welfare loss?

Define elicitation method. The major alternatives are:
- Open ended; 'how much are you willing to pay?' (this produces a continuous bid variable and may therefore be analysed using least squares approaches).
- Take-it-or-leave-it (dichotomous choice); 'are you willing to pay £X?' (this produces a discrete bid variable and requires logit-type analysis).

Other elicitation methods include the use of payment cards and bidding games with suggested starting points. Provide information regarding the quantity/quality change in provision of the good; who will pay for the good; and who will use the good. Define the payment vehicle, for example: higher taxes; entrance fees; or donation to a charitable trust.

Step 2 The survey. Methods include: on site (face to face); house to house (face to face) and mail/telephone (remote) survey techniques. Each of these has its advantages and cost and resource implications. In considering this step the guide by Mitchell and Carson (1989) is illuminating.

Step 3 Calculate mean willingness-to-pay. This calculation depends on whether an open-ended or dichotomous choice willingness-to-pay question has been asked. The following calculation is made in each of these circumstances:

Open ended	simple mean
	trimmed mean (removing outliers)
Dichotomous choice	expected value

Step 4 Estimate the bid function. Most CVM studies will attempt to investigate respondents WTP bids by estimating a bid function. A simple example might by:

$$WTP_{1j} = f(Q_{ij}, E_j, Y_i, S_i, X_i)$$

where Q_{ij} = visits by individuals to site j
WTP_{ij} = individual I's willingness to pay for asset j
E_j = characteristics of site j
Y_i = income of individual i
S_i = relevant socio-economic characteristics of individual i
X_i = other explanatory variables.

Step 5 Aggregation from the mean willingness to pay to gain an overall estimate of value. Calculating total WTP from mean WTP can involve, for example, multiplying the sample mean WTP of visitors to a site by the total number of visitors per annum.

Step 6 Testing the validity and reliability of the estimates produced. This final stage of any CVM study is the most important when the interpretation of the results and their applicability to other environmental goods, or in other situations is important. As discussed in Section 4 on scaling issues, the validity of CVM results is dependent on the acceptability of the hypothetical market by the respondents: attempting to transfer such estimates to other situations with different market and cultural circumstances may not be appropriate.

The numerous CVM studies and the diversity of approaches and goods and services for which values have been elicited using the CVM require careful scrutiny if values are to be compared or transferred to other sites, or policy decisions are to be made on their basis. As a result of the use of such results in the US in the legal process, there has been a call for standardisation. In Box A3.1 the protocol for CVM studies developed by NOAA in the US are outlined. These have been used and tested by many CVM researchers (see Carson, 1997), but should only be taken as a guideline for practice, since they have been developed in the social and cultural context of the US where, for example, referenda on public expenditure on public good provision are regular and hence survey respondents may be familiar with the hypothetical questions used in CVM surveys.

An example of applying the CVM to valuing river water quality improvements

The Monongahela River is a major river flowing through Pennsylvania. Desvousges *et al.* (1987) asked a representative sample of households from the local area what they would be willing to pay in extra taxes in order to maintain or increase the water quality in the river. The analysts conducted several variants of the CVM survey. In one variant households were presented with three possible water quality scenarios and simply asked how much they were willing to pay for each. The scenarios which were described to each respondent group were:

- Scenario 1: Maintain current river quality (suitable for boating only) rather than allow it to decline to a level unsuitable for any activity including boating.
- Scenario 2: Improve the water quality to a level where fishing could take place.
- Scenario 3: Further improve water quality from fishable to swimmable.

Amongst the households surveyed some used the Monongahela river for recreation while others did not. The analysts therefore could look at how much the users were willing to pay compared to the responses of non-users. Results for the sample as a whole were also calculated. Table A3.3 presents the willingness to pay of users, non-users and the whole sample for each proposed river quality change scenario.

Table A3.3 Willingness to Pay (WTP) for river quality scenarios along the Monongahela River, USA.

Water quality scenario	Average WTP of whole sample ($)	Average WTP of users group ($)	Average WTP of non-users group ($)
Maintain boatable river quality	24.50	45.30	14.20
Improve from boatable to fishable quality	17.60	31.30	10.80
Improve from fishable to swimmable quality	12.40	20.20	8.50

Notes: Full details given in Desvousges *et al.* (1987).

Box A3.1 NOAA Panel Protocol for Contingent Valuation Studies

General Guidelines

1. Sample Type and Size: Probability sampling is essential. The choice of sample specific design and size is a difficult, technical question that requires the guidance of a professional sampling statistician.
2. Minimise Non-responses: High non-response rates would make CV survey results unreliable.
3. Personal Interview: It is unlikely that reliable estimates of values can be elicited with mail surveys. Face-to-face interviews are usually preferable, although telephone interviews have some advantages in terms of cost and centralised supervision.
4. Pre-testing for Interviewer Effects: An important respect in which CV surveys differ from actual referendum is the presence of an interviewer (except in the case of mail surveys). It is possible that interviewers contribute to 'social desirability' bias, since preserving the environment is widely viewed as something positive. In order to test this possibility, major CV studies should incorporate experiments that assess interviewer effects.
5. Reporting: Every report of a CV study should make clear the definition of the population sampled, the sampling frame used, the sample size, the overall sample non-response rate and its components (e.g., refusals), and item non-response on all important questions. The report should also reproduce the exact wording and sequence of the questionnaire and of other communications to respondents (e.g., advance letters). All data from the study should be archived and made available to interested parties.
6. Careful Pre-testing of a CV questionnaire: Respondents in a CV survey are ordinarily presented with a good deal of new and often technical information, well beyond what is typical in most surveys. This requires very careful pilot work and pre-testing, plus evidence from the final survey that respondents understood and accepted the description of the good or service offered and the questioning reasonably well.

Guidelines for Value Elicitation Surveys

7. Conservative design: When aspects of the survey design and the analysis of the responses are ambiguous, the option that tends to underestimate willingness to pay is generally preferred. A conservative design increases the reliability of the estimate by eliminating extreme responses that can enlarge estimated values wildly and implausibly.
8. Elicitation Format: The willingness-to-pay format should be used instead of compensation required because the former is the conservative choice.
9. Referendum Format: The valuation question generally should be posed as a vote on a referendum.
10. Accurate Description of the Program or Policy: Adequate information must be provided to respondents about the environmental program that is offered.
11. Pretesting of Photographs: The effects of photographs on subjects must be carefully explored.
12. Reminder of Substitute Commodities: Respondents must be reminded of substitute commodities. This reminder should be introduced forcefully and directly prior to the main valuation to assure that the respondents have the alternatives clearly in mind.
13. Temporal Averaging: Time dependent measurement noise should be reduced by averaging across independently drawn samples taken at different points in time. A clear and substantial time trend in the responses would cast doubt on the 'reliability of the value information obtained from a CV survey.
14. 'Non-answer' Option: A 'non-answer' option should be explicitly allowed in the addition to the 'yes' and 'no' vote options on the main valuation (referendum) question. Respondents who choose the 'no-answer' option should be asked to explain their choice.
15. Yes/No Follow-ups: Yes and no responses should be followed up by the open-ended question: 'Why did you vote yes/no?'
16. Cross-tabulations: The survey should include a variety of other questions that help interpret the responses to the primary valuation question. The final report should include summaries of willingness to pay broken down by these categories (e.g., income, education, attitudes toward the environment).
17. Checks on Understanding and Acceptance: The survey instrument should not be so complex that it poses tasks that are beyond the ability or interest level of many participants.

Source: Adapted from the report of the National Oceanic and Atmospheric Administration Panel on the Contingent Valuation Method (Arrow *et al.*, 1993).

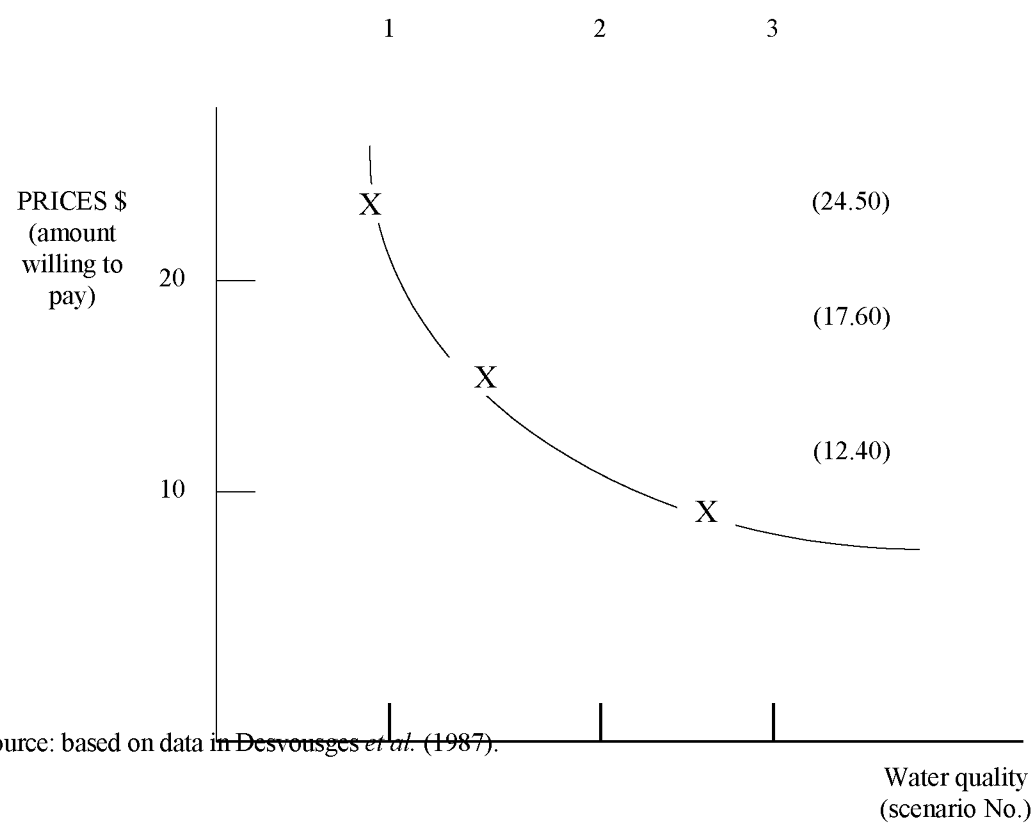
Households were told that the payment vehicle (the increased tax) would either be as a direct tax (e.g. income tax) or an indirect tax (e.g. a tax upon purchases such as VAT).

A number of conclusions can be drawn from these results. Considering the results for the whole sample we can see that the stated WTP sums draw out a conventional demand curve for water quality i.e. people are prepared to pay a relatively high amount for an initial basic level of quality. However, they are prepared to pay progressively less for higher levels of water quality. Figure A3.3 draws out the demand curve indicated by the results for the whole survey, representing the demand for the average household.

From this demand curve we could attempt to calculate the total value of environmental quality at the river. More importantly the value gain experienced by the average household when a water quality improvement is achieved could be derived. The total benefit value of a specific improvement could then be estimated by multiplying this average household value by the number of households which it is thought would be affected by such an improvement. This benefit can then be compared against the cost of achieving such a quality improvement to see if it was worthwhile.

Turning to results for the users and non-users group, both map out conventional downward sloping demand curves. Furthermore, as would be expected, at every quality level the willingness to pay of the users group exceeds that of the non-users, which again would be expected in economic theory.

Figure A3.3 Demand curve for water quality along the Monongahela River derived from contingent valuation data



Source: based on data in Desvousges *et al.* (1987).

Finally notice that the WTP of non-users is not zero. This is due to the fact that such households, while not personally wishing to visit the river, nevertheless do value its continued existence and even upgrading so that others can enjoy its benefits. This non-use existence value (see Figure 3.7) derives from people's altruistic public preferences showing that the concentration upon people's 'private preferences' as demonstrated by the market prices of marketed goods does not always fully capture the entire range of values which people have for things.

Constraints, biases and difficulties with the CVM method

There are a number of methodological issues in CVM outlined in Figure A3.4. These are dealt with in detail in many texts on CVM and are outlined only briefly here. Concentration of effort in the design phase of CVM studies on these difficulties can make the results more robust. However, the elimination of all 'biases' in Figure A3.4 is a misnomer, in that there is no true unbiased value for any asset under this technique: all values are *contingent* on the circumstances and the information provided concerning the hypothetical market.

Will respondents answer honestly? Free riding. If the individual has the opportunity to, in effect, name their own price for a good (as in the open-ended WTP approach) then economics predicts the individual will pretend to have less interest in a given collective activity than he really has, which is known in economics as free-riding. A number of CVM-type experiments have examined the extent of free riding by comparing individuals' stated WTP with what they actually paid for a good.

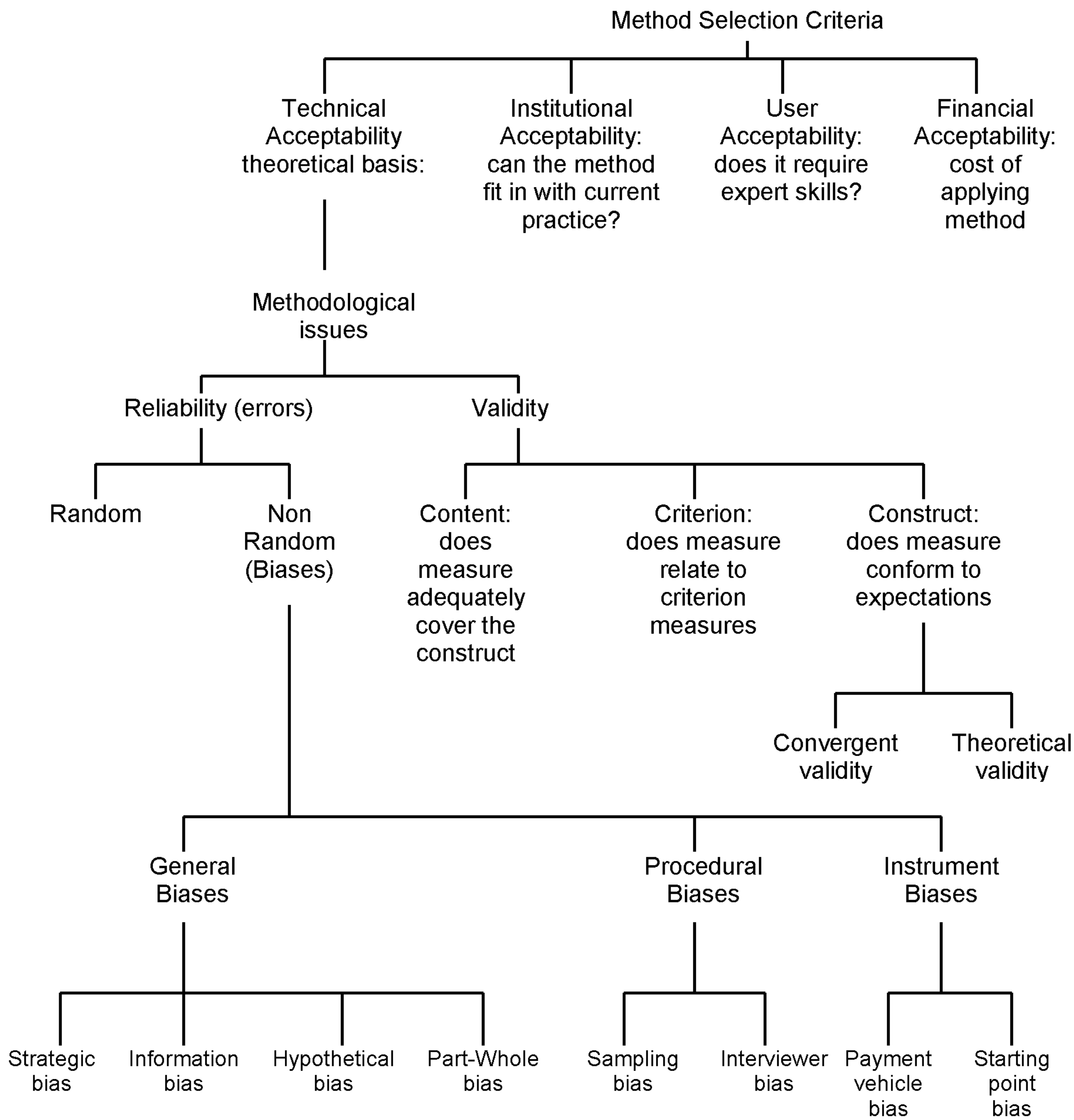
In a classic series of related experiments, US researchers examined the statements and actual payments of recreational hunters with regard to their hunting permits (e.g. Bishop and Heberlein, 1979). In testing a willingness to accept (WTA) approach, Bishop and Heberlein note that, while only 40% of hunters stated that they would be willing to accept $50 in return for losing their recreational hunting, over 90 percent subsequently accepted a $50 cheque in return for cancelling their permit. In other words, true WTA was considerably lower than stated WTA.

In one test of a WTP format, one group of hunters were asked to state how much, hypothetically, they would be WTP for a permit while a second sample were offered permits at various prices with actual payment required to determine true WTP. Using the payment vehicle of a sealed bid auction submitted by mail, the stated WTP was $24 while the actual WTP was $32. Therefore stated WTP was 75 percent of actual WTP i.e. a weak free-rider effect was detected.

