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ECONOMIC-ECOLOGICAL MODELING

L.C. BRAAT
W.F.J. VAN LIEROP
editors

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ECONOMIC-ECOLOGICAL MODELING

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ÅKE E. ANDERSSON
WALTER ISARD
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Economic-Ecological Modeling

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INTRODUCTION TO THE SERIES

Regional Science and Urban Economics are two interrelated fields of research that have developed very rapidly in the last three decades. The main theoretical foundation of these fields comes from economics but in recent years the interdisciplinary character has become more pronounced. The editors desire to have the interdisciplinary character of regional sciences as well as the development of spatial aspects of theoretical economics fully reflected in this book series. Material presented in this book series will fall in three different groups:

- interdisciplinary textbooks at the advanced level,
- monographs reflecting theoretical or applied work in spatial analysis,
- proceedings reflecting advancement of the frontiers of regional science and urban economics.

In order to ensure homogeneity in this interdisciplinary field, books published in this series will:

- be theoretically oriented, i.e. analyse problems with a large degree of generality,
- employ formal methods from mathematics, econometrics, operations research and related fields, and
- focus on immediate or potential uses for regional and urban forecasting, planning and policy.

Åke E. Andersson
Walter Isard
Peter Nijkamp

THE INTERNATIONAL INSTITUTE FOR APPLIED SYSTEMS ANALYSIS

is a nongovernmental research institution, bringing together scientists from around the world to work on problems of common concern. Situated in Laxenburg, Austria, IIASA was founded in October 1972 by the academies of science and equivalent organizations of twelve countries. Its founders gave IIASA a unique position outside national, disciplinary, and institutional boundaries so that it might take the broadest possible view in pursuing its objectives:

To promote international cooperation in solving problems arising from social, economic, technological, and environmental change

To create a network of institutions in the national member organization countries and elsewhere for joint scientific research

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To inform policy advisors and decision makers about the potential application of the Institute's work to such problems

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Preface

This book is about mathematical models for environmental and resource policy and management. The authors present an overview of the theory, methods, techniques, and experience relevant to the analysis of problems of the interface of society and its natural environment. We also evaluate the scientific adequacy and policy effectiveness of a wide variety of applied economic–ecological models. Furthermore, we indicate the reasons for success and failure of these model applications and summarize options for their improvement.

The contributors' aims in writing this book are:

(1) To provide a systematic overview of models for environmental and resource analyses.
(2) To formulate general guidelines for the evaluation, selection, and optimal design of this type of model.

The book results from a study in which the Institute for Environmental Studies (IvM), Free University, Amsterdam, and the International Institute for Applied Systems Analysis (IIASA), Laxenburg, Austria, cooperated to examine the "relevance of economic–ecological models for environmental and resource policy."

This IvM–IIASA study included a questionnaire survey of over 100 models. Analyses concentrated on model structure and properties, policy problems, and model applications. After evaluation of the initial results of the survey, a number of experts were invited to evaluate the adequateness and effectiveness of technically different types of applied models in their respective fields of expertise (agriculture, water quality, etc.). A workshop was held at IIASA in December 1983 at which these model evaluation studies and major survey results were presented and discussed among an international group of modelers, policy analysts, and policy advisers. The results of the analyses conducted within the project, the invited papers, and the essence of the discussions at the workshop are brought together in this book.

The book consists of three parts:

(1) In Part I, Theory and Methods, we introduce the environmental and resource problems and policy issues for which models are developed. The reader is subsequently taken, via economic and ecological approaches in environmental and resource analysis, toward integrated economic ecological models. Much of the information in this Part is meant as an introduction for economists and ecologists to each others discipline. Consequently, for scientists already trained in multidisciplinary modeling, Part I will offer relatively few new viewpoints.

(2) In Part II, Practice of Environmental and Resource Modeling, the authors present a selection of modeling approaches that were developed and applied in nine sectors of environmental and resource management, namely fisheries, forestry, agriculture, water resources, water quality, outdoor recreation, multiple resource use, regional systems policy, and national and global systems policy. All the corresponding chapters evaluate the models comparatively for their effectiveness in management.

(3) In Part III, Policy and Modeling, the contributors discuss the process of environmental and resource modeling and evaluate it from the perspectives of scientists and policymakers. Possible conflicts and options for cooperation between modelers and policy analysts are illustrated and analyzed. The book is completed with an evaluation and some general conclusions.

Obviously, a book such as this is the product of many people. We are indebted to all the participants in the economic–ecological modeling survey project who together delivered the data base that made the IvM–IIASA study possible. We also thank all our colleagues at IvM and IIASA who have contributed to the project and the book. Last, but certainly not least, we thank the secretarial and editorial staff, both at IvM and at IIASA for all their work in the fine-tuning of this book.

Leon C. Braat
Wal F.J. van Lierop

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PART I

Theory and Methods

Introduction to Part I

L.C. Braat
W.F.J. van Lierop

The growing awareness of environmental problems has stimulated much research in economics and ecology over the last two decades. At the same time, both in economics and in ecology mathematical modeling approaches have increasingly become more important. The pioneer work of Lotka (1920) and Volterra (1931) in population ecology, of Lindeman (1942) at the ecosystem level, and of Tinbergen (1956) in economics, has been followed by extensive efforts to obtain more insight into the complexities of the real world by means of statistical, econometric, simulation, and analytical modeling techniques. In the last 15 years academic researchers as well as policy analysts became increasingly aware of the limitations of monodisciplinary modeling. A series of attempts was undertaken which aimed at the improvement of the existing models.

In the 1960s studies on *environmental* and *resource economics* started to be published (Barnett and Morse, 1963; Ridker, 1967; Kneese *et al.*, 1969). In the 1970s both these new subdisciplines of economics proliferated (Maler, 1974; Krutilla and Fisher, 1975; Pearce, 1976; Nijkamp, 1977; Kneese, 1977).

Special attention to the relationship between economic growth and environmental constraints and impacts was given by Barkley and Seckler (1972), and Hueting (1974). The economics of pollution effects, environmental damage, improvement, and control was addressed by Victor (1972), Maler and Wyzga (1976), Smith (1976), and Freeman (1979).

Resource economics in the 1970s has concentrated on minerals and fuel resources, the so-called nonrenewables (Herfindahl and Kneese, 1974; Pearce, 1975; Pearce and Walter, 1977). Renewable resource economics has received most of its attention under traditional names such as agricultural, forestry, and fisheries economics (Ciriacy-Wantrup, 1968; Fisher, 1977; Smith, 1978). Several books were published in these areas calling for renewed attention to the problems of depleting stocks and especially to the problems in developing countries (Dasman, 1973; Hufschmidt and Hyman, 1982). In the latter part of the 1970s and early 1980s resource and environmental economics were dealt with in a single context by Smith (1976), Ayres (1978), Fisher (1980), and Hufschmidt and Hyman (1982).