It is, by definition, impossible to repeat this experiment for a pure public good. However, these results indicate that the responses obtained from open-ended WTP format studies do provide a reasonable lower bound estimate of true WTP (true welfare measure) i.e. free riding does not completely invalidate such an approach (however WTA formats perform badly in these tests). It has been pointed out that dichotomous choice approaches (as recommended in the NOAA protocol in Box A3.1) should limit possibilities for free-riding behaviour (Hoehn and Randall, 1987).

Strategic Overbidding. Conversely to the free-riding incentive, some respondents, perceiving that analysts are interested in mean WTP, may overstate their WTP in an effort to inflate the mean and so improve the prospects for provision of the good in question. Bateman *et al.* (1993) speculate on the possibility of strategic overstatement in their open-ended WTP responses with respect to a survey on the preservation of landscape assets in the Norfolk Broads area in the UK. Here, truncation of the top 5 percent of bids resulted in a drop in mean WTP of over 30 percent, perhaps suggesting that a small number of survey respondents can inflate the mean WTP by strategic bidding to enhance the value of the environmental good to the policy maker who is using the results. However, such a result is a poor test of strategic overbidding.

Figure A3.4 Criteria for the selection of a monetary evaluation method and issues within the validity of contingent valuation studies.



Part Whole Bias. Kahneman and Knetsch (1992) report no significant differences in WTP from a sample of respondents asked to value a small number of Canadian lakes and another sample asked for their WTP for all lakes. One would expect that there would be significant differences in this case as the value of one lake should be less than the value of all lakes. This indicates two problems: respondents have difficulty in separating out one aspect of a larger asset; and respondents have difficulty taking into account available income and other demands when making their WTP bids. This issue can be overcome in survey design by asking respondents to calculate a total yearly budget for all environmental issues and then to consider all the other demands upon this budget before asking the WTP for the good in question.

Other biases. These include issues such as information effects on the responses: the values are dependent on the type and presentation of the information about the good in question, particularly if respondents are non-users, or even of the demeanour and cultural setting of the interviewer. The issue of whether to include non-users as the relevant population across which the mean WTP should be aggregated to give an aggregate WTP is a further issue of some debate.

Two further issues in CVM surround the means by which the hypothetical WTP is presented: some respondents may be directly opposed to taxation on principle, so asking WTP for environmental protection as determined by a contribution to taxes may result in many zero bids, under-representing the underlying WTP of respondents. Alternative so-called payment vehicles in CVM include hypothetical contributions to taxes, entrance fees (for recreation sites and other location-specific assets), or a contribution to a trust fund specifically set up to bring about the environmental goal being specified in the survey. Studies have shown that changes in the method by which respondents would have to pay (the payment vehicle) result in changes to mean WTP. Bateman *et al.* (1993) suggest that much of this can be attributed to varying rates of refusal to pay.

The same study also highlights differences in estimates of WTP resulting from different so-called elicitation methods. In the study valuing landscape preservation in the Norfolk Broads (Bateman *et al.*, 1993) used three methods to elicit WTP responses:

- open ended 'What are you willing to pay?'
- dichotomous choice 'Would you pay £X?' (X is varied across sample)
- iterative bidding: asking a series of yes or no questions to arrive at a refined WTP.

The differences in the resulting estimates of WTP are illustrated in Table A3.4.

Table A3.4 Estimates of willingness to pay for recreation and amenity for Norfolk and Suffolk Broads, UK.

	Sample size	Mean WTP[1]	Median WTP (£)	Std devn	S.E. mean	Min bid (£)	Max bid (£)
Open Ended WTP Study	846	67.19	30.0	113.58	3.91	0.0	1250.0
Iterative Bidding WTP Study	2051	74.91	25.0	130.1	2.87	0.0	2500.0
Dichotomous Choice WTP Study	2070	140	139	n/a	n/a	n/a	n/a

Notes:
1. Includes as zeros, those who refused to pay anything at all.

Source: Adapted from Bateman *et al.* (1993).

There is therefore clear evidence from these results of upward anchoring in the dichotomous survey, by which the first amount asked would be responded to positively by respondents even if their WTP was somewhat lower. This effect, however, potentially compounds free-riding in the open-ended study, where, as outlined above, there may be upward bidding by some respondents. Different elicitation methods lead to different respondent characteristics. Dichotomous choice approaches appear to result in upward anchoring whilst open-ended elicitation approaches engendered downward free riding.

All of these effects in contingent valuation are important to consider at the design stage. As each of these is refined, the range of potential environmental goods and valuation issues to be addressed can be increased. The limitation of the technique, however, remains those outlined at the beginning of this section, namely that the validity and robustness of the estimates is dependent on the acceptance of a hypothetical market by respondents. Critics of the technique argue that this issue invalidates the technique both because decision-makers are led to create the markets which have been suggested as hypothetical in such studies, and because decisions on environmental preservation are separated in many respondents respondents' perceptions from market transactions (see for example Sagoff, 1998; Burgess *et al.*, 1997).

Conclusions on the contingent valuation and other valuation techniques

Arguments concerning the CVM are often put in terms of whether some estimate, however flawed, of monetary values of the environment are better than an absence of estimates. This is the view that economics *demonstrates* the value of ecosystem services and functions and hence leads to arguments about impacts and response (within the P-S-I-R framework) being taken with at least a partial view of economic dimensions of value, which are but one element: "because ecosystem services are not fully captured in commercial markets or adequately quantified in terms comparable with economic services and manufactured capital, they are often given too little weight in policy decisions" (Costanza *et al.*, 1997, p. 253).

Leaving aside the arguments over the role of valuation when applying these techniques, the review in this appendix has demonstrated that CVM is the most flexible technique in terms of the range of environmental quality changes which can be tackled, subject to these hypothetical markets being acceptable and familiar to survey respondents. However, evaluation of the technique itself stirs up the most vehement of rhetoric. This appendix takes a middle view that for use-values with well perceived environmental goods, CVM seem to give a valid order of magnitude estimate of economic use and non-use values. Such information is useful in the decision-making process. For non-use values, particularly where goods are poorly perceived or understood, the method may give a spurious air of economic analysis to what is, on the part of respondents, guesswork.

The range of techniques outlined in this Appendix is well-established, and the methods have been applied by a variety of researchers to coastal management issues. The critical issue of interpretation of these values, and indeed what the term value means in the context of decision-making, is highlighted in the main sections of the report and in Turner and Adger (1996, pp. 34-49).

Appendix 4. Glossary

Definitions supplied in this section are operational in the context of this report and simplified for communication with a broad audience.

Abatement cost curves	Calculation of the least costs to reduce a given environmental pressure by adopting available technical environmental measures.
Alternative Futures Scenarios	Similated sequence of events possible in the future.
Ambient	Relating to a condition of the environment that surrounds a body or object. Therefore, ambient environmental quality is to signify the quality of the surrounding environment.
Annualised (homogenised) data	Information that has been uniformed to relate to a period of one year.
Anthropic	Relating to humankind.
Anthropocentric	Regarding humankind as the central or most important element of existence.
Anthropogenic	Influenced or caused by activities of human beings.
Assimilative capacity	Ability of a system to incorporate and absorb substances of diverse chemical composition.
Benefit	Gain in economic and/or social welfare.
Benefits transfer	The transfer of economic valuation estimates across time and space
Bequest value	Value of a resource that will be preserved for future generations.
Biogenic	Resulting from activities of living organisms.
Biogeochemical cycle	Circulation of chemical components through the biosphere from atmosphere, hydrosphere and lithosphere.
Biome	Ecosystems across a landscape scale.
Budget model	Estimate of inventories, inputs and outputs of a system over a specified period of time, based on accounting for material balances and flows.
Capital	Resources, both man-made and natural.
Carbon cycle	Circulation and movement of carbon atoms through the biosphere, atmosphere, hydrosphere and lithosphere.
Carbon sequestration	Net accumulation of carbon via a range of terrestrial and marine processes.
Carbon storage	The stock of carbon accumulated in various sinks.

Catch crop	Crop serving to hold or restrain certain components usually free-moving in the soil.
Ceteris paribus	Latin expression for 'all other things being equal'.
Clean technology	Application of improved technology to production processes in order to decrease the resulting impacts on the environment (as opposed to end of pipe add-on technology).
Climate change	Alterations in the Earth's energy balance (mainly consisting of incoming and outgoing radiation that determine the surface temperature) and climate.
Coastal	Referring to processes or features of the coastal zone.
Coastal typology	Characterisation of regions of the coastal zone on the basis of specific related types and features constituting the objective of research. *See also 'typology'*
Coastal zone	"Extending from the coastal plains to the outer edge of the continental shelves, approximately matching that region that has been alternatively flooded and exposed during sea-level fluctuations of the late Quaternary period" (Holligan, P. M. and de Boois, H. 1993. *Land-Ocean Interactions in the Coastal Zone (LOICZ) Science Plan*. IGBP Report no. 25, 50pp.). Note: LOICZ has at least other three possible definitions of coastal zone. The above relates to an area extending between +15 m and -150 m.
Continental shelf	Gentle sloping submerged platform, part of the continental margin, between the shoreline and the continental slope. Usually taken to be shallower than 200 metres.
Contingent Valuation Method (CVM) ***or simply, Contingent Valuation (CV)***	Expressed preference valuation method, whereby evaluations are derived from direct questioning of individuals to determine their willingness to pay (WTP) for a certain environmental good or service or what they are willing to accept (WTA) for its loss.
Cost	Loss in utility or practical use. Also, the price required or paid for acquiring, producing or maintaining something, usually expressed in monetary terms.
Cost effectiveness	Achieving one or more targets at lowest costs possible.
Cost-benefit analysis (CBA)	Appraisal of the total social and economic costs and benefits derived from the development of a project, programme or decision.
Cross section data	Economic observations taken at the same time.

Defensive expenditures Expenditure by households or other agents on measures to combat the ill effects of pollution e.g. noise insulation, water filters.

Demand Ability to purchase certain goods / services.

Denitrification Biologically induced conversion of nitrate to gaseous forms of nitrogen.

Determinism Philosophical doctrine or outcome in a mathematical model that all phenomena are inevitable consequences of prior events.

Direct use value Value derived from the direct use of an asset's resources and services.

Discounting Process of calculating the present value of a certain amount by applying a discount rate (interest) to a sum. Usually used to determine the equivalent present value of sums payable in the future (for instance, given a discount rate of 10% on a sum of £110 receivable in one year's time, the present value corresponds to £100).

Dissolved Matter Freely available matter.

(Operational) Molecular or ionic species in water, capable of passing a filter of defined size (typically < 0.001 mm).

Drainage basin Area occupied by a topographically defined drainage system; a region that collects surface runoff and supplies it to a specific body of water (e.g. streams, lakes). *Also, catchment area.*

Driver A force/action causing change.

Econometrics Branch of statistics testing economic hypotheses and estimating economic parameters making use of multiple regression techniques and other methodology.

Economic efficiency In an economy, allocation of resources leading to a net gain to society, estimated by subtracting the costs from the benefits.

Economic welfare The part of human well-being (at individual or group level) resulting from the consumption of goods and services.

Economics Science concerned with the efficient allocation of scarce resources within different contexts.

Ecosystems Functional units of the environment with characteristics of its interactions among biological, physical and chemical components.

Embedding In contingent valuation method (CVM), influence of other interests and concerns, apart those explicitly questioned, in the estimation of willingness to pay (WTP) values.

Energy Metrics Analytical approach which converts economic factors of production (land, labour and capital) into energy equivalent terms.

Estuary	A semi-enclosed body of water with a fresh-water input and a free connection to open sea.
Eutrophication	Increase of the amount of nutrients, especially nitrogen and phosphorus, in a marine or aquatic ecosystem.
Evaluation	Process of determining the value of something.
Evaluation method	Ranking of alternatives by using rules (decision rules) that facilitate this process.
Existence value	Value of a resource that will never be used by present or future generations.
Extreme event	Phenomenon of unexpected or statistically unlikely magnitude (e.g. flood, drought, earthquake).
Feedback	(Mathematical) Non-linear effect of a product or action in a multistage process on a subsequent stage in the same process.
Flow	Movement of matter or energy.
Flux	Measure of the flow of some quantity per unit time; it may also be expressed per unit area or unit volume per unit time.
Function	Specific role of a person, system or thing in a determinate context.
Global Carbon Cycle	One of the main biogeochemical cycles on Earth; carbon is cycled around the globe among different reservoirs by physical, chemical and biological processes.
Global Environmental Change	Cumulative process of change, driven by human use of environmental space and resources.
Good	Commodity that is tangible, usually movable and in general not consumed at the same time as it is produced.
Gross National Product (GNP)	Total value of all final goods and services produced by a nation in a year.
Hedonic price	Implicit or shadow price of a good's characteristics.
Hedonic pricing method	Revealed preference method whereby individuals' valuations of environmental goods are determined by the market prices of the goods purchased (e.g. house prices) which, it is assumed, is done to enjoy the environmental good in consideration.
Human welfare	Social and economic well-being of a human group related to the things that contribute to it. *See also economic welfare and social welfare.*
Impact analysis	Assessment of the negative effects (damages) accruing to a site and to regional or local economy from an environmental impact.

Income elasticities of demand	Proportional change in quantity demand for a given change in income.
Indirect use values	Support and protection provided to the economic activity by the natural functions of the ecosystem or by regulatory 'environmental' services, such as flood alleviation.
Inner shelf	Portion of the continental shelf closest to the shore.
Integrated Coastal Zone Management (ICZM)	Management of coastal areas based on the integration of knowledge of coastal zone processes and different possible management options, to develop policies for a sustainable use of littoral areas.
Integrated management	Control of processes and actions by managing them as a whole, taking into account interactions throughout the system.
Integrated prognostic assessment capability	Capacity of predictive estimations based on the incorporation of scientific and socio-economic data on the system under scrutiny.
Interface	Boundary between two media, environments or areas.
Intergenerational equity	Taking into account the preferences of future generations in current actions and decision-making.
Land use/cover	Pattern of vegetation and settlement across the terrestrial environment.
Leach	To remove or be removed from a substance by a percolating liquid.
Ley grass	Grass cultivated temporarily on arable land.
Limnic organisms	Organisms living in freshwater ecosystems.
Load	Material moved or carried by a natural transporting agent; the total content of a material (often detrimental) within a system.
Longitudinal data	Information collected on constant experimental units over a period of time.
Macro scale	Study of a greater proportion of the subject or area under consideration, possibly as a whole. 'Macro' is a prefix meaning *large* or *great*.
Marginal cost	Additional cost of producing an extra unit of output.
Market analysis	Valuation method based on market prices.
Morphodynamics	Changes in form and structure.
Multi-criteria analysis	Appraisal of different projects by considering criteria that affect groups or individuals in different ways.
Multi-criteria evaluation	*See multi-criteria analysis.*

Nimbyism	The 'not in my backyard' syndrome i.e. local resistance to facilities such as waste disposal sites, incinerators because of fear of pollution and loss of local amenity.
Nitrogen fixation	Conversion of nitrogen gas (which most organisms cannot use) to organic nitrogen, nitrate or ammonium; these are all forms of nitrogen which can be readily used by organisms.
Nonuse value	Value that an individual may give to an asset even without personally using or intending to use it.
Nutrient budget	Quantitative estimate of the inputs and losses of nutrients to an ecosystem.
Nutrient cycle	Movement of nutrients through a system.
Nutrient uptake	Conversion of nutrients (carbon, nitrogen and phosphorus) from inorganic forms to organic matter.
Nutrients	Raw materials needed for life. Major plant nutrients are carbon, nitrogen and phosphorus.
Observational study	Research in which experimental conditions are not controlled and conclusions are drawn based on historical relationships among variables. *Also called comparative study.*
Opportunity cost	Market value of things that would be forgone (or lost or given up) to obtain something else.
Option value	Value of a resource not being used at present, but with the option of being used in the future.
Organic matter	Molecules in organisms derived from degradation of organisms or excreted by organisms after being synthesised.
Organic production	Conversion of inorganic materials (especially carbon) to organic matter.
Outer shelf	Part of the continental shelf most remote from the shore.
Oxygen deficits	Reduced amount of oxygen, lower than expected or required.
Partial valuation	Assessment of two or more alternative use options of an environmental asset.
Particulate matter	Matter composed of particles that are not superficially bound together.
Phosphorus/Nitrogen limited water	Water in which the availability of the nutrients is approaching a lower threshold likely to cause changes in the growth of plants.
Phytoplankton	*See Plankton.*

Plankton	Organisms - bacteria (bacterioplankton), algae and cyanobacteria (phytoplankton) or animals (zooplankton) - living suspended in fresh-water or marine environments.
Point sources	Confined sources that can be identified as the origin of inputs into the surrounding medium (as opposed to non-point sources).
Policy instruments	Economic and social variables manipulated by the government to influence policy variables.
Pressures	Present and forecasted socio-economic activity levels.
Redfield ratio	Common ratio (by moles, not mass) of carbon, nitrogen and phosphorus in organic matter, especially in the ocean (C:N:P = 106:16:1 for plankton and plankton-derived organic matter).
Redox potential	Measurement of the ability of an element to act as a reducing (by acquiring electrons) or oxidising (removal of electrons) agent.
Remote sensing	Gathering and analysis of data from an object physically removed from the sensing equipment (e.g. satellite or aerial photography, subsurface detection instruments).
Residence times	Total amount of material present in a system divided by the rate of delivery of that material.
Residuals	Non-product outputs from human activities; these become wastes if not re-used or recycled.
Restoration	Return to a previous state or condition, especially a condition of well-being, obtained by replacing those components lost.
Scoping	Determining the area covered by an activity, topic.
SEK	Swedish currency (Swedish Krone).
Sensitivity analysis	Changing parameters of a decision problem or mathematical model in order to evaluate how this affects the outcome.
Services	(In an economic sense) performed functions or tasks for which there is a demand and hence a market price.
Shadow Projects	Projects compensating for environmental damage generated by existing/planned set of economic activities with the provision of an equal alternative elsewhere.
Sink	Area, device or environmental 'compartment' that absorbs, retains or transforms a flow of matter or energy.
Social welfare	Well-being of a society or community. In general, social welfare is seen as an aggregate of the welfare of different members of society.

Stakeholder — Individual, group or institution potentially affected by a specific event, process or change.

Stakeholder analysis — Management tool to ensure that policy planning is carried out efficiently and effectively. In economics, it concerns the quantification in monetary terms of costs and benefits accruing from a project / proposal / decision to the different groups in society that have a related interest in it. This is based on the notion that policies, programmes or projects have differential effects on a range of actors, who gain or lose according to their interests. From a sociological viewpoint, it aims to help organisations work through different courses of action and to identify the actors that are likely to favour and press for particular kinds of change and to consider the opinions and interests of these groups (techniques such as Focus Group analysis are employed).

Strategic mitigation — Re-creation of habitat in a location different to where originally situated in order to compensate for its loss elsewhere.

Supply — (In economics) goods or services available for purchase.

Sustainability — **Strong:** maintenance of the amount of capital available to a population, keeping the quantity of natural capital (or 'critical components of ecosystems') constant over time. An axiom of this condition of sustainable development is that natural capital can not be replaced by other forms of capital; the four forms of capital (natural, human, human-made and social-moral) are complementary to one another.

Weak: constant maintenance of the overall amount of capital available to a population, allowing for exchanges (unlimited substitution possibilities) among different forms of capital, achievable through technological progress.

Sustainable development — "Development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (World Commission on Environment and Development (1987) *Our Common Future*, Oxford University Press, Oxford, p. 43). *See also weak and strong sustainability*.

Sustainable management — Sustainable utilisation of the multiple goods and services provision generated by coastal resources.

Total economic value — Total of use and non-use values.

Total valuation — Assessment of the total net benefits to society (total economic contributions) of the ecosystem under consideration.

Trace gas — Gas occurring in very small quantities ('trace quantities').

Transition economy — Country whose economic activity is progressing towards assuming those characteristics of developed or industrialised nations.

Travel cost	Estimation of the value of visiting an ecosystem derived from the cost of travel to that location, including the recognition of the opportunity costs of travel time [revealed preference method].
Turbidity	Condition (usually of a liquid) resembling cloudiness created by the suspension of particles.
Typology	See the definition given in Appendix 1. Also: A system of classification or grouping of entities based on similarities among combinations of characteristics.
Valuation	Quantification of the values of a good or service usually calculated by examining the demand for it showing how much people would use at varying prices.
Value	The worth of a good or service measured in terms of willingness to pay minus the costs to supply it. *Refer also to: direct use values, indirect use value, nonuse value, option value, total economic value and different methods of estimating these: contingent valuation, market analysis, hedonic pricing etc.).*
Wetlands	"Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six metres" (art. 1.1) and "may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands" (art. 2.1 of the Ramsar Convention (1996) *Strategic Plan 1997-2002*, Ramsar Convention Bureau, Gland, Switzerland).
Willingness to pay (WTP)	Amount that an individual is prepared to pay to purchase a good or the use of a service, independently of the existence of a prevailing market price or if the good / service is free of charge.
Zooplankton	*See Plankton.*

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Appendix 1. LOICZ Typology

The current 'typology' plan of action involves the following conceptual structure:

1. The short-range objective is a system that makes it possible to globalise or extrapolate local and regional flux estimates derived from budget models or other sources by applying such results to other coastal reaches of the same or similar type.

2. The products should be as rigorous, versatile, and comprehensive as possible, but the twin constraints of time schedule and applicability to biogeochemical flux globalisation are paramount in the early stages of typology evolution.

3. The objective is the development and continued expansion of a number of data sets which could be used for testing typologies for coastal zone processes and compartments, and especially the estuarine and inner shelf portions of the coastal zone. Data sets directed towards typologies for the outer shelf and exchanges with the open ocean should be handled separately; extensive development - primarily terrestrial data sets - should be coordinated with other relevant programmes.

4. The objectives of identifying fluxes through, and transformations of, materials in the estuarine zone may be effectively achieved through the identification of data sets which might be used for three component sets of typologies, each of which would have the potential for including multiple typologic approaches:

   a) an '**input**' typology class representing primarily natural and anthropogenic fluxes from land and atmosphere into the estuarine zone;
   b) a '**transformation**' typology class characterising the biogeochemical reactions within the zone (e.g. net primary production, biomineralisation); and,
   c) an '**exchange**' typology class representing the exchange of material with the inner shelf - primarily the outer shelf and ocean.

5. The formulation of data sets for 'transformation' and 'exchange' typologies was relatively straightforward and depended on a reasonable number of primarily marine components. In contrast, the data for the 'input' typology deal primarily with terrestrial variables and their interactions, making its development more challenging.

There are three central issues in the typology process. One is the nature, appropriate scale, and potential problems with readily available digital databases of relevant environmental variables. A second point of concern is that the variables should to be sampled at the same spatial and temporal scales, so that there would inevitably be certain types of desirable statistical or modelling manipulations that should not be used with certain typology data sets. Third, the problem of defining the landward portion of the coastal zone in a practical fashion depends on, and is constrained by, both of the other issues, as well as by operational considerations.

One key question is whether to use a definition of the landward extreme of the coastal zone based on a simple topographic criterion, such as elevation, or to use a definition that incorporates some aspects of drainage basins. Although a drainage-basin approach is favoured in principle, the whole-basin approach would expand the definition of the coastal zone to full continental coverage and greatly increase data needs and processing requirements. The possible use of topographically defined coastal basins, or of coastline assignment to basins defined on the basis of divides between major river watersheds is a longer-term objective of the typology development process.

The initial data collection for the typology initiative has been undertaken on the basis of a coastal strip defined by the 50 m depth and elevation contours and the coastline. Although relatively arbitrary, this is considered conservative in terms of including the inner shelf, or 'estuarine zone', and the most relevant portion of the terrestrial coastal zone. This definition is also relatively quick and easy to implement for trial applications and to test against alternative topographic definitions. When combined with classified, as opposed to continuous, numerical values of environmental data, a reasonable number of the basic goals of typologic extrapolation or globalisation could be met without violating calculational principles.

The initial LOICZ Typology Data Set has been compiled for coastal cells on a $1^{\circ} \times 1^{\circ}$ grid between the -50 m to +50 m global elevation generated from TerrainBase (NOAA, 1995).

Data for Input typology. The LOICZ Typology data set is under development and can be viewed and downloaded at **http://www.nioz.nl/loicz/projects/core/typo/**. The following list indicates both the desired data sets and those available (*) as of September 1998 (refer also to the full Typology Data Set breakdown available on page 96 from the LOICZ url).

- * Vegetation class (3.17);
- Land cover;
- * Soil type (3.11; 3.13);
- * Soil carbon content (3.13);
- * Soil texture (3.12);
- * Soil moisture (3.33);
- * Monthly precipitation and evaporation - mean and extremes;
- * Vegetation Index (NDVI);
- * Monthly temperature - mean and extremes (3.28, 3.29);
- Fertiliser (N and P) use;
- * Population density (CIESIN gridded data - http://www.ciesin.org/) (3.22); and,
- River discharge of fresh water (3.23), sediments and nutrients.

Data required for Transformation typology:
- * Coastal Zone Color Scanner (CZCS) data (SeaWifs - Feldman *et al.*, 1989) (3.34);
- * Sea Surface Temperature - mean and extremes w/months of occurrence (SeaWifs); and,
- * Monthly irradiance - mean and extremes w/months of occurrence (ISLSCP, 1996) (3.27).

Data required for Exchange typology:
- * Freshwater flow - monthly mean and extremes;
- * Tidal frequency (3.7);
- * Tidal magnitude (3.8);
- * Wind speed and direction (3.24, 3.25, 3.30);
- Coastal sinuosity; and,
- Areal extent.

From the integrated modelling/assessment perspective, a number of key issues now emerge:

- what other demographic and socio-economic data sets are available or could be constructed in order to improve the utility of the 'input' typology, by increasing its comprehensiveness to cover all significant environmental change pressures?;

- is it desirable and feasible to formulate a fourth component typology 'human welfare' characterising the initial spatial location and density of populations and their economic activities in juxtaposition to changing C, N & P flux situations and other climatic etc. change factors (perhaps along the lines of vulnerability indexes)?

Typology Data Set (Compiled and edited by M. van der Zijp).

This can be found at: **http://www.nioz.nl/loicz/projects/core/typo/frame1b.htm**

META DATA

3.1. Variable: Grid cell ID

3.2. Variable: Longitude and latitude

3.3. Variable: Country name, region and continent

3.4. Variable: Basin ID

3.5. Variable: Cell location ID

3.6. Variable: Wave height

3.7. Variable: Tidal type

3.8. Variable: Tidal range

3.9. Variable: Cultivation intensity

3.10. Variable: Methane

3.11. Variable: Soil type

3.12. Variable: Dominant soil texture

3.13. Variable: Soil carbon content

3.14. Variable: DSRF, Dunes, swamps and glaciers

3.15. Variable: Ecosystem

3.16. Variable: Coral

3.17. Variable: Vegetation class

3.18. Variable: Tropical forest destruction

3.19. Variable: Morphologic and tectonic classification

3.20. Variable: LGP

3.21. Variable: GNP

3.22. Variable: Population density

3.23. Variable: Runoff

3.24. Variable: Tropical storms

3.25. Variable: Winter gales

3.26. Variable: Precipitation

3.27. Variable: PAR

3.28. Variable: Dew point temperature

3.29. Variable: Mean air temperature

3.30. Variable: U-wind

3.31. Variable: NDVI

3.32. Variable: Surface temperature

3.33. Variable: Soil moisture

3.34. Variable: CZCS

3.35. Variable: Salinity

3.36. Variable: Ocean current

Appendix 2. The use of Input Output economic modelling for integration of environmental impacts

What is an IO model?

Input Output (IO) models are a representation of all the economic activity which takes place in a national economy based on flows of economic value (e.g. in dollars) between sectors. They are widely used by national economic planners to estimate the impacts of exogenous changes in the economic system on particular sectors, such as on the agricultural sector or the household sector, or on final demand and employment. The models are based on matrix tables where the non-leading diagonal elements make up the inter-sectoral flows. The data for these models are normally held by government statistical services with the major coefficients being re-estimated periodically through sectoral surveys, but perhaps only every decade. A major text on the IO approach is that of Miller and Blair (1985).

IO models can be extended in various ways such that they can potentially contribute to integrated modelling in coastal areas. Firstly, regional models can be calibrated such that the flow of goods and services is specified for a specific region of a country. Secondly, and most importantly for this purpose, a set of environmental coefficients can be developed such that flows of economic activity between sectors can be represented as flows of materials or pollutants. Early examples of this analysis include Ayres and Kneese (1969) who demonstrated that the production of environmental impacts from all sectors of the economy is pervasive and that increasing the overall scale of economic activity increases the sector-specific production of pollutants and other waste products, known in economics as externalities. The major limitations of the IO approach in general are in relating even a regional IO matrix to a particular coastal zone since the models themselves are not spatial in nature; and in the availability of data, particularly of the pollution coefficients from each sector. On this latter issue, the approach signifies the periodic flows of pollutants, where the coefficients can be estimated, but does not distinguish between those pollutants which are cumulative in the environment, such as many heavy metals and other substances which accumulate in coastal marine life, from those which are non-persistent. This issue highlights one of the general limitations of the IO modelling framework: that it is in general static, and has difficulty in handling both materials flows and technological change across time.

Despite these limitations, the IO modelling framework can be used in integrated modelling of coastal change by demonstrating the impact of scenarios of driving forces or pressures on the coastal zone on the state of these resources through loading of pollutants and other materials. The example given below comes from a regional IO model in the Philippines (Mendoza, 1994; Orbeta *et al.*, 1996) which is being utilised in conjunction with other models to examine, among other things, the impact of land-based environmental changes on the coastal environment of the Lingayen Gulf, under the SARCS/WOTRO/LOICZ project in the Philippines (contact Liana MacManus and Doug McGlone).

What are the basics of computation of IO models?

As outlined above, IO models are made up of matrices representing flows of goods in the economy. Matrix algebra forms a convenient shorthand for outlining how the computations are made, while the tables themselves are presented through this section, drawing on the regional IO developed in the Philippines.

An Input Output relationship for an economy can be expressed in matrix form as:

1) $\mathbf{X} = \mathbf{AX} + \mathbf{Y}$,

where $\mathbf{X}$ = an n x 1 vector of gross output, $[X_i]$, with X_i being the gross output from each production sector,
$\mathbf{A}$ = an n x n technical coefficient matrix, $[a_{ij}]$, with a_{ij} as defined above,
$\mathbf{Y}$ = an n x 1 vector of final demands, $[Y_i]$, with Y_i being total final demand for sector i.

Equation 1 relates supply (**X**) to demand (**AX** + **Y**), where intermediate demand is now represented by the matrix **AX**. Matrix manipulation of equation 1 yields:

2) $$\mathbf{X} = (\mathbf{I} - \mathbf{A})^{-1} \mathbf{Y},$$

where **I** is an identity matrix, and
$(\mathbf{I} - \mathbf{A})^{-1}$ is referred to as the Leontif inverse.

The elements of the Leontif inverse are known as output multipliers. Each row element indicates the value of the change of a sector's output due to a unit change in final demand for the sector's output. A low column sum reveals a weak sectoral interlinkage; otherwise, it shows a sector's strong dependence on the other sectors' output to meet a unit increase in final demand for its output. The sector with the largest multiplier provides the largest total impact on the economy.

One common use of the IO framework is to examine the effects of an exogenous change in final demands (for example, an increase in population that causes an increase in household demand). These effects are determined from the following:

3) $$d\mathbf{X} = (\mathbf{I} - \mathbf{A})^{-1} d\mathbf{Y},$$ where $d\mathbf{X}$ = changes in sectoral gross outputs,
$d\mathbf{Y}$ = projected changes in final demands.

Thus, if an increase in population were to double the final demand from the household sector, equation 3 could be used to determine the changes in production ($d\mathbf{X}$) necessary to supply this extra demand.

Changes in sectoral gross output may not be the only item of interest to policy-makers. For example, there are certain production outputs (such as residuals, or pollution) that are not included in typical demand categories. Consider the adjustment of the basic model for the inclusion of residuals generation. This adjustment may be captured in a vector of impact variables.

Start with a matrix of residual or direct impact coefficients $\mathbf{v} = [v_{kj}]$, where v_{kj} is the amount of pollution of type k generated per (currency unit)'s worth of industry j 's output. Thus, the total pollution associated with a given level of output (**V**) is given by:

4) $$\mathbf{V} = \mathbf{v}\,\mathbf{X},$$

or total pollution = pollution per unit output times output. This approach assumes that each sector produces pollution in fixed proportion to its output.

Inserting equation 2 into equation 4 gives:

5) $$\mathbf{V} = [\,\mathbf{v}\,(\mathbf{I} - \mathbf{A})^{-1}\,]\,\mathbf{Y},$$

where the bracketed quantity is a matrix of total impact (residual) coefficients. An element of this matrix is the total residual production generated per (currency unit)'s worth of final demand.

Changes in impact variables due to a change in final demand may be found using:

6) $\mathbf{dV} = \mathbf{v}(\mathbf{I} - \mathbf{A})^{-1}\,\mathbf{dY}$,

or, substituting from equation 6:

7) $\mathbf{dV} = \mathbf{vdX}$

Equation 6 may be used to estimate changes in pollution emissions brought about by a projected change in final demands. Equation 7 may be used in the case of projected changes in gross outputs.

An application in the Philippines

Orbeta *et al.* (1996) applied the above methodology in a policy simulation study for the Philippines. This study was prepared for the Philippine Environmental and Natural Resource Accounting Project and applied the Environmental and Natural Resource Accounting Framework (Mendoza, 1994) to analyse the resource and environmental impacts of economic policies at a regional level. This framework included modification of an 11 sector by 11 sector inter-industry transactions table to account for income from non-marketed, nature-based household production and environmental variables. The 11 x 11 transactions table was also extended with the endogenisation of the household sector to account for the household consumption response to changes in labour income, which is dependent upon sectoral gross output. This procedure involves movement of the personal consumption expenditure (PCE) sector out of final demand and into the technically interrelated table. In addition, the compensation of employees component of the value added sector rows is moved into the technically interrelated table. Endogenisation of the household sector can be important, since a considerable amount of pollution is discharged by this sector (Mendoza, 1994).