On the other side, the development of ecological theory and methods for the analysis of impacts of and constraints for socioeconomic activities has followed a similar dichotomy. *Environmental biology* has become an established field dealing mainly with pollution aspects, both eutrophication and ecotoxicology (Jørgensen, 1979; Van Steenkiste, 1978; Rinaldi, 1982) and *resource ecology* has focused on renewable resources such as harvestable plant and animal populations in forestry and fisheries respectively (Watt, 1968; May, 1976; Holling, 1978).

Many of these studies focused on developing existing monodisciplinary analytical methods into models more suited for the multidisciplinary problems in environmental and resource management.

In Chapter 1 an overview of these multidisciplinary problems, and a brief introduction to environmental and resource policy, management, and economic–ecological modeling is given.

Nijkamp, in Chapter 2, and Jeffers, in Chapter 3, present a review and critical evaluation of monodisciplinary and "extended" monodisciplinary modeling approaches. These chapters function as introductions to the background, theory, and methods of economic and ecological modeling. They are principally written for ecologists and economists respectively. Both authors have published acknowledged standard reference books on modeling in their respective disciplines. Their vast knowledge has been expertly condensed in these two chapters.

In Chapter 4, the focus is on integrated economic–ecological models. First, methods to integrate existing monodisciplinary models are discussed. Subsequently, the alternative, the integration of economic and ecological theory in multidisciplinary model building is considered. The technical problems of integrating economic and ecological theory and methods are examined and evaluated in the light of theoretical requirements. The practice of economic–ecological modeling is discussed as well as a number of solutions to the various technical problems of integrated modeling.

The training and experience of the reader should determine how to read Part I. Chapter 1 provides a basis for understanding the various types of models, problems, and policies, discussed further on. Chapter 2 has been written for readers with a limited training in economics, and environmental economic modeling. In the same fashion, Chapter 3 is included for readers without experience and training in ecology and ecological modeling. In Chapter 4, as well as in the other chapters of the book, it is assumed that the reader has acquired, here or elsewhere, some proficiency in environmental science, economics, ecology, and modeling.

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CHAPTER 1

Environment, Policy, and Modeling

L.C. Braat
W.F.J. van Lierop

1.1. Introduction

Environmental and resource problems result from the use of ecological systems for socioeconomic production and consumption activities. These problems can be seen as discrepancies between demand for goods and services and their supply by ecological systems. This is primarily an economic viewpoint. It is argued that explicit economic objectives are not met by the resource flows and environmental services. Reasoning from this point of view, *economic analyses* seem to be most appropriate to contribute to the solution of these problems.

On the other hand, however, the degraded state of the natural environment can be regarded as the primary problem, i.e., the fact that nature conservation objectives are not being achieved. From this point of view, *ecological analyses* seem to be the ones most urgently needed to solve the problems.

These two viewpoints illustrate that environmental and resource problems generally have at least an economic and an ecological side. Economic activities are characterized by social and psychological factors, by the law, institutions, politics, and technology. This implies that environmental and resource problems also have these characteristics. Ecological systems are of course very much governed by the laws of physics, chemistry, and geology. Therefore, environmental and resource problems have also physicochemical and geological aspects.

The exact definition of an environmental or resource problem is determined to a great extent by the *temporal* and *spatial* perspective of those who recognize the problem and have to deal with it. For example, the development of an oil well is obviously a traditional economic problem if considered in a short-term and local perspective. It may, however, gain some ecological aspects, e.g., polluted oceans and destroyed soils, if the perspective shifts to longer time and larger spatial scales. Conversely, the restoration of dying fish populations and a polluted lake may appear to be a matter of enforcing strict ecologically based nature conservation rules. But, if the perspective changes to long-term and regional scale, economic aspects, such as financial support needed for effective conservation management, are coming into focus.

Apparently, looking at a particular problem, one may rightly focus on either the economic, or the ecological aspects in case the problem and its side effects are restricted to a local scale and short period of time. If a problem is widespread but has a short life span, a ***monodisciplinary*** approach may still be sufficient, since factors which are exogenous to such problems can be assumed to be constant for that period while the time is too short for feedback and synergistic effects to develop. If, however, a long-term problem is at hand, a ***multidisciplinary*** approach would seem to be much more appropriate. As time proceeds, exogenous factors generally do not remain constant, and feedbacks and synergistic processes will develop.

The conclusion from this short discourse may be that, although resource and environmental quality problems are often considered to be short term and local, and therefore purely economic or ecological problems, they are economic–ecological problems in the long run.

If this deductive reasoning is extended to a particular type of instruments used to analyse and solve scientific and policy problems, i.e., conceptual and mathematical models, it would seem that short-term analyses of environmental and resource problems may be conducted with monodisciplinary models, while long-term analyses require models which include both the economic actors and activities and the ecological components and processes. Models which comply with this requirement would justly be called economic–ecological models.

A score of questions can be raised concerning these hypotheses about the nature of the models which are most appropriate for certain types of problems. This book aims to provide answers to the following questions:

(1) Regarding problems and policy:

- (a) What are the nature and causes of environmental and resource problems?
- (b) What is involved in environmental and resource policymaking and management?

(2) Regarding the state of problem solving instruments, which modeling techniques are used in environmental and resource policy making and management?
(3) Regarding the quality of these instruments, what is the adequacy and effectiveness of various types of environmental and resource models?
(4) Regarding the model design, how does one build adequate and effective models for environmental and resource analysis?

The remaining part of this chapter examines the nature, causes, and classes of the problems of environmental quality and resource availability (Section 1.2). In Section 1.3 the actors, objectives, and classes of the issues in environmental and resource policy and management are discussed. In Section 1.4, a review of general research methodology precedes the discussion of the role of economics, ecology, and modeling for environmental and resource problem analysis.

1.2. Environment

1.2.1. Introduction

The generic term "environment" is used to describe the physical–biological parts of the surroundings of man which are not designed and constructed by human activities. An untouched tundra or rain forest is obviously natural and not man-made. Concrete, plastic, and steel cities, such as New York, are almost completely man-made. In between these extremes there are systems like parks, agricultural land, and ponds, which combine natural system components and processes with human designs and labor.

1.2.2. Environmental and resource problems

Two major areas of problems are distinguished. First, there are the problems of supply of goods from ecological systems to socioeconomic production and consumption processes, generally designated as resource problems. They include ecological problems due to resource extraction activities and economic problems of cost-effective resource development and exploitation. Second, there is the disposal of waste and by-products of socioeconomic activities, and the energy in the activities themselves which lead to environmental quality problems.