For this regional analysis, a 1988 intraregional 11x11 inter-industry transactions table of the non-competitive type (i.e., inter-industry transactions are confined to the region and refer purely to regionally produced goods and services) was used to simulate the impacts of four alternative development scenarios and the regional growth targets espoused in the Medium-Term Development Plan of Southern Mindanao, Philippines, for the period 1993-1998. The intraregional inter-industry transactions table was derived using the national IO coefficients as a first approximation of the region's IO structure (Secretario, 1995). This procedure assumes that the production technology in the region is the same as that in the nation as a whole. The coefficients are made region-specific using the simple location quotient approach.

The regional inter-industry transactions table is provided in Table A2.1, which is an empirical example of **X** in equation 1, but in expanded form. Table A2.1 disaggregates the purchasing sector category into 'compensation of employees' (CE), 'other value added' (OVA), and 'imports' (M) sectors. Table A2.1 also disaggregates the final demand sector into personal consumption expenditures (PCE), government consumption expenditures (GCE), gross fixed capital formation (GFCF), changes of stocks (CS), exports, imports (M) and 'total productive input' (TPI).

Note that the Total Intermediate Demand column of Table A2.1 represents the vector of intermediate demands, **AX**, in equation 1. The column Total Output is the vector of gross outputs, **X**, in equation 1. The column Total Final Demand is the vector of final demand, **Y**, in equation 1.

To derive the '**A**' matrix of equation 1, each of the z_{ij} elements of Table A2.1 are divided by the appropriate column sums, X_j. The column sums X_j are provided in Table A2.1 by the Total Input (TI) row. It should be noted that the column sum X_j is the sum of all inputs; those of both the production and payments sectors. The resulting '**A**' matrix is provided in Table A2.2. Creation of the Leontif inverse matrix $(\mathbf{I} - \mathbf{A})^{-1}$ follows from derivation of the '**A**' matrix, and is given in Table A2.3.

The residual discharge coefficient matrix **v** is given in Table A2.4. This table provides discharge coefficients v_{kj} for air pollution (particulate matter, sulphur oxides, nitrogen oxides, volatile organic compounds, carbon monoxide), and water pollution (biological oxygen demand, suspended solids, total dissolved solids, oil, nitrogen, and phosphorus) for each production sector plus the endogenised household sector. Residual coefficients are measured in tonnes per thousand pesos of a sector's total output. These coefficients were derived from a variety of local sources in addition to the World Health Organisation's Rapid Assessment of Sources of Air, Water, and Land Pollution (WHO, 1993).

Estimates of water effluent and air emission discharges are presented in Table A2.5. This table represents the total pollution matrix **V** in equations 4 and 5. The discharges are determined by multiplying the total regional output for each sector by the corresponding residual coefficient, as in equation 4.

This basic framework can then be used to develop scenarios and estimate changes in the outputs of both monetary flows in the economy and environmental residuals. Among the policy simulations carried out in the ENRAP study, for example, was the evaluation of the projected impacts of four alternative development scenarios on gross output and the environment. These scenarios involve changes in final demand, and the resulting impacts on residual generations were therefore determined using equation 6. These changes in the state of the environment can therefore form a part of assessment under the Pressure State Impact Response framework. The IO analysis is therefore useful in integrated modelling in the coastal zone with the constraints, as outlined above, being primarily on the spatial downscaling and the availability of data, particularly relating to the environmental coefficients.

Table A2.1 Example of a modified 12 x 12 industry IO table (Region XI, The Philippines, 1988).

SECTOR	Modified (in '000 pesos)											Total Intermediate
	1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH	Demand (+ Labour PCE)
1 agriculture	1,430,038	161	9,481	382	4,943,426	11,642	216	47	3,007	525	18,647	1,191,524	7,609,096
2 fisheries	2,589	301,646	396	1,075	426,468	4,138	36	4	1,285	3,496	17,383	433,634	1,192,150
3 forestry and hunting	197	0	382,811	8,875	1,028,264	2,666	8,723	930	113,037	1	13,478	166,380	1,725,362
4 mining and quarrying	4,624	1,776	901	11,226	15,605	331,407	98	79	92,551	3,306	18,391	41,821	521,785
5 manufacturing I	305,165	75,902	54,269	111,088	2,913,739	72,188	1,426	357	281,077	32,387	579,993	7,483,074	11,910,665
6 manufacturing II	230,259	65,551	56,590	228,250	232,868	347,868	6,358	1,521	307,598	189,475	208,922	167,214	2,024,474
7 electricity and gas	7,437	3,246	2,325	132	40,264	9,790	243	419	1,759	2,072	73,194	19,798	160,679
8 waterworks & supply	3,504	1,755	8	656	8,953	73	9	1	1,340	10,272	68,776	5,004	100,351
9 construction	10,397	10,454	297	31,616	20,785	3,178	934	20	498	4,703	126,082	4,643	213,607
10 transportation	158,642	41,181	23,816	40,230	238,190	43,460	108	271	65,607	116,353	583,696	646,094	1,957,648
11 other services	386,979	122,791	58,146	244,712	1,444,287	210,279	4,716	9,238	152,099	354,462	1,610,261	4,156,512	8,754,482
CE (HH)	5,839,675	480,807	932,063	305,350	1,128,466	174,401	20,613	20,338	443,887	307,364	4,662,733		14,315,697
TII Total Intermediate Inputs	2,539,831	624,463	589,040	678,242	11,312,849	1,036,689	22,867	12,887	1,019,858	717,052	3,318,823	14,315,697	36,188,298
M	1,532,830	588,241	316,697	1,155,305	3,227,324	1,524,355	33,931	11,918	1,108,113	986,231	3,037,189		13,522,134
CE	5,839,675	480,807	932,063	305,350	1,128,466	174,401	20,613	20,338	443,887	307,364	4,662,733		14,315,697
OVA	14,311,664	1,566,489	3,010,204	2,134,158	8,145,479	994,614	101,960	59,933	1,509,410	1,098,683	12,933,972		45,866,566
TPI	20,151,339	2,047,296	3,942,267	2,439,508	9,273,945	1,169,015	122,573	80,271	1,953,297	1,406,047	17,596,705		60,182,263
TI Total inputs	24,224,000	3,260,000	4,848,004	4,273,055	23,814,118	3,730,059	179,371	105,076	4,081,268	3,109,330	23,952,717	14,315,697	109,892,695

Table A2.1. Continued.

Final Demand Matrix						Total Final Demand (TFD)	Modified Total Output (TO)	Sector
PCE Original (Modified)	GCE	GFCF	CS	Exports	M			
2,316,478	0	123,001	(29,962)	15,396,911	0	17,806,428	24,222,726	1
843,041	0	5,092	(270)	1,653,621	0	2,501,484	3,253,416	2
323,464	0	8,781	(76,113)	3,032,890	0	3,289,022	4,848,004	3
81,305	0	0	(14,862)	3,726,648	0	3,793,091	4,273,055	4
14,548,074	0	112,255	(193,021)	4,919,219	0	19,386,527	23,814,118	5
325,085	0	469,505	(28,755)	1,088,964	0	1,854,799	3,730,059	6
38,490	0	0	0	0	0	38,490	179,371	7
9,729	0	0	0	0	0	9,729	105,076	8
9,026	0	1,816,774	0	2,046,504	0	3,872,304	4,081,268	9
1,256,091	0	48,720	0	492,965	0	1,797,776	3,109,330	10
8,080,803	765,658	512,542	0	9,995,744	0	19,354,747	23,952,717	11
							14,231,785	HH
27,831,586	765,658	3,096,670	(342,983)	42,353,466	(25,566,344)	73,704,397	109,800,926	Total
8,613,505	1,313,871	2,586,973	(470,139)	0	0	(13,522,134)	0	M
0	0	0	0	0	0	0	14,315,697	CE
0	0	0	0	0	0	0	45,866,566	OVA
0	0	0	0	0	0	0	60,182,263	TPI
36,445,091	2,079,529	5,683,643	(813,122)	42,353,466	(25,566,344)	60,182,263		TI

Note: 1. Modified TO is the value of total output adjusted for household production (forestry sector) and environmental damages (agriculture, fishery and household sector). Source: Orbeta *et al.* (1996)
TFD = PCE + GCE + GFCF + CS + E - M
TO = TID + TFD

Table A2.2 A' Matrix, ENRAP 12 x 12 industry IO table (Region XI, the Philippines, 1988).

SECTOR	Modified 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
1 agriculture	0.05903	0.00005	0.00196	0.00009	0.20758	0.00312	0.00120	0.00045	0.00074	0.00017	0.00078	0.08323
2 fishery	0.00011	0.09253	0.00008	0.00025	0.01791	0.00111	0.00020	0.00004	0.00031	0.00112	0.00073	0.03029
3 forestry and hunting	0.00001	0.00000	0.07896	0.00208	0.04318	0.00071	0.04863	0.00885	0.02770	0.00000	0.00056	0.01162
4 mining and quarrying	0.00019	0.00054	0.00019	0.00263	0.00066	0.08885	0.00055	0.00075	0.02268	0.00106	0.00077	0.00292
5 manufacturing I	0.01260	0.02328	0.01119	0.02600	0.12235	0.01935	0.00795	0.00340	0.06887	0.01042	0.02421	0.52272
6 manufacturing II	0.00951	0.02011	0.01167	0.05342	0.00978	0.09326	0.03545	0.01448	0.07537	0.06094	0.00872	0.01168
7 electricity and gas	0.00031	0.00100	0.00048	0.00003	0.00169	0.00262	0.00135	0.00399	0.00043	0.00067	0.00306	0.00138
8 waterworks and supply	0.00014	0.00054	0.00000	0.00015	0.00038	0.00002	0.00005	0.00001	0.00033	0.00330	0.00287	0.00035
9 construction	0.00043	0.00321	0.00006	0.00740	0.00087	0.00085	0.00521	0.00019	0.00012	0.00151	0.00526	0.00032
10 transportation	0.00655	0.01263	0.00491	0.00941	0.01000	0.01165	0.00060	0.00258	0.01608	0.03742	0.02437	0.04513
11 other services	0.01598	0.03767	0.01199	0.05727	0.06065	0.05637	0.02629	0.08792	0.03727	0.11400	0.06723	0.29035
CE (HH)	0.24107	0.14749	0.19226	0.07146	0.04739	0.04676	0.11492	0.19356	0.10876	0.09885	0.19466	0.00000

Source: Orbeta *et al.* (1996)

Table A2.3 Leontief Inverse matrix '(I-A)-1' for ENRAP 12 x 12 industry IO table (Region XI, the Philippines, 1988).

SECTOR	Modified 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
1 agriculture	1.13847	0.05642	0.06455	0.03396	0.29406	0.03265	0.04159	0.06168	0.05907	0.04195	0.06774	0.27307
2 fishery	0.01525	1.11276	0.01251	0.00644	0.03112	0.00662	0.00820	0.01247	0.01026	0.00951	0.01381	0.05594
3 forestry and hunting	0.01433	0.01095	1.09752	0.00911	0.06166	0.00694	0.06067	0.02140	0.04162	0.00818	0.01373	0.05108
4 mining and quarrying	0.00350	0.00466	0.00333	1.00923	0.00382	0.09982	0.00551	0.00423	0.03196	0.00886	0.00402	0.00817
5 manufacturing I	0.21334	0.16637	0.17600	0.10537	1.25864	0.09094	0.11427	0.16975	0.19152	0.12043	0.19493	0.74644
6 manufacturing II	0.02263	0.03414	0.02340	0.06591	0.02440	1.11459	0.04660	0.02643	0.09424	0.07786	0.02215	0.03920
7 electricity and gas	0.00169	0.00224	0.00165	0.00093	0.00308	0.00361	1.00226	0.00537	0.00169	0.00199	0.00445	0.00465
8 waterworks and supply	0.00086	0.00123	0.00058	0.00064	0.00113	0.00052	0.00048	1.00082	0.00094	0.00416	0.00372	0.00233
9 construction	0.00166	0.00463	0.00106	0.00832	0.00229	0.00249	0.00602	0.00163	1.00142	0.00298	0.00667	0.00395
10 transportation	0.02918	0.03093	0.02358	0.02057	0.02850	0.02334	0.01343	0.02288	0.03221	1.05444	0.04552	0.07971
11 other services	0.13958	0.13162	0.11366	0.11287	0.15086	0.11426	0.09442	0.19458	0.12078	0.19563	1.17773	0.44846
CE (HH)	0.32147	0.21931	0.26296	0.11622	0.18140	0.09907	0.16652	0.26505	0.17551	0.16674	0.26728	1.21828

Source: Orbeta *et al.* (1996)

Table A2.4 Matrix of residual coefficients for IO 12 x 12 matrix

Impact Variables	Sector 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
Residuals:												
PM	0.00001	0.00000	0.00008	0.00201	0.00045	0.00054	0.00044	0.00000	0.00053	0.00031	0.00002	0.00441
SO_x	0.00000	0.00008	0.00005	0.00092	0.00017	0.00025	0.00629	0.00000	0.00004	0.00019	0.00001	0.00002
NOx	0.00001	0.00015	0.00010	0.00056	0.00012	0.00017	0.00109	0.00000	0.00010	0.00028	0.00002	0.00016
VOC	0.00002	0.00005	0.00010	0.00045	0.00010	0.00011	0.00003	0.00001	0.00010	0.00047	0.00006	0.00690
CO	0.00011	0.00014	0.00058	0.00270	0.00060	0.00061	0.00011	0.00001	0.00053	0.00137	0.00013	0.03265
BOD5	0.00878	0.00000	0.07153	0.00000	0.00039	0.00007	0.00000	0.00000	0.00000	0.00000	0.00284	0.01196
SS	0.92180	0.00000	14.19494	1.86304	0.00038	0.00009	0.00541	0.00000	0.00000	0.00000	0.00300	0.00547
TDS	0.00000	0.00000	0.00000	0.00000	0.00281	0.00013	0.00002	0.00000	0.00000	0.00000	0.00000	0.00000
OIL	0.00000	0.00000	0.00000	0.00000	0.00003	0.00001	0.00000	0.00000	0.00000	0.00000	0.00014	0.00000
N	0.00482	0.00000	0.05502	0.00000	0.00001	0.00000	0.00000	0.00000	0.00000	0.00000	0.00010	0.00096
P	0.00005	0.00000	0.00087	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00003	0.00039

Source: Orbeta *et al.* (1996)

Table A2.5 Estimated matrix of residual discharges

Impact Variable	Sector 1	2	3	4	5	6	7	8	9	10	11	(Labour PCE) HH
PM	314	0	400	8,606	10,725	2,021	78	0	2,167	970	505	63,095
SO_x	88	259	242	3,941	4,026	924	1,129	0	182	589	252	346
NO_x	243	479	505	2,407	2,890	641	195	0	393	857	591	2,291
VOC	404	171	464	1,942	2,326	392	5	1	395	1,473	1,522	98,747
CO	2,744	445	2,792	11,558	14,378	2,283	19	1	2,176	4,263	3,210	467,368
BOD5	212,593	0	346,766	0	9,274	244	0	0	0	0	68,038	171,220
SS	22,329,619	0	68,817,134	7,960,853	9,111	345	971	0	0	0	71,780	78,353
TDS	0	0	0	0	67,003	501	3	0	0	0	0	0
OIL	0	0	0	0	703	40	0	0	0	0	3,290	0
N	116,814	0	266,743	0	145	8	0	0	0	0	2,383	13,805
P	1,322	0	4,214	0	0	0	0	0	0	0	631	5,521

Source: Orbeta *et al.* (1996)

Regional Input-Output model linkages to regional flux budget: applications in Merbok, Malaysia.

The basis of the I-O model to be applied to the Merbok site in Malaysia is to bring all energy into a standard form - i.e., the common currency of carbon. But, unlike the traditional multiplier analysis using the Leontief inverse, we adapted Johansen's (1974) multisector model by introducing and environmental capital component into the equation system. Following the suggestion made by Pearce and Turner (1990, p.153), the usual Cobb-Douglas production equation also included in the system was modified into a dichotomy between market inputs (a combination of labour and machines) and environmental inputs rather than, as usually is the case, between labour and capital.

This Johansen framework was presented at our October 1996 meeting in Hanoi and further debate led to the development of carbon budgets by assessing the flux embedded in the respective ecosystems which constitute our individual study areas. Carbon has an energy equivalent measured in calories or joule which makes it appealing as we attempt to examine how energy is transformed from one form into another as we undertake economic production and whether the rate of this transformation *vis-à-vis* the carbon flux is sustainable over the long term.

The problem is therefore how best to incorporate carbon fluxes into an economic-ecological framework thereby bridging the gap between the ecology in energy equivalent terms with economic behaviour operating in the market system. Input-output systems have much potential for such an application. The critical issue is to select a suitable unit of measurement (a numeraire) with which to capture energy flows.

Energy equivalents and economics

The survey by Faucheux and Pillet (1994) indicated three main views on energy valuation. The first involves estimating the ratio of energy to money (see Odum and Odum, 1981, p. 44) so that we can measure money in energy terms or vice-versa. This view is a misconception because energy does not have the same properties that money has. It is a mistake to think that energy and money are convertible from one to the other. Money can be transformed from one form of asset into another and back again. Fluctuations in money values encountered in the conversion process are not due to transformation losses as happens for energy due to thermodynamic laws but according to changing market demand and supply conditions.

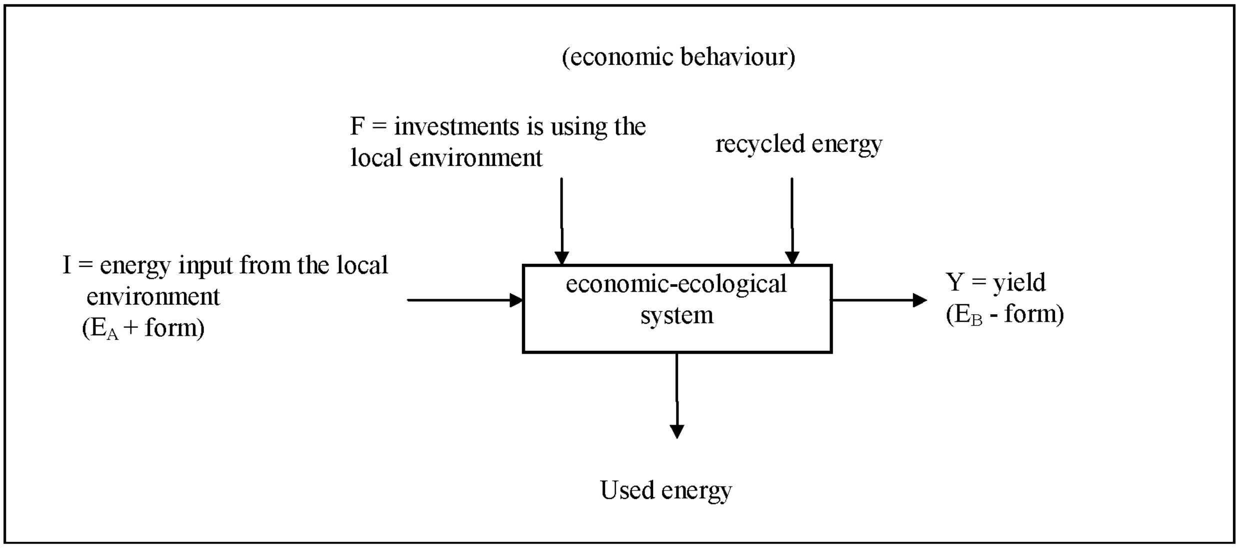
The second view concerns energy theories of value that attempt to attribute labour, materials, capital and all other production factors into energy terms. The limitation of this approach is that when we lose sight of the money values for these items we also lose sight of the price signals that affect how these items are brought into play within the production process. Thus, while an accounting of energy within the ecosystem is a useful inventory exercise, it will not help much when we wish to incorporate economic considerations that impact on the ecosystem.

The third view leaves energy and money as distinct entities and does not attempt to replace one by the other, but attempts to relate them. Economic activities are seen as a continuous transformation of low entropy energy sources into high entropy and in the process emits irreversible waste. Responding to this transformation of energy, composite indicators are developed that show to what extent a threshold is drawing near, beyond which the ecosystem will undergo a major change. The next section will discuss details of this view.

Energy metrics

The most effective way to make an assessment of the energy fluxes found in various forms within a local ecosystem is in terms of the solar energy that was used to produce them. All energy forms found are thus standardised in relation to solar energy, which is the embodied energy denoted as eMergy contained in the various forms of energy. The principle of this approach initiated by Odum (1983) is illustrated by Figure A2.1.

Figure A2.1. eMergy-energy relationship



As E_A^+, which is the embodied energy (or eMergy), is transformed through the process of economic activities into another form of energy E_B, we obtain an eMergetic balance by the ration E_B/E_A^+ expressed in joules by solar joules or emjoules. This ratio defines the solar transformity of E_B^- telling us the amount of E_A^+ incorporated in E_B^-. Both the first and second laws of thermodynamics are thus taken into account with respect to energy transformation and losses. The degree of solar transformity thus serves as a qualitative description of the ecosystem being assessed. The biomass of the local ecosystem expressed in E_A^+ emjoules indicate the amount of solar energy that had gone into generating this ecosystem.

To attempt bridging what we know of the energy state of the ecosystem, in terms of the degree of transformity, with economic production another term called monergy is introduced (see Pillet, 1994).

$$\text{monergy} = \frac{\text{eMergy}^{\text{nation}}\,(\text{emjoule})}{\text{GDP }(\$)}$$

This is a macroeconomic indicator which relates the total energy state of the country, that is eMergy in emjoules against the total economic production of the country in dollars.

Our intention is to estimate the unknown ecological price for a given hectare of land, say located within our study area, for a given year. If we assert that this price, $P_I(\$)$, in proportion to the country's total income, GDP(4), is exactly equal to the proportion of the energy inventory of that hectare of land to the total energy state of the country, that is:

$$\frac{P_I(\$)}{\text{GDP}(\$)} = \frac{\text{eMergy}^{\text{local}}(\text{emjoule})}{\text{eMergy}^{\text{nation}}(\text{emjoule})}$$

then, we can obtain an estimate of $P_I(\$)$ as follows:

$$P_I(\$) = \frac{GDP(\$).\ eMergy^{local}(emjoule)}{eMergy^{nation}(emjoule)}$$

$$P_I(\$) = eMergy^{local}(emjoule).\ \frac{GDP\ (\$)}{eMergy^{nation}}(emjoule)$$

$$P_I(\$) = eMergy\ ^{local}(emjoule).\ \ \frac{1}{monergy}$$

In other words if we can separately estimate the monergy of the country and if we perform an energy inventory of the local ecosystem in eMergy terms, we will be able to estimate the price of the local ecosystem, $P_I(\$)$.

The economy-cum-energy input-output model

The standard input-output model is established in the following way. Consider an n x n matrix $Z=\{z_{ij}\}$ of inter-industry flows expressed in millions of ringgit. Such flows only account for intermediate demands, i.e., purchases of industry outputs to be used as inputs into further production. Total output by the economy is an nx 1 vector $X=\{x_j\}$ obtained after adding final demands $Y=\{y_i\}$. In other words,

$$Z + Y = X$$

Analysis begins by calculating the intermediate inputs per dollar of output for each of the elements of Z to form a technical coefficients matrix $A = \{_{ij}\}$that is,

$$A = a_{ij} = \frac{Z_{ii}}{X_i}$$

Since Z = AX, we have

$$AX + Y = Y$$
$$Y = (I - A)\ X$$

and therefore,

$$X = (I - A)^{-1}Y$$

This equation is called the Leontief inverse. It forecasts the level of economic activity given by the direct, indirect and induced economic impacts, X, for the different industry sectors given some assumptions or scenarios of the pattern of final demand Y under a given set of technological structures fixed by the technical coefficients set $(I\text{-}A)^{-1}$.

There are many versions of economic-cum-energy variation of this model. The one which was built for analysis in this paper is based on Miller and Blair (1985). It used revised form so matrices we will call Z^*, Y^* and X^* in which are contained energy flows in energy units alongside industry flows in ringgit. In addition a diagonal matrix of total energy consumption, F^* is established.

The following are defined

$$\delta = F^* (X^*)^{-1} A^*$$

$$\delta = F^* (X^*)^{-1} (I - A)^{-1*}$$

Here , X^* is diagonalised matrix of the otherwise n x 1 vector containing both energy and non-energy sectors to facilitate matrix multiplication. The resulting matrix indicated by δ shows the direct energy intensities by sectors. The matrix α shows the total energy intensities which incorporate secondary impacts made up of indirect and induced effects.

The matrices α and δ contain values identical to the A^* and $(I-A^*)^{-1}$ matrices respectively except that pre-multiplication by F^* and $(X^*)^{-1}$ removes the inter-industry money flows. Such flows are irrelevant here because they should be analysed under standard input-output analysis.

Regional tables expand on the national table by recording flows between sectors and between regions (see Miller and Blair, 1985). To simplify the regional table, concern is only given to flows inside the region; flows with the rest of the country are considered as another composite region.

Incorporating the environment into the input-output framework is complicated by the need to introduce an elaborate set of environmental sectors which have indicated flows among themselves and among these environmental sectors with the various economic sectors. Furthermore there is the need to resolve the units of measurement for the environmental sectors.

Within the eMergy concept, the environmental component needed on the input-output table is reduced to one sector flowing out as eMergy, E_A^+ and becomes energy, E_B^-. A sketch of the input-output framework is shown in Figure A2.2.

Figure A2.2. Economic-eMergetic input-output table framework.

	local economic sectors	E_B - local	economic sectors in rest of country	E_B - rest
local economic sectors	A	B	C	D
E_A + local	E	F	G	H
economic sectors in rest of country	I	J	K	L
E_A + rest	M	N	O	P

In figure A2.2, the usual inter-industry flows within the locality are entered into A and the economic investments into the energy transformation process of the local environment (referred to as F in the figure) goes into B. In C and D economic inputs affecting the rest of the country are entered. E contains data on eMergy inputs into economic production in the locality while F records transformation losses involved from E^+_A to E^-_B. Again, G and H are meant for interactions from the locality to the rest of the country. The remaining parts of the table contain similar inputs but this time dealing either with flows within the rest of the country or from the rest of the country into the locality.

Input-Output Coefficients

The first step to input-output analysis is to transform the above table format into what is called a technical coefficients table by dividing the column entries by gross economic output in dollar terms. The resulting entries become input-output flows per dollar of gross output. Notice that wherever the nominators are in dollars, we obtain the usual input-output coefficients. Wherever the nominators are in eMergy terms, the coefficients become monergy values. Thus from equations described above, environmental prices of the local ecosystem can be expressed as its total eMergy divided by monergy values on the coefficients table.

Beyond such descriptive indicators, standard input-output analysis procedures can be introduced from which we obtain secondary and induced impacts based on the Leontief inverse and the interconnectedness between input and output sectors based on Rasmussen's power and sensitivity indices.

Appendix 3. Monetary Valuation Methods and Techniques

Alternative and appropriate methods

The state to impact and impact to policy response model linkages require that ecosystem changes with direct or indirect effects on human welfare (i.e. well-being in terms of income and wealth creation and quality of life, including health effects) be evaluated in order to determine their magnitude and significance. Monetary valuation methods and techniques provide one approach to the evaluation of impacts exercise. They can be deployed in any of the three resource assessment categories (impact analysis, partial valuation and total valuation) defined in section 3.4.

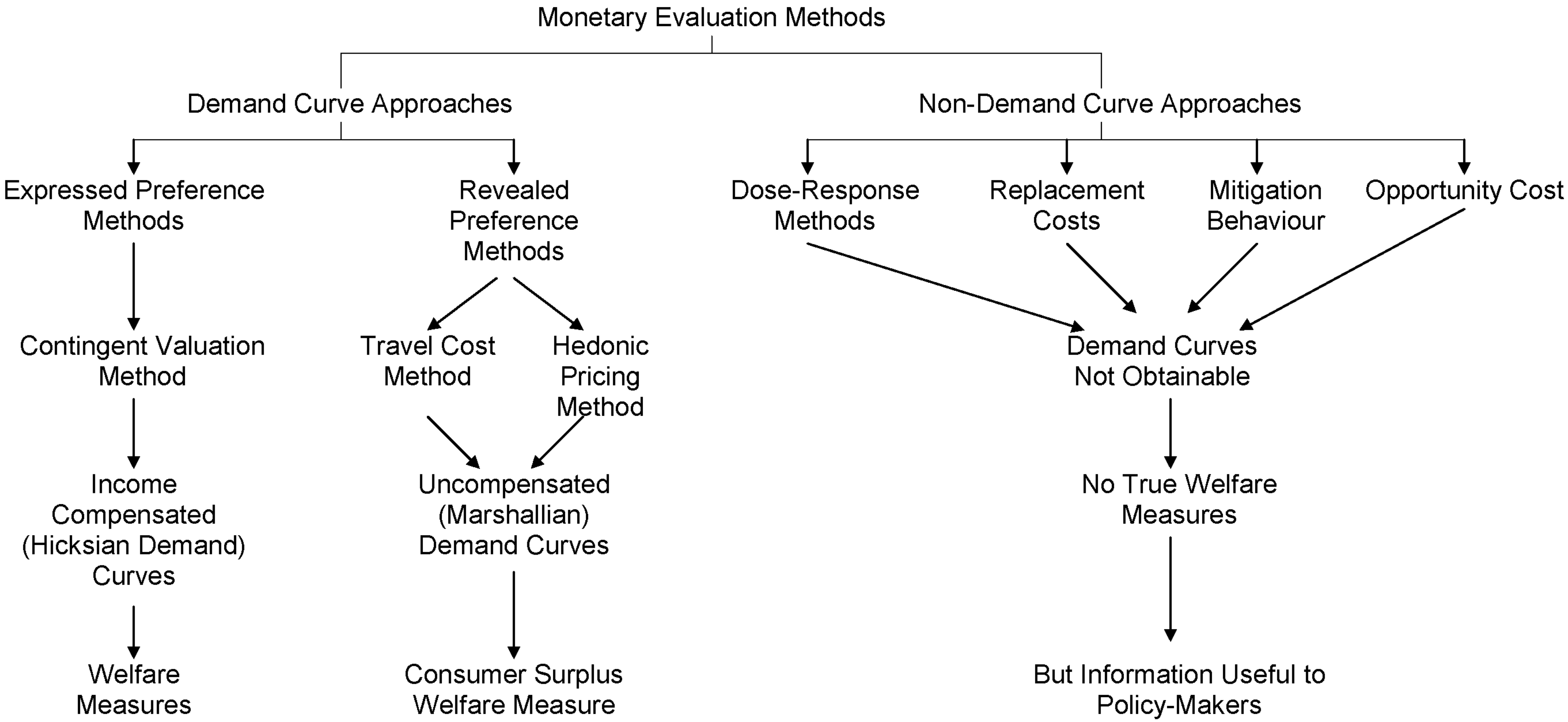
The environmental effects that require valuation can be classified into productivity changes, health effects, amenity gains and losses and assets existence value conservation or loss. Each of these effects is amenable to particular valuation methods as discussed in section 3.4 (see Table 3.8). A general survey of these valuation methods, plus some outline case study examples that have utilised such methods is provided in Turner and Adger (1996). In this appendix the methods themselves are reviewed in more detail and the text is supported by relevant references to empirical studies.

As reported in Turner and Adger (1996) it is possible to divide the monetary valuation methods into demand-curve approaches and non-demand curve approaches, as outlined in Figure A3.1. The former are more strictly valid in economic theory terms, but the latter are sometimes the only option because of data deficiencies and conceptual problems. While non-demand curve approaches are not capable of providing valid economic welfare estimates, they do provide useful monetised information on impact significance and are widely used in representing the relative importance of many environmental goods and services (see Dixon and Sherman, 1990 for a review). So, for example, the replacement cost of the loss of soil due to erosion from agricultural land may involve the market cost of fertilisers to replace the soil fertility so that productivity is maintained. But this replacement cost estimate does not reflect the demand for soil fertility by the farmer and is somewhat hypothetical. The cost to the farmer is best represented as the actual value of the loss of production of agricultural outputs from the less fertile land, rather than the hypothetical cost of replacement of soil. The following sections first outline the non-demand curve approaches and then concentrate on methods within the demand curve approaches to valuation, namely the travel cost method, the hedonic pricing method and the contingent valuation method. Further information on these can be found in Bateman (1993), Bateman and Turner (1993), Barbier (1994), Turner *et al.* (1994) and other texts and examples of applications of these in Adger *et al.* (1995), who attempt to aggregate total economic value for forest resources, and other examples highlighted in the text.

Non-demand curve approaches

The opportunity cost method quantifies what society has to give up if an environmental resource such as a wetland or a coral reef is to be conserved. An estimate of the monetary costs of the conservation option (the social opportunity cost) is made in terms of the alternative development option that is given up e.g. alternative uses for the wetland as drained farm land, or industrial, housing, or port facilities. The development option is assessed in economic terms in order to determine what net economic benefit (if any) society would have to give up when deciding to favour the environmental asset conservation option.

Figure A3.1 Demand curve and non-demand curve methods for the monetary evaluation of the environment



The replacement cost method examines the functions that a given environmental system provides, when it is operating in a 'normal' and 'healthy' state. It might then be possible to estimate what it would cost society if the system was lost or damaged, in terms of replacing some of the functions. A coastal wetland might, for example, be providing a storm buffering function and a nutrient sink function. If the wetland was converted to some industrial or other use, society would need to invest in a replacement sea defence system (or an augmented existing system) and perhaps a sewage treatment plant.

Another variant of this approach involves estimating the costs of so-called shadow projects. Thus it might be possible to re-create the threatened wetland elsewhere in the same general area, or to restore existing nearby but already degraded wetlands. The costs of these shadow project options would then need to be estimated and fed into the policy process.