In many parts of the world *the present state* of the environment is considered problematic, either because resource stocks are less productive or because of faltering services of the environment. The resource problems may regard renewable resources like top soils, fish populations, and timber; or nonrenewables like concentrated minerals and fuels. The pollution problems are partly problems of quantity overloading, like phosphate loads in lakes and acid rain, and partly problems of a qualitative nature, i.e., the type of substance, like chlorinated hydrocarbons (PCBs). In addition, there are the problems related to lack of space and time for natural system components and processes to develop mechanisms to cope with alien inputs, and to recover from excessive exploitation and to mineralize, and thus recycle decomposable waste (Ehrlich *et al.*, 1977: Conservation Foundation, 1984; Brown, 1984).

1.2.3. Causes of environmental and resource problems

By now, few would deny that a major cause of environmental and resource problems is the total of human economic production and consumption activities. Regularly, however, it is still argued that the impact of man on the natural environment is nothing to worry about, since man has influenced his natural environment throughout history. Over a long period of time the influence, however, appears to have been compatible with the carrying capacity of the natural systems early man lived in. Most people agree that this is no longer the case.

The availability of resources and the quality of the environment only became a serious problem when the exploitation of the resources and the disposal of waste from productive and consumptive activities began to take place at rates and to an extent, which were no longer compatible with the capacity of the natural systems to produce raw resources and absorb and process waste. In addition, lately some substances produced by man are of such a nature that the natural systems have not yet developed the biomechanisms to cope with them, nor to fight or resist their toxicity (e.g., PCB, DDT).

These problems were first thought of as local, hence relative scarcity problems. When resources turned out to be depleted over large areas, and renewable resources proved to require longer recovery times than before, or became nonrenewable by being forced below crucial thresholds, they were finally perceived to transcend local allocation and distribution questions.

Of course, environments change not only in proportion to man's influence. Natural systems develop and change, e.g., in growth, successional, and evolutionary processes, thereby changing relative abundance of species numbers and the abiotic storages. The incidental state of the resource base and the environmental quality are therefore also very much dependent on the dynamics of the natural ecological systems.

In summary, man has considerably contributed to the present problematic state of the environment and large groups of people do certainly have something to worry about (*see*, e.g. Ahmad and Muller, 1982; Hufschmidt and Hyman, 1982; OECD, 1981).

1.2.4. The relationship between man and environment

A distinction can be made between "influence on the environment" and "environmental impacts". The first is an activity originated by human beings, individually or as groups in society, and directed towards the extraction of resources from or use of services performed by the natural environment. The impacts are the resulting changes in components, patterns, and processes of the natural systems (*see* IvM, 1980).

Two major types of environmental influence are distinguished: negative and positive. Negative influences are those (human) activities, and the related energy and matter flows, that lead to deregulation of natural systems. Exploitation of renewable and nonrenewable resources, disturbance of breeding bird communities, and polluting activities are familiar examples.

Positive environmental influence is that type of human activity directed at control and mitigation of negative influences and recovery of affected natural systems. It includes policies aiming at nature conservation and at an environment that is not dangerous to human health, and at renewable resource management and pollution control.

These concepts of the relationships between society and its natural environment constitute, of course, still only a limited means of representing the enormously complex reality.

For the discussion in this book of models for environmental and resource management an even simpler approach is chosen: a conceptual model which consists of two black boxes, related by two composite flows (*see Figure 1.1*).

The left-side arrow in *Figure 1.1* represents the flows of energy, matter, and information from an ecological system to a socioeconomic system. These flows are typically the resources used as an input to the economic production and consumption processes. The right-side arrow also indicates flows of matter, energy, and information. Here, they indicate human activity in outdoor recreation and environmental management as well as the waste products of the social and economic processes. In the next section this simple model will be used to develop a classification of environmental and resource policy issues.

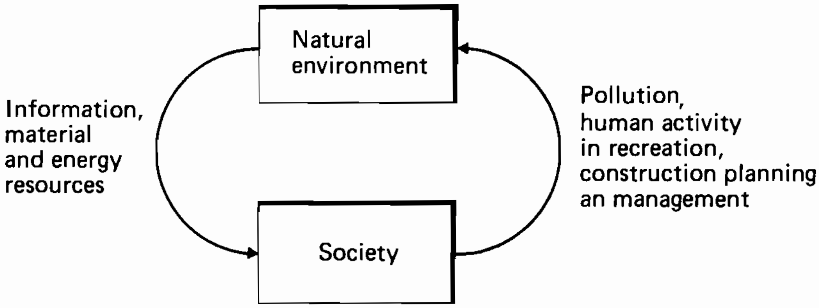

Figure 1.1. Relationships between society and the natural environment.

1.3. Policy

1.3.1. Introduction

Academic scientists, policy analysts, and decision makers have special roles in identifying and dealing with environmental and resource problems. "The primary role of a decision maker is to make right decisions on the basis of available information, and within the allowable time and resource constraints" (Biswas, 1975). The primary task of scientists and policy analysts is to supply and selectively transform the relevant information, respectively. The essential information in each environmental and resource problem situation are the causes and impacts and the set of feasible policy options and their consequences for the socioeconomic and ecological systems involved. This also implies information about the systems behavior in general. Academic scientists are trained to collect, analyze, and synthesize relevant information. They anticipate problem situations by solving hypothetical problems and by attempting to understand the system and its behavior. Policy analysts generally do the same, except that they do not deal with hypothetical problems and fundamental understanding of systems but with problems that originate in an actual or expected discrepancy between supply of goods and services.

1.3.2. A characterization of policy issues

Environmental and resource policy issues can be characterized by two major aspects: (I) the objectives with which the problem is addressed; and (II) the location of the problem.

(I) Objectives

In environmental and resource policymaking the following three main types of policy objectives can be distinguished.

(1) *Nature conservation objectives*. The first group can be characterized as: "minimum exploitation and damage of natural systems". The objective may take the extreme form of complete preservation, no access, and no use (nature reserves, sanctuaries). In general, these objectives are voiced explicitly for limited areas only, sometimes with the implicit purpose of saving resources for later use. Another aspect is the protection of natural systems from consumptive use for the nonconsumptive forms of use, such as recreation, aesthetics, and scientific research.

(2) *Economic objectives*. The second group of objectives share the characteristic of "maximum production of goods and services at minimum cost". Here, the extreme case might be the total destruction of the structure of the system (clear cutting in forestry, anchoveta fisheries). The satisfaction of present needs is predominant. The needs may be very basic, such as food and shelter, or not so, such as individual luxury and wealth.

(3) *Mixed objectives*. The third group of objectives is not as well known or common in politics. They imply 'maximum sustainable use of resources and environmental services'. The crucial concept is sustainability. It means that the various forms of use are compatible with the productive and carrying capacity of the natural systems involved. It implies that this compatibility extends over an unlimited period of time. These objectives are "mixed" in that economic and nature conservation objectives are considered at the same time.