Statistical techniques can sometimes be used to relate differing levels of pollution (the 'dose') to differing levels of damage (the 'response'). They are known as dose-response functions. Applications in the coastal zone context would include fisheries and coral reef damage from pollution and water quality-related human health damage effects. Many of these techniques are dealt with in Dixon and Sherman (1990) and Turner *et al.* (1994).

Travel Cost Method (TCM)

The TCM evaluates the recreational use value of resources, hence measures one aspect of indirect use values. The Travel Cost Method is a survey technique, whereby visitors to a site are asked a series of questions to ascertain their place of residence; necessary socio-economic information; frequency of visits to the particular and other similar substitute sites; means of travel; and cost information about the trip. From these data visit costs can be calculated and related to visit frequency so that a demand function can then be used to estimate the recreation value of the whole site.

The method was developed in the 1960s in the US for estimating the value of outdoor recreation, particularly as information for management of national parks and other assets. The method is somewhat restricted in the range of impacts and changes in which it can provide economic values, but it is of use in the estimation of value in coastal environments where recreational use of beaches and other resources represents a significant demand. Examples of travel cost estimates include those for beach resorts, where the quality of the beach affects demand, hence the environmental quality has a marginal value (Bell and Leeworthy, 1990). Various estimates of recreational value of forests and non-coastal resources exist including Tobias and Mendelsohn (1991), Maille and Mendelsohn (1993) and Mercer *et al.* (1995) which all investigate the recreational value of forest resources in the tropics.

The value for a specific recreation site is estimated under this method by relating demand for that site (measured as site visits) to its price (measured as the costs of a visit). A simple TCM model can be defined by a trip-generation function such as:

$$V = f(C,X)$$

where V = visits to a site
C = visit costs
X = other socio-economic variables which significantly explain V.

The literature can be divided into two basic variants of this model according to the particular definition of the dependent variable V. The 'Individual Travel Cost Method' (ITCM) simply defines the dependent variable as the number of site visits made by each visitor over a specific period, say one year. The Zonal

Travel Cost Method (ZTCM) on the other hand, partitions the entire area from which visitors originate into a set of visitor zones and then defines the dependent variable as the visitor rate (i.e., the number of visits made from a particular zone in a period divided by the population of that zone).

The ZTCM approach redefines the a trip-generation function as:

$$V_{hj}/N_h = f\,(C_h, X_h)$$

where V_{hj} = Visits from zone h to site j
N_h = Population of zone h
C_h = Visit costs from zone h to site j
X_h = Socio-economic explanatory variables in zone h

The visitor rate, V_{hj}/N_h, is often calculated as visits per 1,000 population in zone h.

The underlying theory of the TCM is presented with reference to the zonal variant, and discussion of the differences between this and the individual variant is presented subsequently before consideration of more general issues. Discussion of the ZTCM is illustrated by reference to a constructed example detailed in Table A3.1 which estimates the recreation value of a hypothetical site. The method proceeds in nine steps as follows:

Step 1 Data on the number of visits made by households in a period (say annually) and their origin is collected via on-site surveys.

Step 2 The area encompassing all visitor origins is subdivided into zones of increasing travel cost (column 1 of Table A3.1) and the total population (number of households) in each zone noted (column 2).

Step 3 Household visits per zone (column 3) is calculated by allocating sampled household visits to their relevant zone of origin.

Step 4 The household average visit rate in each zone (column 4) is calculated by dividing the number of household visits in each zone (column 3) by the zonal population (number of households; column 2). Note that this will often not be a whole number and commonly less than one.

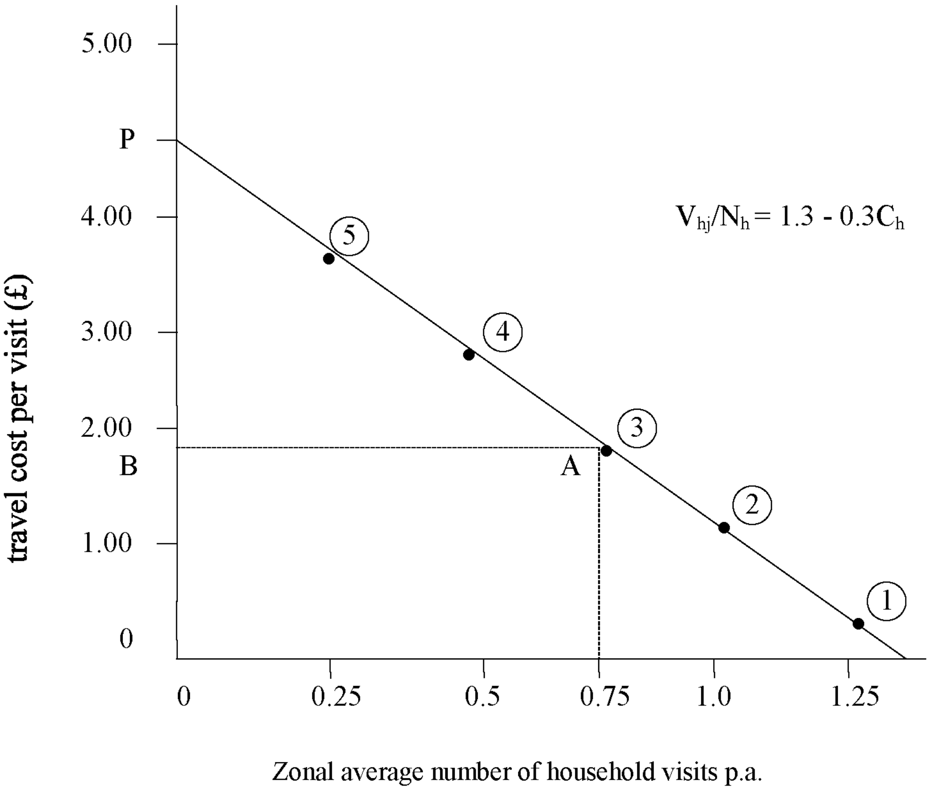
Step 5 The zonal average cost of a visit (column 5) is calculated with reference to the distance from the trip origin to the site.

Table A3.1 Worked example of consumer surplus estimates for recreation experience using zonal travel cost method

Column No.	1	2	3	4	5	6	7	8
	Zone No.	Zonal population (no. of households)[1] (N_h)	No. of household visits to site p.a.[2] (V_{nj})	Average no. of visits per household p.a.[3] (V_{nj}/N_h)	Average travel cost per household visit[4] (£) (C_h)	Consumer surplus per household all visits p.a. (£)	Consumer surplus per household per visit (£)	Total consumer surplus p.a. (£)
	1	10,000	12,500	1.25	0.16	2.60	2.08	26,040
	2	30,000	30,000	1.00	1.00	1.67	1.67	50,100
	3	10,000	7,500	0.75	1.83	0.94	1.25	9,400
	4	5,000	2,500	0.50	2.66	0.42	0.84	2,100
	5	10,000	2,500	0.25	3.50	0.10	0.40	1,000
				Total annual consumer surplus of the recreational experience			=	88,000

Notes: Trip generating function $V_{hj}/N_h = 1.3\text{-}0.3C_h$.
1. from census records.
2. from survey; annual totals derived by extrapolating from sample data according to available information regarding tourism rates.
3. column 4 = column 3/column 2.
4. either calculated with reference to zonal distance or via survey .

Figure A3.2 Demand curve for the whole recreation experience



Key: 1 = zone number 1.

Step 6 A demand curve is then fitted relating the zonal average price of a trip (travel cost) to the zonal average number of visits per household. This curve estimates demand for the 'whole recreation experience' rather than just the time spent on-site. In our hypothetical example this demand is explained purely by visit cost and the curve has the (unlikely) linear form given by:

$$V_{hj}/N_j = 1.3 - 0.3\ C_h$$

where V_{hj}/N_j = visit rate (average number of visits per household) from each zone
C_h = visit costs from each zone

Figure A3.2 illustrates this particular whole recreation experience demand curve. The estimation of this curve involves the implicit assumption that households in all distance zones react in a similar manner to visit costs. They would all make the same number of trips if faced with the same costs i.e. they are assumed to have identical tastes regarding the site.

Step 7 In each zone the household consumer surplus for all visits to the site (column 6) is calculated by integrating the demand curve between the price (cost) of visits actually made from each zone and that price at which the visitor rate would fall to zero (i.e. the vertical intercept of the demand curve at point P in Figure A3.2). Households in zone 3 for example would have a consumer surplus equal to area ABP for all their trips to the site:

$$\text{Consumer surplus for zone 3} = \int_{C_h=B}^{P} (1.3 - 0.3C_h).dC_h$$

Step 8 In order that annual total consumer surplus for the whole recreation experience can be estimated in each zone, total household consumer surplus must firstly be divided by the zonal average number of visits made by each household to obtain the zonal average consumer surplus per household visit (column 7). This can then be multiplied by the zonal average number of visits per annum (column 3) to obtain annual zonal consumer surplus (column 8).

Step 9 Cumulative annual zonal consumer surplus (column 8) across all zones gives our estimate of total consumer surplus per annum for the whole recreational experience of visiting the site.

These steps, leading to consumer surplus estimates, give a value of the recreational experience. There are various caveats to this value being used directly for any coastal resource. These caveats include whether there are substitute sites, whether the visitors are valuing particular attributes of the site; and whether the visitation rate and distance can be taken as an indication of recreational value (see Bateman, 1993). Given these caveats, the method is useful for determining recreational value of coastal resources. It is, however, ultimately limited in the scope of environmental impacts which can be addressed.

Hedonic Pricing Method

The Hedonic Pricing Method (HPM) relies upon the assumption that the local environmental quality (or lack of it) will determine the price of property and that differences in these prices can be taken as an indicator of marginal value of environmental change. The environmental factors, however, are only a subset of property price determinants which, for residential houses, may include amongst other factors the number of rooms and accessibility to shops and workplaces. The general specification of a hedonic price model is therefore:

HOUSE PRICE = f (ROOMS, ACCESS, ENVIRONMENT)

The equation states that house price is a function of (f) the number of rooms in the house (ROOMS), the distance in miles to local facilities from the house (ACCESS) and some measure of local environmental quality (ENVIRONMENT). If we were interested in valuing the environmental impact of local traffic noise then we could measure this in terms of decibels of traffic noise inside the houses in question.

We then need to measure each of the items HOUSE PRICE, ROOMS, ACCESS and ENVIRONMENT for a large number of houses so that we can begin to see how, on average, house price changes when each of the influencing factors change. We would expect house price to rise as the number of rooms increase; that house price would fall as the distance to local facilities rises, and finally, for house price to fall as the traffic noise increased, i.e. a typical demand curve relationship. This is indeed the results obtained in an US study of road noise. The following table (A3.2) shows the average percentage fall in house price which corresponded to a one unit increase in traffic noise in a number of US areas.

As an example, if a new road scheme was likely to raise traffic noise by one unit in Washington DC, then a monetary value for this increased noise pollution could be found by taking 88 percent of average house prices in the affected area.

The hedonic method has also been applied to the impact of water frontage, amenity, and other quality factors on house prices. This technique has been utilised in coastal areas to examine the impact of proximity to beach property, and hence derive a value for that environmental good (see Brown and Pollakowski, 1977). For a summary of the methods and applications see Brookshire *et al.* (1982) Garrod and Willis (1992), Freeman (1979) and Bateman (1993). Again this method, as with the travel cost method, is limited in its applicability to valuing the impacts of environmental change in coastal zones. The method is data-dependent and can only reasonably be applied where the environmental asset under consideration is well understood within the purchasing decisions of house or property owners. However, it can give some estimates of both the availability of recreational assets, as well as the impact of risk of inundation or flooding in coastal areas.

Table A3.2 The impact of traffic noise on house prices in the US

City	% fall in house price due to a one unit increase in noise
North Virginia	0.15
Tidewater	0.14
North Springfield	0.18 - 0.50
Towson	0.54
Washington DC	0.88
Kingsgate	0.48
North King Country	0.40
Spokane	0.08
Chicago	0.65

Source: Nelson (1978)
Note: Traffic noise measured as the equivalent continuous sound level (in decibels) which would have the same sound energy over a given period as the actual fluctuation sound level measured at houses in the study.

Contingent Valuation Method

The Contingent Valuation Method (CVM) is a method for placing monetary values upon assets and impacts which do not have market prices. It achieves this by constructing a hypothetical market and asking individuals, for example, what they are willing-to-pay (WTP) towards preservation of a particular environmental good. Therefore CVM relies upon individuals' expressed preferences (rather than the revealed preferences indicated by market prices).

The advantages include that the method allows us to ask questions about and estimate both use and non-use values (see Figure 3.7) and provides direct Hicksian welfare measures, which overcome some of the problems with non-demand curve approaches to valuation, as discussed above. The disadvantages of the method are that respondents may not believe in the credibility of the hypothetical markets; and that without an actual market place, stated WTP may not equate to what would actually be paid. The flexibility of these techniques in valuing many aspects of environmental quality have led to a voluminous literature on this subject (e.g. reviewed in Mitchell and Carson, 1989; Cropper and Oates, 1992), as well as voluminous critiques of the method and economic valuation more generally (e.g. Sagoff, 1998). Part of the controversy stems from the influence that this technique now holds in determining liability for damage assessment in the US legal system, to the extent that the estimation of damages from the oil spill of the Exxon Valdez in Prince William Sound, Alaska in the early 1990s was partially determined by using a CVM survey.

How to carry out a CVM

The steps in applying the contingent valuation method are set out in note form as follows:

Step 1 Preparation of survey and study

Set up the hypothetical market: individuals may be presented with two basic variants:
- How much are you willing to pay (WTP) for a welfare gain?
- How much are you willing to accept (WTA) in compensation for a welfare loss?

Define elicitation method. The major alternatives are:
- Open ended; 'how much are you willing to pay?' (this produces a continuous bid variable and may therefore be analysed using least squares approaches).
- Take-it-or-leave-it (dichotomous choice); 'are you willing to pay £X?' (this produces a discrete bid variable and requires logit-type analysis).

Other elicitation methods include the use of payment cards and bidding games with suggested starting points. Provide information regarding the quantity/quality change in provision of the good; who will pay for the good; and who will use the good. Define the payment vehicle, for example: higher taxes; entrance fees; or donation to a charitable trust.

Step 2 The survey. Methods include: on site (face to face); house to house (face to face) and mail/telephone (remote) survey techniques. Each of these has its advantages and cost and resource implications. In considering this step the guide by Mitchell and Carson (1989) is illuminating.

Step 3 Calculate mean willingness-to-pay. This calculation depends on whether an open-ended or dichotomous choice willingness-to-pay question has been asked. The following calculation is made in each of these circumstances:

Open ended	simple mean
	trimmed mean (removing outliers)
Dichotomous choice	expected value

Step 4 Estimate the bid function. Most CVM studies will attempt to investigate respondents WTP bids by estimating a bid function. A simple example might by:

$$WTP_{1j} = f(Q_{ij}, E_j, Y_i, S_i, X_i)$$

where Q_{ij} = visits by individuals to site j
WTP_{ij} = individual I's willingness to pay for asset j
E_j = characteristics of site j
Y_i = income of individual i
S_i = relevant socio-economic characteristics of individual i
X_i = other explanatory variables.

Step 5 Aggregation from the mean willingness to pay to gain an overall estimate of value. Calculating total WTP from mean WTP can involve, for example, multiplying the sample mean WTP of visitors to a site by the total number of visitors per annum.

Step 6 Testing the validity and reliability of the estimates produced. This final stage of any CVM study is the most important when the interpretation of the results and their applicability to other environmental goods, or in other situations is important. As discussed in Section 4 on scaling issues, the validity of CVM results is dependent on the acceptability of the hypothetical market by the respondents: attempting to transfer such estimates to other situations with different market and cultural circumstances may not be appropriate.

The numerous CVM studies and the diversity of approaches and goods and services for which values have been elicited using the CVM require careful scrutiny if values are to be compared or transferred to other sites, or policy decisions are to be made on their basis. As a result of the use of such results in the US in the legal process, there has been a call for standardisation. In Box A3.1 the protocol for CVM studies developed by NOAA in the US are outlined. These have been used and tested by many CVM researchers (see Carson, 1997), but should only be taken as a guideline for practice, since they have been developed in the social and cultural context of the US where, for example, referenda on public expenditure on public good provision are regular and hence survey respondents may be familiar with the hypothetical questions used in CVM surveys.

An example of applying the CVM to valuing river water quality improvements

The Monongahela River is a major river flowing through Pennsylvania. Desvousges *et al.* (1987) asked a representative sample of households from the local area what they would be willing to pay in extra taxes in order to maintain or increase the water quality in the river. The analysts conducted several variants of the CVM survey. In one variant households were presented with three possible water quality scenarios and simply asked how much they were willing to pay for each. The scenarios which were described to each respondent group were:

- Scenario 1: Maintain current river quality (suitable for boating only) rather than allow it to decline to a level unsuitable for any activity including boating.
- Scenario 2: Improve the water quality to a level where fishing could take place.
- Scenario 3: Further improve water quality from fishable to swimmable.

Amongst the households surveyed some used the Monongahela river for recreation while others did not. The analysts therefore could look at how much the users were willing to pay compared to the responses of non-users. Results for the sample as a whole were also calculated. Table A3.3 presents the willingness to pay of users, non-users and the whole sample for each proposed river quality change scenario.

Table A3.3 Willingness to Pay (WTP) for river quality scenarios along the Monongahela River, USA.

Water quality scenario	Average WTP of whole sample ($)	Average WTP of users group ($)	Average WTP of non-users group ($)
Maintain boatable river quality	24.50	45.30	14.20
Improve from boatable to fishable quality	17.60	31.30	10.80
Improve from fishable to swimmable quality	12.40	20.20	8.50

Notes: Full details given in Desvousges *et al.* (1987).

Box A3.1 NOAA Panel Protocol for Contingent Valuation Studies

General Guidelines

1. Sample Type and Size: Probability sampling is essential. The choice of sample specific design and size is a difficult, technical question that requires the guidance of a professional sampling statistician.
2. Minimise Non-responses: High non-response rates would make CV survey results unreliable.
3. Personal Interview: It is unlikely that reliable estimates of values can be elicited with mail surveys. Face-to-face interviews are usually preferable, although telephone interviews have some advantages in terms of cost and centralised supervision.
4. Pre-testing for Interviewer Effects: An important respect in which CV surveys differ from actual referendum is the presence of an interviewer (except in the case of mail surveys). It is possible that interviewers contribute to 'social desirability' bias, since preserving the environment is widely viewed as something positive. In order to test this possibility, major CV studies should incorporate experiments that assess interviewer effects.
5. Reporting: Every report of a CV study should make clear the definition of the population sampled, the sampling frame used, the sample size, the overall sample non-response rate and its components (e.g., refusals), and item non-response on all important questions. The report should also reproduce the exact wording and sequence of the questionnaire and of other communications to respondents (e.g., advance letters). All data from the study should be archived and made available to interested parties.
6. Careful Pre-testing of a CV questionnaire: Respondents in a CV survey are ordinarily presented with a good deal of new and often technical information, well beyond what is typical in most surveys. This requires very careful pilot work and pre-testing, plus evidence from the final survey that respondents understood and accepted the description of the good or service offered and the questioning reasonably well.

Guidelines for Value Elicitation Surveys

7. Conservative design: When aspects of the survey design and the analysis of the responses are ambiguous, the option that tends to underestimate willingness to pay is generally preferred. A conservative design increases the reliability of the estimate by eliminating extreme responses that can enlarge estimated values wildly and implausibly.
8. Elicitation Format: The willingness-to-pay format should be used instead of compensation required because the former is the conservative choice.
9. Referendum Format: The valuation question generally should be posed as a vote on a referendum.
10. Accurate Description of the Program or Policy: Adequate information must be provided to respondents about the environmental program that is offered.
11. Pretesting of Photographs: The effects of photographs on subjects must be carefully explored.
12. Reminder of Substitute Commodities: Respondents must be reminded of substitute commodities. This reminder should be introduced forcefully and directly prior to the main valuation to assure that the respondents have the alternatives clearly in mind.
13. Temporal Averaging: Time dependent measurement noise should be reduced by averaging across independently drawn samples taken at different points in time. A clear and substantial time trend in the responses would cast doubt on the 'reliability of the value information obtained from a CV survey.
14. 'Non-answer' Option: A 'non-answer' option should be explicitly allowed in the addition to the 'yes' and 'no' vote options on the main valuation (referendum) question. Respondents who choose the 'no-answer' option should be asked to explain their choice.
15. Yes/No Follow-ups: Yes and no responses should be followed up by the open-ended question: 'Why did you vote yes/no?'
16. Cross-tabulations: The survey should include a variety of other questions that help interpret the responses to the primary valuation question. The final report should include summaries of willingness to pay broken down by these categories (e.g., income, education, attitudes toward the environment).
17. Checks on Understanding and Acceptance: The survey instrument should not be so complex that it poses tasks that are beyond the ability or interest level of many participants.

Source: Adapted from the report of the National Oceanic and Atmospheric Administration Panel on the Contingent Valuation Method (Arrow *et al.*, 1993).

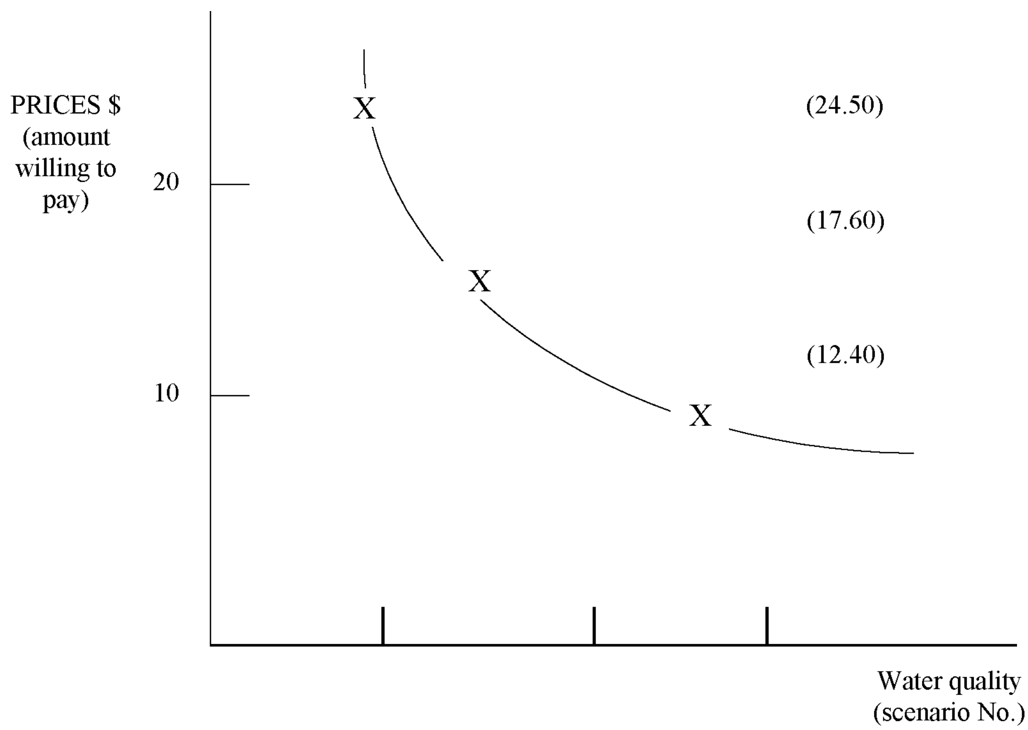
Households were told that the payment vehicle (the increased tax) would either be as a direct tax (e.g. income tax) or an indirect tax (e.g. a tax upon purchases such as VAT).

A number of conclusions can be drawn from these results. Considering the results for the whole sample we can see that the stated WTP sums draw out a conventional demand curve for water quality i.e. people are prepared to pay a relatively high amount for an initial basic level of quality. However, they are prepared to pay progressively less for higher levels of water quality. Figure A3.3 draws out the demand curve indicated by the results for the whole survey, representing the demand for the average household.

From this demand curve we could attempt to calculate the total value of environmental quality at the river. More importantly the value gain experienced by the average household when a water quality improvement is achieved could be derived. The total benefit value of a specific improvement could then be estimated by multiplying this average household value by the number of households which it is thought would be affected by such an improvement. This benefit can then be compared against the cost of achieving such a quality improvement to see if it was worthwhile.

Turning to results for the users and non-users group, both map out conventional downward sloping demand curves. Furthermore, as would be expected, at every quality level the willingness to pay of the users group exceeds that of the non-users, which again would be expected in economic theory.

Figure A3.3 Demand curve for water quality along the Monongahela River derived from contingent valuation data



Source: based on data in Desvousges *et al.* (1987).

Finally notice that the WTP of non-users is not zero. This is due to the fact that such households, while not personally wishing to visit the river, nevertheless do value its continued existence and even upgrading so that others can enjoy its benefits. This non-use existence value (see Figure 3.7) derives from people's altruistic public preferences showing that the concentration upon people's 'private preferences' as demonstrated by the market prices of marketed goods does not always fully capture the entire range of values which people have for things.

Constraints, biases and difficulties with the CVM method

There are a number of methodological issues in CVM outlined in Figure A3.4. These are dealt with in detail in many texts on CVM and are outlined only briefly here. Concentration of effort in the design phase of CVM studies on these difficulties can make the results more robust. However, the elimination of all 'biases' in Figure A3.4 is a misnomer, in that there is no true unbiased value for any asset under this technique: all values are *contingent* on the circumstances and the information provided concerning the hypothetical market.

Will respondents answer honestly? Free riding. If the individual has the opportunity to, in effect, name their own price for a good (as in the open-ended WTP approach) then economics predicts the individual will pretend to have less interest in a given collective activity than he really has, which is known in economics as free-riding. A number of CVM-type experiments have examined the extent of free riding by comparing individuals' stated WTP with what they actually paid for a good.

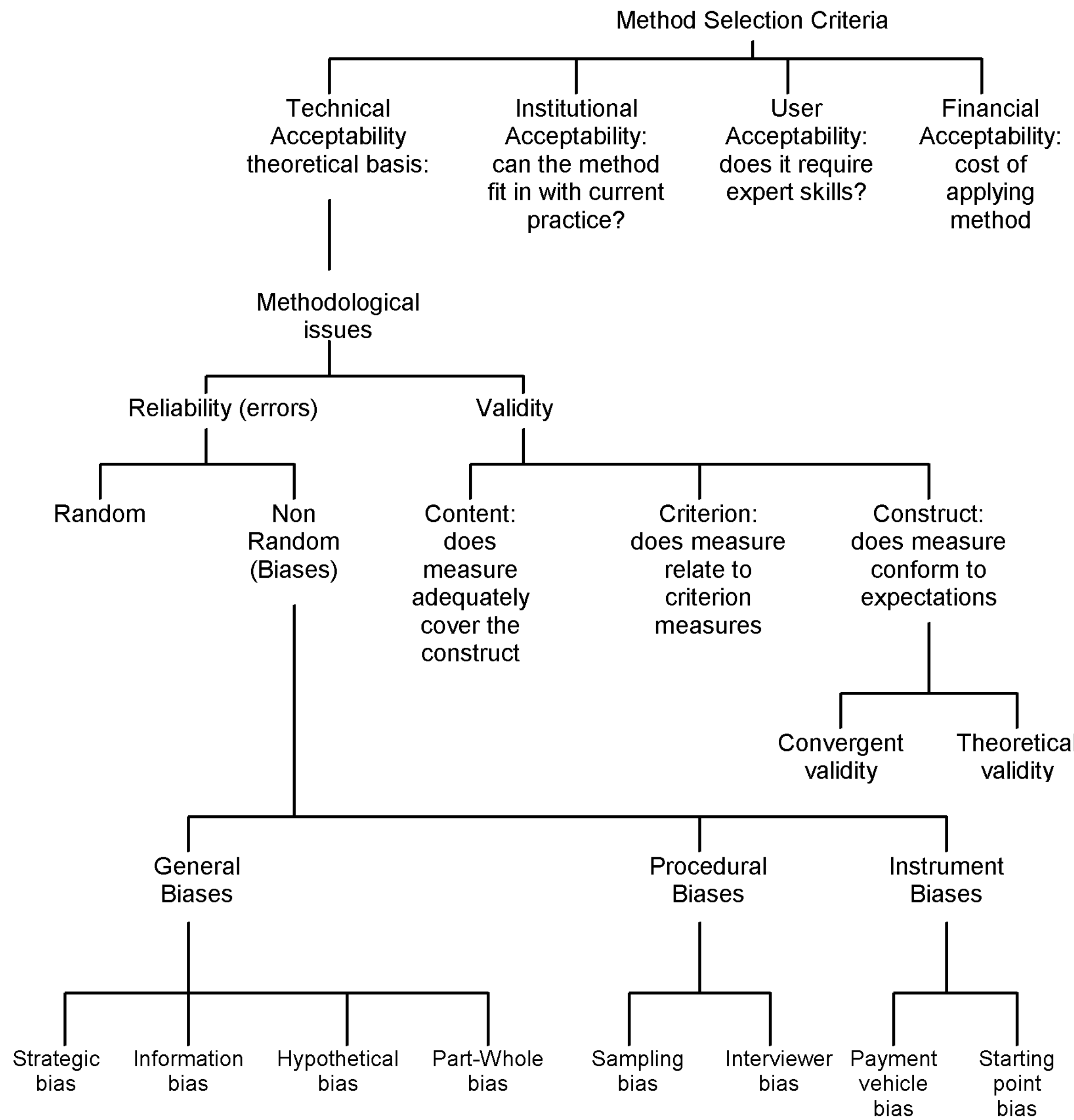
In a classic series of related experiments, US researchers examined the statements and actual payments of recreational hunters with regard to their hunting permits (e.g. Bishop and Heberlein, 1979). In testing a willingness to accept (WTA) approach, Bishop and Heberlein note that, while only 40% of hunters stated that they would be willing to accept $50 in return for losing their recreational hunting, over 90 percent subsequently accepted a $50 cheque in return for cancelling their permit. In other words, true WTA was considerably lower than stated WTA.

In one test of a WTP format, one group of hunters were asked to state how much, hypothetically, they would be WTP for a permit while a second sample were offered permits at various prices with actual payment required to determine true WTP. Using the payment vehicle of a sealed bid auction submitted by mail, the stated WTP was $24 while the actual WTP was $32. Therefore stated WTP was 75 percent of actual WTP i.e. a weak free-rider effect was detected.

It is, by definition, impossible to repeat this experiment for a pure public good. However, these results indicate that the responses obtained from open-ended WTP format studies do provide a reasonable lower bound estimate of true WTP (true welfare measure) i.e. free riding does not completely invalidate such an approach (however WTA formats perform badly in these tests). It has been pointed out that dichotomous choice approaches (as recommended in the NOAA protocol in Box A3.1) should limit possibilities for free-riding behaviour (Hoehn and Randall, 1987).

Strategic Overbidding. Conversely to the free-riding incentive, some respondents, perceiving that analysts are interested in mean WTP, may overstate their WTP in an effort to inflate the mean and so improve the prospects for provision of the good in question. Bateman *et al.* (1993) speculate on the possibility of strategic overstatement in their open-ended WTP responses with respect to a survey on the preservation of landscape assets in the Norfolk Broads area in the UK. Here, truncation of the top 5 percent of bids resulted in a drop in mean WTP of over 30 percent, perhaps suggesting that a small number of survey respondents can inflate the mean WTP by strategic bidding to enhance the value of the environmental good to the policy maker who is using the results. However, such a result is a poor test of strategic overbidding.

Figure A3.4 Criteria for the selection of a monetary evaluation method and issues within the validity of contingent valuation studies.



Part Whole Bias. Kahneman and Knetsch (1992) report no significant differences in WTP from a sample of respondents asked to value a small number of Canadian lakes and another sample asked for their WTP for all lakes. One would expect that there would be significant differences in this case as the value of one lake should be less than the value of all lakes. This indicates two problems: respondents have difficulty in separating out one aspect of a larger asset; and respondents have difficulty taking into account available income and other demands when making their WTP bids. This issue can be overcome in survey design by asking respondents to calculate a total yearly budget for all environmental issues and then to consider all the other demands upon this budget before asking the WTP for the good in question.

Other biases. These include issues such as information effects on the responses: the values are dependent on the type and presentation of the information about the good in question, particularly if respondents are non-users, or even of the demeanour and cultural setting of the interviewer. The issue of whether to include non-users as the relevant population across which the mean WTP should be aggregated to give an aggregate WTP is a further issue of some debate.

Two further issues in CVM surround the means by which the hypothetical WTP is presented: some respondents may be directly opposed to taxation on principle, so asking WTP for environmental protection as determined by a contribution to taxes may result in many zero bids, under-representing the underlying WTP of respondents. Alternative so-called payment vehicles in CVM include hypothetical contributions to taxes, entrance fees (for recreation sites and other location-specific assets), or a contribution to a trust fund specifically set up to bring about the environmental goal being specified in the survey. Studies have shown that changes in the method by which respondents would have to pay (the payment vehicle) result in changes to mean WTP. Bateman *et al.* (1993) suggest that much of this can be attributed to varying rates of refusal to pay.

The same study also highlights differences in estimates of WTP resulting from different so-called elicitation methods. In the study valuing landscape preservation in the Norfolk Broads (Bateman *et al.*, 1993) used three methods to elicit WTP responses:

- open ended 'What are you willing to pay?'
- dichotomous choice 'Would you pay £X?' (X is varied across sample)
- iterative bidding: asking a series of yes or no questions to arrive at a refined WTP.

The differences in the resulting estimates of WTP are illustrated in Table A3.4.

Table A3.4 Estimates of willingness to pay for recreation and amenity for Norfolk and Suffolk Broads, UK.

	Sample size	Mean WTP[1]	Median WTP (£)	Std devn	S.E. mean	Min bid (£)	Max bid (£)
Open Ended WTP Study	846	67.19	30.0	113.58	3.91	0.0	1250.0
Iterative Bidding WTP Study	2051	74.91	25.0	130.1	2.87	0.0	2500.0
Dichotomous Choice WTP Study	2070	140	139	n/a	n/a	n/a	n/a

Notes:
1. Includes as zeros, those who refused to pay anything at all.
Source: Adapted from Bateman *et al.* (1993).