The problems or questions facing environmental and resource policymakers and managers are addressed with either one of these objectives. The problems are called policy issues as soon as a problem-solving process is not purely an academic exercise, but has been initiated with the purpose of developing a policy or management strategy.

(II) Location

To explain this second aspect we refer to the conceptual model shown in *Figure 1.1*. An environmental or resource policy problem can be located at an "output" or "input" side of either system, include both the input to and output from one system, extend from the output side of one system to the input side of the other system or, finally, involve the whole loop through the two compartments.

1.3.3. A classification of policy issues

For the analysis and evaluation of the adequacy and effectiveness of models for environmental and resource management, a classification of policy issues has been developed, based on the two aspects of policy issues discussed above.

Three groups of classes are distinguished, each group containing three classes. This is done according to the distinction in policy objectives as mentioned above. The classes are based on the differences in location of the problem.

The following classes of policy issues have thus been identified:

(1) Ecological policy issues (*see Figure 1.2*):

- (a) Class 1 ecological impacts of resource use.
- (b) Class 3 impacts of pollution and disturbance.
- (c) Class 5 ecosystem conservation management.

(2) Economic policy issues (*see Figure 1.3*):

- (a) Class 2 economic impacts of resource use.
- (b) Class 4 economic impacts of pollution and disturbance.
- (c) Class 6 economic optimization management.

(3) Mixed economic–ecological issues (*see Figure 1.4*):

- (a) Class 7 sustainable use of resources.
- (b) Class 8 sustainable use of environmental services.
- (c) Class 9 total system management.

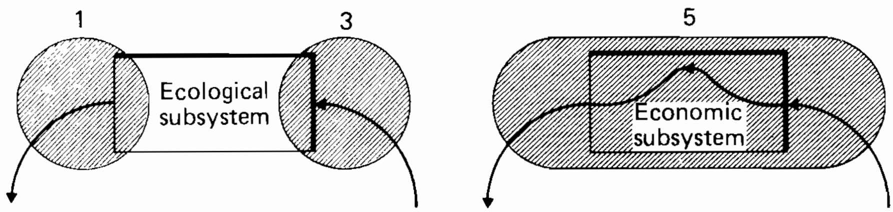

Figure 1.2. Ecological policy issues.

The concern in the group of ecological policy issues is determined by the nature conservation objectives. The policy issues are the impacts of human use of natural systems and how to manage them for maximum conservation. Economic objectives underly the second group of issues. The concern is at the economic side, often the costs, but also allocation. The general problem is how to maximize welfare at minimum cost given various constraints. In the last group, the objectives are combined. This leads to policy issues in which the concern is to

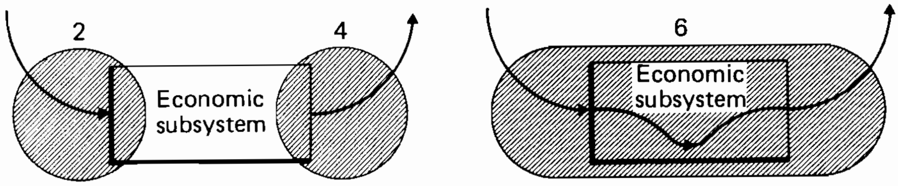

Figure 1.3. Economic policy issues.

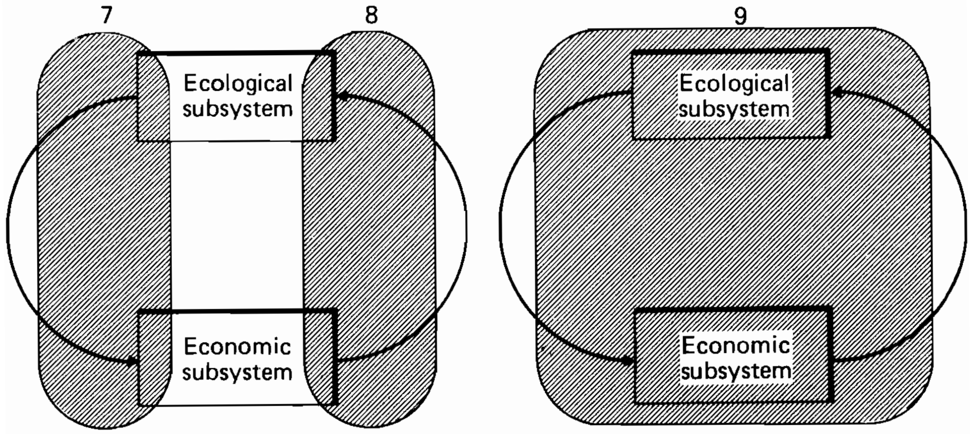

Figure 1.4. Mixed economic–ecological policy issues.

maximize the utilization of ecological system resources and services within accepted constraints of productive and carrying capacity of those ecosystems. In *Figures 1.2–1.4* the shaded areas indicate the "location" of the policy issues.

To maximize the effectiveness of policies in resolving environmental and resource problems, the character of the policy issue, as defined above, should obviously strongly influence the design of the model which is expected to assist in the policymaking process. Therefore, this classification of policy issues was developed.

1.4. Modeling

1.4.1. Phases in the policymaking process

As indicated in Section 1.3, the *problem solving process* involves a variety of activities: description of the problematic state, analysis of the causes, prediction of the consequences, prescription of an optimal solution or evaluation of alternative solutions. In analogy, a ***policymaking process*** can be divided into a number of phases. Some of those are recognizable as forms of problem solving research, others belong to the area of decision making and implementation. Since different phases may call for different instruments to complete them, identifying and defining these phases is relevant to the selection and design of the appropriate type of model for environmental and resource policy and management.

The following phases in the process of policymaking are distinguished (*see also* van Lierop, 1986):

(1) Signals. Initially, signals which emerge from society about the presence and recognition of a problem (e.g., frictions, options, needs which are not fulfilled) lead to demand for a solution.

(2) Objective research:

- (a) Description of the actual state of history of the system, for instance by monitoring or mapping.
- (b) Explanation by analysis of the factors that influence the state, e.g., by causal experimentation and statistical analysis.
- (c) Prediction by extrapolation of the present state or historical trends using the causal mechanism determined in (b).
- (d) Validation and technical evaluation of the analysis and predictions, for example through field checks and sensitivity analysis.

(3) Normative research:

- (a) Scientific diagnosis: the confrontation of facts and figures from (2a) to (2c) with criteria, standards, and a priori objectives, e.g., to assess the need for new policies.
- (b) Plan design: using the same elements as in (3a), alternative plans are outlined.
- (c) Plan evaluation on the basis of comparison between plan impact predictions and standards.
- (d) Selection of optimal plan: using criteria and weights for them, scores are developed for each plan.

(4) Implementation. The policy (plan) chosen is put to work. This often means additional research and detailed planning. Rules are formulated, as an interpretation of the more general policy, at the appropriate temporal and spatial scale. Funds are allocated to execute the actions.

(5) Ex post evaluation. Although large scale policies are rarely evaluated completely for their effectiveness in dealing with the

problem they were developed for, very often changes in the existing rules and regulations are made to solve problems neglected in the initial analysis.

Many phases in the policymaking process are carried out with the use of models. In some cases only conceptual schemes are applied, in other cases sophisticated mathematical structures are in use.

1.4.2. Model use and model types

As stated before, models are primarily instruments for research and understanding. A model is, by definition, not a 1:1 copy, but a simplified version of a part of reality. It is this simplification that makes a model useful because it offers a comprehensible version of a problem situation. The simplification is, however, at the same time its greatest drawback. To produce a comprehensible, operational representation of a part of reality, which grasps the essential elements and mechanisms of that real world system, is a hard task indeed. To make other people accept the chosen simplifications and aggregations and to make them interpret the predictions and prescriptions produced with a model in the way intended has proved almost impossible. This is why it is considered important to explicate the steps and concepts in the selection and design of the appropriate model for a particular problem.

Over the years whole families of quantitative modeling techniques have been developed for all kinds of purposes within the processes of scientific discovery and problem solving. The following types of models can be distinguished, when looking at their intended use (*see* Bennett and Chorley, 1978; Ören, 1979; Van Steenkiste, 1979):

(1) Descriptive or exploratory models, intended for a preliminary analysis of the relevant problem or to give an initial overview which could provide a basis for more careful research of its structure and relationships.

(2) Explanatory models, developed on the basis of observation of both input and output and aimed at clarifying the working of a system.

(3) Predictive models, based on known input and system structure, they are to be used to extrapolate developments or forecast changes.

(4) Prescriptive, control or management models, i.e., specific types of predictive models, designed to optimize objective functions and to define the conditions under which it is possible to achieve the policy objectives. Here the output is determined a priori, the input is to be assessed.

(5) Evaluative models, which provide a structure and algorithm to present the consequences (impacts) of alternative choices according to selected sets of criteria and weights.

In policy design and management, models are generally used to generate alternative solutions to the problem and to assess the impacts of these solutions. Obviously, in those cases, descriptive and explanatory models are merely considered as necessary stages in model development. After that stage, either simulation (predictive) or optimization models search for feasible solutions. *Figure 1.5* gives an illustra-

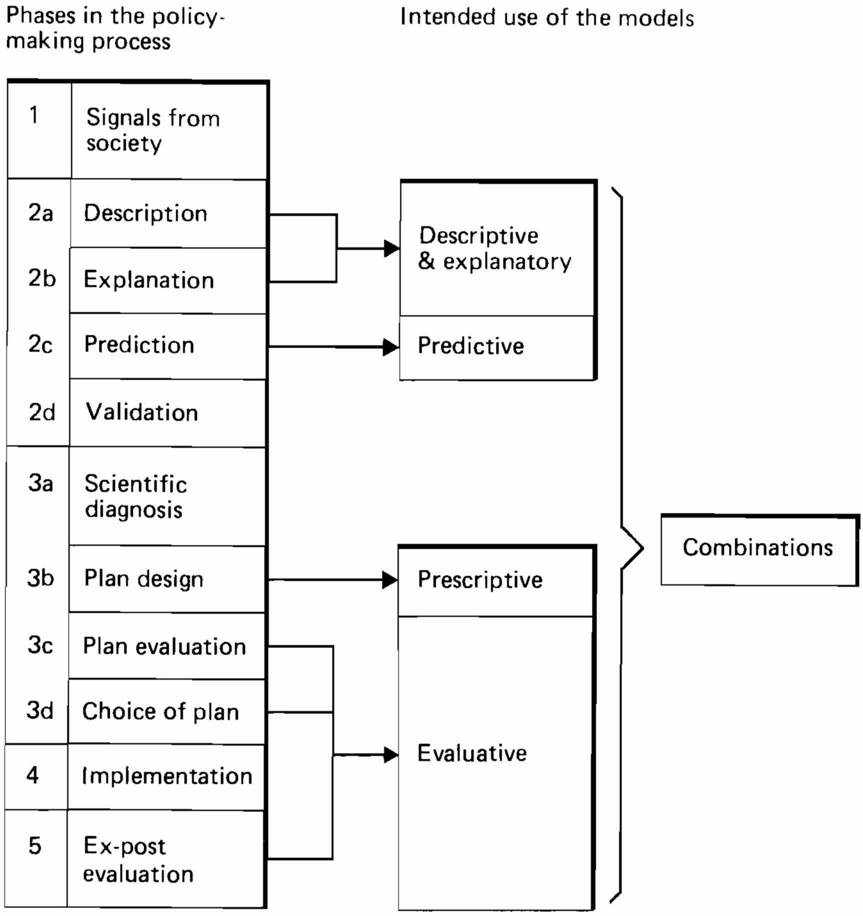

Figure 1.5. Selection of most appropriate modeling technique for a policy issue.

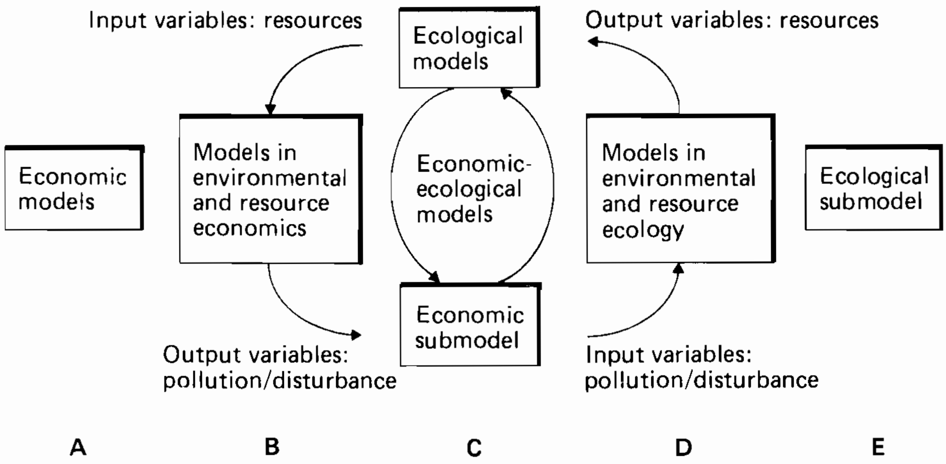

Figure 1.6. Monodisciplinary and multidisciplinary model types.

tion of the modeling approaches and techniques that are considered most appropriate for the respective phases in the problem solving process.