There is therefore clear evidence from these results of upward anchoring in the dichotomous survey, by which the first amount asked would be responded to positively by respondents even if their WTP was somewhat lower. This effect, however, potentially compounds free-riding in the open-ended study, where, as outlined above, there may be upward bidding by some respondents. Different elicitation methods lead to different respondent characteristics. Dichotomous choice approaches appear to result in upward anchoring whilst open-ended elicitation approaches engendered downward free riding.

All of these effects in contingent valuation are important to consider at the design stage. As each of these is refined, the range of potential environmental goods and valuation issues to be addressed can be increased. The limitation of the technique, however, remains those outlined at the beginning of this section, namely that the validity and robustness of the estimates is dependent on the acceptance of a hypothetical market by respondents. Critics of the technique argue that this issue invalidates the technique both because decision-makers are led to create the markets which have been suggested as hypothetical in such studies, and because decisions on environmental preservation are separated in many respondents respondents' perceptions from market transactions (see for example Sagoff, 1998; Burgess *et al.*, 1997).

Conclusions on the contingent valuation and other valuation techniques

Arguments concerning the CVM are often put in terms of whether some estimate, however flawed, of monetary values of the environment are better than an absence of estimates. This is the view that economics *demonstrates* the value of ecosystem services and functions and hence leads to arguments about impacts and response (within the P-S-I-R framework) being taken with at least a partial view of economic dimensions of value, which are but one element: "because ecosystem services are not fully captured in commercial markets or adequately quantified in terms comparable with economic services and manufactured capital, they are often given too little weight in policy decisions" (Costanza *et al.*, 1997, p. 253).

Leaving aside the arguments over the role of valuation when applying these techniques, the review in this appendix has demonstrated that CVM is the most flexible technique in terms of the range of environmental quality changes which can be tackled, subject to these hypothetical markets being acceptable and familiar to survey respondents. However, evaluation of the technique itself stirs up the most vehement of rhetoric. This appendix takes a middle view that for use-values with well perceived environmental goods, CVM seem to give a valid order of magnitude estimate of economic use and non-use values. Such information is useful in the decision-making process. For non-use values, particularly where goods are poorly perceived or understood, the method may give a spurious air of economic analysis to what is, on the part of respondents, guesswork.

The range of techniques outlined in this Appendix is well-established, and the methods have been applied by a variety of researchers to coastal management issues. The critical issue of interpretation of these values, and indeed what the term value means in the context of decision-making, is highlighted in the main sections of the report and in Turner and Adger (1996, pp. 34-49).

Appendix 4. Glossary

Definitions supplied in this section are operational in the context of this report and simplified for communication with a broad audience.

Abatement cost curves	Calculation of the least costs to reduce a given environmental pressure by adopting available technical environmental measures.
Alternative Futures Scenarios	Similated sequence of events possible in the future.
Ambient	Relating to a condition of the environment that surrounds a body or object. Therefore, ambient environmental quality is to signify the quality of the surrounding environment.
Annualised (homogenised) data	Information that has been uniformed to relate to a period of one year.
Anthropic	Relating to humankind.
Anthropocentric	Regarding humankind as the central or most important element of existence.
Anthropogenic	Influenced or caused by activities of human beings.
Assimilative capacity	Ability of a system to incorporate and absorb substances of diverse chemical composition.
Benefit	Gain in economic and/or social welfare.
Benefits transfer	The transfer of economic valuation estimates across time and space
Bequest value	Value of a resource that will be preserved for future generations.
Biogenic	Resulting from activities of living organisms.
Biogeochemical cycle	Circulation of chemical components through the biosphere from atmosphere, hydrosphere and lithosphere.
Biome	Ecosystems across a landscape scale.
Budget model	Estimate of inventories, inputs and outputs of a system over a specified period of time, based on accounting for material balances and flows.
Capital	Resources, both man-made and natural.
Carbon cycle	Circulation and movement of carbon atoms through the biosphere, atmosphere, hydrosphere and lithosphere.
Carbon sequestration	Net accumulation of carbon via a range of terrestrial and marine processes.
Carbon storage	The stock of carbon accumulated in various sinks.

Catch crop	Crop serving to hold or restrain certain components usually free-moving in the soil.
Ceteris paribus	Latin expression for 'all other things being equal'.
Clean technology	Application of improved technology to production processes in order to decrease the resulting impacts on the environment (as opposed to end of pipe add-on technology).
Climate change	Alterations in the Earth's energy balance (mainly consisting of incoming and outgoing radiation that determine the surface temperature) and climate.
Coastal	Referring to processes or features of the coastal zone.
Coastal typology	Characterisation of regions of the coastal zone on the basis of specific related types and features constituting the objective of research. *See also 'typology'.*
Coastal zone	"Extending from the coastal plains to the outer edge of the continental shelves, approximately matching that region that has been alternatively flooded and exposed during sea-level fluctuations of the late Quaternary period" (Holligan, P. M. and de Boois, H. 1993. *Land-Ocean Interactions in the Coastal Zone (LOICZ) Science Plan*. IGBP Report no. 25, 50pp.). Note: LOICZ has at least other three possible definitions of coastal zone. The above relates to an area extending between +15 m and -150 m.
Continental shelf	Gentle sloping submerged platform, part of the continental margin, between the shoreline and the continental slope. Usually taken to be shallower than 200 metres.
Contingent Valuation Method (CVM) ***or simply, Contingent Valuation (CV)***	Expressed preference valuation method, whereby evaluations are derived from direct questioning of individuals to determine their willingness to pay (WTP) for a certain environmental good or service or what they are willing to accept (WTA) for its loss.
Cost	Loss in utility or practical use. Also, the price required or paid for acquiring, producing or maintaining something, usually expressed in monetary terms.
Cost effectiveness	Achieving one or more targets at lowest costs possible.
Cost-benefit analysis (CBA)	Appraisal of the total social and economic costs and benefits derived from the development of a project, programme or decision.
Cross section data	Economic observations taken at the same time.
Defensive expenditures	Expenditure by households or other agents on measures to combat the ill effects of pollution e.g. noise insulation, water filters.
Demand	Ability to purchase certain goods / services.

Denitrification Biologically induced conversion of nitrate to gaseous forms of nitrogen.

Determinism Philosophical doctrine or outcome in a mathematical model that all phenomena are inevitable consequences of prior events.

Direct use value Value derived from the direct use of an asset's resources and services.

Discounting Process of calculating the present value of a certain amount by applying a discount rate (interest) to a sum. Usually used to determine the equivalent present value of sums payable in the future (for instance, given a discount rate of 10% on a sum of £110 receivable in one year's time, the present value corresponds to £100).

Dissolved Matter Freely available matter.

(Operational) Molecular or ionic species in water, capable of passing a filter of defined size (typically < 0.001 mm).

Drainage basin Area occupied by a topographically defined drainage system; a region that collects surface runoff and supplies it to a specific body of water (e.g. streams, lakes). *Also, catchment area.*

Driver A force/action causing change.

Econometrics Branch of statistics testing economic hypotheses and estimating economic parameters making use of multiple regression techniques and other methodology.

Economic efficiency In an economy, allocation of resources leading to a net gain to society, estimated by subtracting the costs from the benefits.

Economic welfare The part of human well-being (at individual or group level) resulting from the consumption of goods and services.

Economics Science concerned with the efficient allocation of scarce resources within different contexts.

Ecosystems Functional units of the environment with characteristics of its interactions among biological, physical and chemical components.

Embedding In contingent valuation method (CVM), influence of other interests and concerns, apart those explicitly questioned, in the estimation of willingness to pay (WTP) values.

Energy Metrics Analytical approach which converts economic factors of production (land, labour and capital) into energy equivalent terms.

Estuary A semi-enclosed body of water with a fresh-water input and a free connection to open sea.

Eutrophication Increase of the amount of nutrients, especially nitrogen and phosphorus, in a marine or aquatic ecosystem.

Evaluation Process of determining the value of something.

Evaluation method	Ranking of alternatives by using rules (decision rules) that facilitate this process.
Existence value	Value of a resource that will never be used by present or future generations.
Extreme event	Phenomenon of unexpected or statistically unlikely magnitude (e.g. flood, drought, earthquake).
Feedback	(Mathematical) Non-linear effect of a product or action in a multistage process on a subsequent stage in the same process.
Flow	Movement of matter or energy.
Flux	Measure of the flow of some quantity per unit time; it may also be expressed per unit area or unit volume per unit time.
Function	Specific role of a person, system or thing in a determinate context.
Global Carbon Cycle	One of the main biogeochemical cycles on Earth; carbon is cycled around the globe among different reservoirs by physical, chemical and biological processes.
Global Environmental Change	Cumulative process of change, driven by human use of environmental space and resources.
Good	Commodity that is tangible, usually movable and in general not consumed at the same time as it is produced.
Gross National Product (GNP)	Total value of all final goods and services produced by a nation in a year.
Hedonic price	Implicit or shadow price of a good's characteristics.
Hedonic pricing method	Revealed preference method whereby individuals' valuations of environmental goods are determined by the market prices of the goods purchased (e.g. house prices) which, it is assumed, is done to enjoy the environmental good in consideration.
Human welfare	Social and economic well-being of a human group related to the things that contribute to it. *See also economic welfare and social welfare.*
Impact analysis	Assessment of the negative effects (damages) accruing to a site and to regional or local economy from an environmental impact.
Income elasticities of demand	Proportional change in quantity demand for a given change in income.
Indirect use values	Support and protection provided to the economic activity by the natural functions of the ecosystem or by regulatory 'environmental' services, such as flood alleviation.
Inner shelf	Portion of the continental shelf closest to the shore.
Integrated Coastal Zone	Management of coastal areas based on the integration of knowledge of

Management (ICZM)	coastal zone processes and different possible management options, to develop policies for a sustainable use of littoral areas.
Integrated management	Control of processes and actions by managing them as a whole, taking into account interactions throughout the system.
Integrated prognostic assessment capability	Capacity of predictive estimations based on the incorporation of scientific and socio-economic data on the system under scrutiny.
Interface	Boundary between two media, environments or areas.
Intergenerational equity	Taking into account the preferences of future generations in current actions and decision-making.
Land use/cover	Pattern of vegetation and settlement across the terrestrial environment.
Leach	To remove or be removed from a substance by a percolating liquid.
Ley grass	Grass cultivated temporarily on arable land.
Limnic organisms	Organisms living in freshwater ecosystems.
Load	Material moved or carried by a natural transporting agent; the total content of a material (often detrimental) within a system.
Longitudinal data	Information collected on constant experimental units over a period of time.
Macro scale	Study of a greater proportion of the subject or area under consideration, possibly as a whole. 'Macro' is a prefix meaning *large* or *great*.
Marginal cost	Additional cost of producing an extra unit of output.
Market analysis	Valuation method based on market prices.
Morphodynamics	Changes in form and structure.
Multi-criteria analysis	Appraisal of different projects by considering criteria that affect groups or individuals in different ways.
Multi-criteria evaluation	*See multi-criteria analysis.*
Nimbyism	The 'not in my backyard' syndrome i.e. local resistance to facilities such as waste disposal sites, incinerators because of fear of pollution and loss of local amenity.
Nitrogen fixation	Conversion of nitrogen gas (which most organisms cannot use) to organic nitrogen, nitrate or ammonium; these are all forms of nitrogen which can be readily used by organisms.
Nonuse value	Value that an individual may give to an asset even without personally using or intending to use it.
Nutrient budget	Quantitative estimate of the inputs and losses of nutrients to an

ecosystem.

Nutrient cycle	Movement of nutrients through a system.
Nutrient uptake	Conversion of nutrients (carbon, nitrogen and phosphorus) from inorganic forms to organic matter.
Nutrients	Raw materials needed for life. Major plant nutrients are carbon, nitrogen and phosphorus.
Observational study	Research in which experimental conditions are not controlled and conclusions are drawn based on historical relationships among variables. *Also called comparative study.*
Opportunity cost	Market value of things that would be forgone (or lost or given up) to obtain something else.
Option value	Value of a resource not being used at present, but with the option of being used in the future.
Organic matter	Molecules in organisms derived from degradation of organisms or excreted by organisms after being synthesised.
Organic production	Conversion of inorganic materials (especially carbon) to organic matter.
Outer shelf	Part of the continental shelf most remote from the shore.
Oxygen deficits	Reduced amount of oxygen, lower than expected or required.
Partial valuation	Assessment of two or more alternative use options of an environmental asset.
Particulate matter	Matter composed of particles that are not superficially bound together.
Phosphorus/Nitrogen limited water	Water in which the availability of the nutrients is approaching a lower threshold likely to cause changes in the growth of plants.
Phytoplankton	*See Plankton.*
Plankton	Organisms - bacteria (bacterioplankton), algae and cyanobacteria (phytoplankton) or animals (zooplankton) - living suspended in fresh-water or marine environments.
Point sources	Confined sources that can be identified as the origin of inputs into the surrounding medium (as opposed to non-point sources).
Policy instruments	Economic and social variables manipulated by the government to influence policy variables.
Pressures	Present and forecasted socio-economic activity levels.
Redfield ratio	Common ratio (by moles, not mass) of carbon, nitrogen and phosphorus in organic matter, especially in the ocean (C:N:P = 106:16:1 for plankton and plankton-derived organic matter).

Redox potential Measurement of the ability of an element to act as a reducing (by acquiring electrons) or oxidising (removal of electrons) agent.

Remote sensing Gathering and analysis of data from an object physically removed from the sensing equipment (e.g. satellite or aerial photography, subsurface detection instruments).

Residence times Total amount of material present in a system divided by the rate of delivery of that material.

Residuals Non-product outputs from human activities; these become wastes if not re-used or recycled.

Restoration Return to a previous state or condition, especially a condition of well-being, obtained by replacing those components lost.

Scoping Determining the area covered by an activity, topic.

SEK Swedish currency (Swedish Krone).

Sensitivity analysis Changing parameters of a decision problem or mathematical model in order to evaluate how this affects the outcome.

Services (In an economic sense) performed functions or tasks for which there is a demand and hence a market price.

Shadow Projects Projects compensating for environmental damage generated by existing/planned set of economic activities with the provision of an equal alternative elsewhere.

Sink Area, device or environmental 'compartment' that absorbs, retains or transforms a flow of matter or energy.

Social welfare Well-being of a society or community. In general, social welfare is seen as an aggregate of the welfare of different members of society.

Stakeholder Individual, group or institution potentially affected by a specific event, process or change.

Stakeholder analysis Management tool to ensure that policy planning is carried out efficiently and effectively. In economics, it concerns the quantification in monetary terms of costs and benefits accruing from a project / proposal / decision to the different groups in society that have a related interest in it. This is based on the notion that policies, programmes or projects have differential effects on a range of actors, who gain or lose according to their interests. From a sociological viewpoint, it aims to help organisations work through different courses of action and to identify the actors that are likely to favour and press for particular kinds of change and to consider the opinions and interests of these groups (techniques such as Focus Group analysis are employed).

Strategic mitigation Re-creation of habitat in a location different to where originally situated in order to compensate for its loss elsewhere.

Supply (In economics) goods or services available for purchase.

Sustainability **Strong:** maintenance of the amount of capital available to a population, keeping the quantity of natural capital (or 'critical components of ecosystems') constant over time. An axiom of this condition of sustainable development is that natural capital can not be replaced by other forms of capital; the four forms of capital (natural, human, human-made and social-moral) are complementary to one another.

Weak: constant maintenance of the overall amount of capital available to a population, allowing for exchanges (unlimited substitution possibilities) among different forms of capital, achievable through technological progress.

Sustainable development "Development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (World Commission on Environment and Development (1987) *Our Common Future*, Oxford University Press, Oxford, p. 43). *See also weak and strong sustainability*.

Sustainable management Sustainable utilisation of the multiple goods and services provision generated by coastal resources.

Total economic value Total of use and non-use values.

Total valuation Assessment of the total net benefits to society (total economic contributions) of the ecosystem under consideration.

Trace gas Gas occurring in very small quantities ('trace quantities').

Transition economy Country whose economic activity is progressing towards assuming those characteristics of developed or industrialised nations.

Travel cost Estimation of the value of visiting an ecosystem derived from the cost of travel to that location, including the recognition of the opportunity costs of travel time [revealed preference method].

Turbidity Condition (usually of a liquid) resembling cloudiness created by the suspension of particles.

Typology See the definition given in Appendix 1. Also: A system of classification or grouping of entities based on similarities among combinations of characteristics.

Valuation Quantification of the values of a good or service usually calculated by examining the demand for it showing how much people would use at varying prices.

Value The worth of a good or service measured in terms of willingness to pay minus the costs to supply it. *Refer also to: direct use values, indirect use value, nonuse value, option value, total economic value and different methods of estimating these: contingent valuation, market analysis, hedonic pricing etc.).*

Wetlands "Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six metres" (art. 1.1) and "may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands" (art. 2.1 of the Ramsar Convention (1996) *Strategic Plan 1997-2002*, Ramsar Convention Bureau, Gland, Switzerland).

Willingness to pay (WTP) Amount that an individual is prepared to pay to purchase a good or the use of a service, independently of the existence of a prevailing market price or if the good / service is free of charge.

Zooplankton *See Plankton.*