1.4.3. Economic–ecological models

In the discussion of models for environmental and resource analysis and management, we have mentioned a number of different types of models. *Figure 1.6* summarizes the disciplinary classification of environmental and resource models. At the extreme left and right of the figure the purely economic and ecological, monodisciplinary models are indicated. Their use in environmental and resource analysis is quite limited and therefore they will not be discussed further in this book.

The classes of environmental and resource economic models are indicated as economic model structures with pollution output and resource input flows respectively. In Chapter 2, Nijkamp discusses a selection of models from these classes. Ecological models with resource output and pollutant input flows constitute the classes of resource ecology and environmental biology models respectively. Jeffers in Chapter 3 describes the various types of models in these classes.

The class of economic–ecological models is depicted as two separate submodels linked by resource and pollution data flows. This class will further be described in Chapter 4.

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CHAPTER 2

Economic Modeling: Shortcomings and Perspectives

P. Nijkamp

2.1. Introduction

Human activities have never had a neutral impact on the environment. Since the early history of mankind, there has been a permanent transformation of man's physical surroundings. This environmental transformation process has rarely received much interest from economists.

Economics is, generally speaking, concerned with choice problems emerging from alternative uses of scarce resources. In earlier time periods, in which space and nature were not regarded as a scarce commodity, economics did not show much interest in environmental and resource problems.

Classical economists (e.g., Smith and Ricardo) concentrated their attention on the functioning of the market mechanism in relation to production, consumption, and prices of private commodities. Public interference with the economy had to be as low as possible in order not to destroy the market system. In their view, public control of scarce environmental goods could hardly be a subject of study for economic theory.

Political economists (e.g., Marx) questioned the passive role of economics in regard to environmental issues. They clearly pointed out that negative impacts of the production technology (e.g., air and water pollution) affected especially the industrial proletariat, as they were forced to live in the vicinity of dirty industrial areas. Also in the post-war period, political economists were among the first who had an open eye for the environmental disruption due to industrial growth and its impacts on the living and housing conditions of the working class (Kapp, 1950). It is also increasingly being realized that environmental decay is not a specific consequence of a capitalist mode of production, though the competition inherent in such a system will tend to neglect those phenomena that cannot be provided with a price tag (Baumol and Oates, 1975).

Neoclassical economists have, to a certain extent, paid attention to environmental problems, especially through the notion of compensation for environmental decay via pollution taxes (Pigou, 1920; Coase, 1960). From the 1960s onward, an avalanche of economic literature has been published on environmental issues. The debate on environmental effects of economic activities was based on neoclassical concepts (market equilibrium, prices, production function with substitution effects, marginality principles, etc.). From the late 1960s onward various formal models have been developed in which economics and environmental effects were treated in one framework (*see* for instance Mäler, 1974; Seneca and Taussig, 1974; Nijkamp, 1978). The major problem remained of course the intangible nature of environmental commodities, so that it was very troublesome to include appropriately environmental effects in a neoclassical framework. Even the well-known concept of monetary compensation for loss of environmental quality could hardly be operationalized owing to lack of price tags on environmental commodities. Furthermore, the economic impacts of antipollution technology on the welfare positions of polluters and pollutees are hard to assess (*see*, for some of such attempts, Getz and Hwang, 1978; Kuz, 1978). A central concept in neoclassical economics is that of externalities (*see also* Hafkamp, 1984). Externalities are nonmarket effects caused by the economic activities of consumers, entrepreneurs, or governments that affect the welfare position of others who are not compensated for the change in their welfare.

From the mid-1970s, a wide variety of economic approaches to resource environmental issues has come about, not only in regard to pollution, but also in regard to the use of natural resources and man-made environment. The sudden interest in these issues was mainly due to two factors:

(1) A *supply* factor: the rapid economic growth in most countries in the post-World War II period has led to a rapid decline in environmental quality (in both a quantitative and a qualitative sense) and to a depletion of natural resources. Hence, the scarcity paradigm became relevant for environmental commodities.

(2) A *demand* factor: the rise in income has led to a shift in priorities regarding economic goods (in a Maslow sense): instead of basic

goods (shelter, food, clothing) a higher priority was attached to environmental goods, so that their availability (quantitatively and qualitatively) had a larger impact on the welfare of people.

It should be noted that, in general, environmental economics is concerned with two related problems:

(1) The analysis of environmental repercussions of economic activities and of the choice problems emerging from this relationship.
(2) The analysis of economic effects of environmental change and of the choice problems emerging from this relationship.

This implies that the scarcity paradigm has now evolved in two directions:

(1) Environmental attributes (for example, beauty of landscape, recreational potential of rivers) have become scarce as a consequence of our current economic–technological system.
(2) Traditional scarcity (e.g., lack of financial resources) has become more apparent as a consequence of societal decisions to cope with environmental decay (for instance, by constructing sewage plants).

These two scarcity issues have led to intricate problems in economic policy. Is, for instance, a continuation of economic growth necessary or harmful for a better protection of the environment? Is it possible to find a common "measuring rod" (e.g., money) in order to arrive at a meaningful tradeoff between economic and environmental aspects? Are there possibilities to specify and assess the linkages between production and consumption on the one hand and processes in complex ecosystems on the other hand?

In the past few years, many attempts have been undertaken to use the price mechanism as a unifying frame of reference for judging the social value of material commodities, services, and environmental goods. However, the success of providing a price tag to environmental goods has been very limited due to the above-mentioned externalities. Only in as far as the price mechanism could be used to assess the opportunity costs of alternative decisions regarding environmental goods, these attempts have to a certain extent been useful.

On the other hand, much more success has been achieved in designing operational models that describe the intricate relationships between the economy and the environment.

2.2. A Review of Environmental–Economic Modeling Approaches.

In the post-World War II period, economic research has increasingly become quantitatively oriented. Econometrics, statistics, mathematical economics, and operations research have paved the way for a rigorous modeling of components and interactions in an economic system. Starting from simplified macromodels, economists have been able to design complex, multicomponent, and multiactor policy models which are nowadays being used in many countries as decision aids for policymakers. The use of economic models can be justified on various grounds:

(1) Models provide a concise and surveyable formulation of regularities in economic behavior.
(2) Models presuppose a consistent definition and use of concepts and variables.
(3) Models are able to represent complex and intertwined phenomena in a stylized and simplified way.
(4) Models provide means to check the consistency of theories or inferences.
(5) Models allow an empirical application and test of real-world patterns and processes.

On the other hand, the popularity of modeling activities in economics has not always been justified from the actual achievements of models. Especially in the past years, economic models have been the subject of much criticism. These critical remarks concerned *inter alia* the following points:

(1) Economic models are often based on past data which are irrelevant in an era of structural economic change (for instance, the phenomenon of asymmetric consumer behavior in a period of an economic upswing and an economic downswing).
(2) Economic life is determined by multiple actors and interest groups which can hardly be included as relevant decision units in an economic model.
(3) Economic models reflect the status quo in economic theorizing on welfare and growth, so that new alternative views (radical economics, supply-side economics, etc.) are hardly taken into account.
(4) Economic models are usually macroscopic in nature and neglect microbehavior, whereas the real roots for understanding economic behavior can be found at the micro level of individual decision making.

It is evident that many attempts that have been undertaken to link the environment to economic processes by means of environmental economic models are suffering from the same drawbacks as mentioned above. In addition to a general criticism regarding the "state-of-the-art" of economic modeling, it may be worthwhile, to pay specific attention to various classes of environmental economic models that have been used.

In the past decade, a wide variety of environmental–economic models has been designed that serve to portray some of the relationships of a complex economic–ecological system. A selection of these models is discussed here.

2.2.1. Materials balance models

Materials balance models aim at providing a comprehensive picture of an economy by means of flows and stocks of energy and materials that are governed by physical principles (*see* Ayres and Kneese, 1969). The basic ideas of these models emerged from the first law of thermodynamics (the conservation of matter and energy in all processes). This approach is fairly flexible, as it may be constructed at any spatial scale.

Some limitations of this model are:

(1) Its physical basis precludes an appropriate analysis of psychosomatic impacts of specific pollutants.
(2) Ecological processes are in general neglected.
(3) Various important economic aspects cannot be dealt with (the monetary part, societal choice processes, and so forth).

2.2.2. Input–output models

Input–output models provide a detailed description of the production side of an economy (sectoral linkages, input requirements, and deliveries of output). The input–output framework is flexible in regard to environmental repercussions, as it is also able to incorporate emissions of various pollutants, and pollution abatement activities (Leontief and Ford, 1972; Muller, 1979; Nijkamp, 1981). Input–output models have been applied numerous times in environmental economic research.

Despite its popularity, the input-output framework has also several limitations, such as:

(1) It is based on linear production processes based on past data.
(2) Input substitution (for instance, due to pollution abatement) is neglected.
(3) The vintage structure of new capital goods and the impacts of new technology (including abatement) cannot be dealt with in the static framework of input–output analysis.

2.2.3. Dynamic stock-flow models

Dynamic stock-flow models describe the trajectory of variables characterizing the structure and evolution of a part of the economy (e.g., forestry, agriculture) in relation to its environmental aspects. Such models are usually designed for specific planning purposes in a certain sector of the economy displaying a dynamic evolution pattern (e.g., optimal conservation strategies, fishery policy). Usually these models are partial in nature, focusing on one specific sector of the economy. Comprehensive models have been designed, at least in a conceptual stage, but only a few of them may be regarded as fully operational.

Dynamic stock-flow models are also hampered by various limitations, for instance:

(1) The majority of these models generate conditional pictures of the evolution of a certain sector, but fail to provide reliable predictions based on solid statistical/econometric techniques.
(2) These models usually fail to take into account consistency requirements with respect to national or global developments of all other sectors (e.g., dynamic additivity conditions in space and time).
(3) The behavioral character of many of these models is fairly limited: growth patterns are often generated by means of mechanistic time-dependent growth curves (e.g., logistic curves) without a clear linkage with behavioral choice patterns.
(4) The integration of policy measures and institutional configurations in such models is usually poor.

2.2.4. Spatially oriented environmental–economic models

Various environmental problems are local or regional in nature, so that models addressing local or regional environmental issues should have a clear spatial orientation (Spofford, 1976). In this regard various classes of environmental economic models have been designed, such as urban environmental quality models, (multi)regional production/pollution models, local land uses, and energy models. Such models may display various configurations (*see also* Issaev *et al.*, 1982): single area models, horizontally linked multiarea models, and hierarchical (multilevel) models. The latter class can be further subdivided into bottom-up models, top-down models, fully integrated multilevel models, and partially integrated multilevel models.

These models have found many applications, but have also some limitations, such as:

(1) Almost all spatially oriented models are static in nature, so that they do not generate dynamic evolution patterns of intertwined economic–ecological systems.
(2) The majority of these models deal with pollution and land use aspects rather than with ecological processes.
(3) Most of these models are unable to take into account drastic changes in the structure of the system (e.g., endogenous adjustments of technology), as they are usually based on a rigid input–output framework.

2.2.5. Evaluation models in environmental economics

Evaluation models serve to judge the feasibility and desirability of alternative courses of action, based on political choice and plausibility criteria.

After the failure of cost–benefit analysis to provide meaningful decision support to environmental–economic planning issues, a new class of evaluation models has originated in the late 1970s, viz. multiple criteria choice and evaluation models (*see* for a survey, among others, Nijkamp, 1980; Rietveld, 1980; Voogd, 1983). The aim of these models is to provide a rational basis for choice problems marked by conflicting priorities or conflicting objectives. These models have been designed and used in case of both small-scale discrete decision problems and large-scale continuous programming problems.

The following remarks can be made in regard to the content and use of such models:

(1) Many of these models are "comparative static" and hence not able to contribute to dynamic, sequential, or procedural planning problems.
(2) The political basis of evaluation models characterized by multiple actors (for example parties, interest groups, multiple levels) is not very strong.
(3) The degree of acceptance of results achieved by means of evaluation models is sometimes low in the political arena, as many decision agencies prefer to keep many options open.

The various classes of models discussed above indicate that economists have developed a variety of analytical tools for studying interactions between the economy and the environment. Many theoretical models have been designed aiming at placing externalities concepts in the framework of general economic equilibrium analysis, but these models have always stayed in a conceptual stage (cf. Baumol and Oates, 1975; Mäler, 1974). The more operational models have been fairly successful in that they provide a real-world picture of complex interactions between environmental and economic compartments. However, sometimes their theoretical and behavioral foundation was not impressive. In addition, the ecological content of many environmental–economic models has not been managed to contribute substantially to complex (and often conflicting) policy and planning issues.

2.3. Successes and Failures of Environmental–Economic Modeling

Clearly, a model has to be judged in the light of the targets set "a priori". Each purpose of a model presupposes some specific features of the model at hand.

As far as successes are concerned, the following points can be made:

(1) The strong quantitative tradition in economics has enabled us to include environmental elements fairly easily in conventional models.

(2) By linking environmental models to macro-oriented economic policy models, it was possible to investigate the effects of macro economic–environmental policies.

(3) At a more disaggregated level, a wide variety of (plan and project) evaluation models have been successfully operated (multicriteria analysis, computer consultancy systems).

(4) Various environmental tradeoff and choice problems have been elucidated by casting environmental problems into the framework of conventional economic modeling.

Despite the progress in environmental–economic modeling, it should also be admitted that the degree of success has not been overwhelming, at least not at the level of predictive and planning models. Clearly, lack of time series on major environmental variables has caused a great deal of suspicion regarding the reliability of model outcomes. But it also has to be mentioned that there are severe methodological problems that are hard to overcome while integrating economic and environmental models. The following issues should be mentioned in particular:

differences in time scales, differences in spatial scales, and differences in measurement levels of the successive variables (*see* Chapter 4 for details).

These differences between economic and environmental models have led to weakly integrated, less applicable, and hardly testable models. Especially in a planning and policy context, several models have exhibited many weaknesses. In this context, Wilbanks and Lee (1985) have pointed out five constraining factors for modeling efforts in the context of policy analysis:

(1) *Targeting and timeliness*. Analytical tools take a while to develop and test, so that they reflect questions, insights, and needs of the past, not the future.
(2) *Dependence on basic research*. The basic research community develops its research priorities in different ways from the policy world (for instance, by focusing on purely disciplinary paradigms), so that sometimes an array of resources is available to solve questions nobody is asking.
(3) *Gaps in knowledge*. Examples of such gaps are: cross-disciplinary interactions, human and institutional behavior (e.g. elasticity, inertia), impacts of exceptional events, feasibility of program implementations, existence, and effects of critical thresholds.
(4) *Limits to integration*. Despite the desire to undertake comprehensive policy analysis, there has been no good methodological ogical treatment of integrating contrasting approaches.
(5) *Lack of learning*. Very often, new models are designed for each particular policy problem without too much reference to already existing approaches. This lack of learning mechanisms from previous experiences (including ex post evaluation) may lead to an enormous waste of efforts.

The foregoing remarks have a straightforward relevance for environmental policy analysis. In conclusion, despite an undeniable progress in environmental–economic modeling, there is no reason for a *hubris* attitude (Timenes, 1982). Instead of claiming scope, applicability and certainty, economic–environmental scientists should be guided by modesty and tentativeness.

2.4. Perspectives

The area of environmental economics has not yet reached a stage of maturity, in which coherent and validated theories provide analysts with sufficient prior information to enable them to construct fully specified models. Information is lacking in regard to the following items:

(1) The mathematical form of relationships.
(2) The causal structure of the system concerned.
(3) The statistical properties of stochastic terms in models.
(4) The level of spatial aggregation.
(5) The measurement level of variables used in the analysis.
(6) The specification of the dynamic structure.
(7) The bridge principles between two different disciplines.

Clearly, the present state of environmental economics is able to generate *generic* structures, but altogether environmental–economic models are often semantically insufficient, leading to a high degree of specification uncertainty (cf. Leamer, 1978).

In the present section, a set of principles are presented that may serve as a frame of reference for judging and guiding economic–environmental modeling activities.

2.4.1. Design of models

From a methodological perspective, a model is a formal homomorphic projection of a complex real world system on a particular field of scientific interest (*see* Hafkamp, 1984). This can be illustrated in a simplified way by means of *Figure 2.1*.

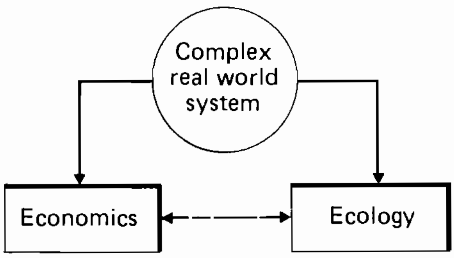

Figure 2.1. A multidimensional projection.

The existence of multiple scientific disciplines leads to the principle of *multidimensional projection*. Separate projections do not lead to crossdisciplinary problems, but simultaneous projections cause interaction problems between different dimensions (indicated by the horizontal arrow in *Figure 2.1*). Coherence and integration between such projections require special measures in the design stage, especially in regard to relevant time periods, choice of variables, spatial scale, measurement level of variables, and structure of the model. A minimum requirement for coherence and integration is a transformation (or linkage) of (attributes of) variables of the one dimension to the other one (*see* later). The multidimensional projection principle itself does not solve the problems inherent in attaining coherence and integration of models, but it provides systematics and completeness in the design stage.

2.4.2. Structure of integrated models

Usually, two different viewpoints are distinguished in dealing with multifacet models:

(1) *Horizontal*: all disciplines involved are regarded as equal constituents for building a multidisciplinary model.
(2) *Vertical*: one discipline is regarded as superior to the others, so that then the relationships from the dominant discipline to the remaining ones receive special attention.

These views often lead to contrasting approaches to modeling, though usually a dogmatic choice in favor of one of these viewpoints is not very fruitful. A more meaningful approach may be to adopt a so-called *satellite* principle.

This principle implies that the kernel of a cross-disciplinary model is made up by the key mechanism (driving forces) of a certain economic–environmental problem area (cf. van Lierop and Nijkamp, 1983). Completeness is not strived for in designing the core of a satellite structure.

The structure of the core of such a satellite approach can *inter alia* be identified by means of structure analysis, causality analysis, and specification analysis of complex systems (*see* Blommestein and Nijkamp, 1983; Brouwer and Nijkamp, 1984). Having identified the key mechanism of a cross-disciplinary model, all other (intra-, inter- or multidisciplinary) components can be added as nested derivatives of processes taking place in the core (*see Figure 2.2*).

Such a systematic and stepwise approach to building cross-disciplinary models is more efficient and promising than the design of fully integrated large-scale comprehensive models (*see also* Issaev *et al.*, 1982).

2.4.3. Information systems for complex systems

In recent years, various efforts have been made to develop up-to-date information systems by means of monitoring, retrieving, converting, modeling, or computer graphics (*see* Nijkamp and Rietveld, 1984). If information systems are to be successful in the context of

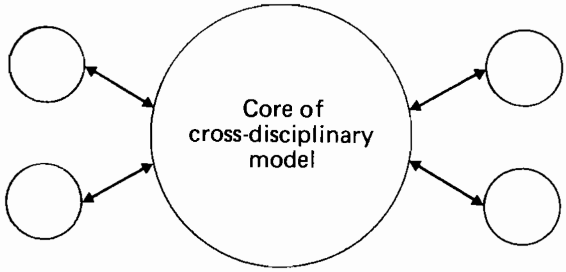

Figure 2.2. A satellite structure for a cross-disciplinary model.

environmental–economic models, they have to follow the above-mentioned principles of multidimensional projection and of a satellite structure. This means that information systems for an integrated environmental–economic system have to reflect systematically multiple components and dominance relationships for the interactions.

A major problem in designing information systems is their orientation toward past trends. In this regard, one may formulate the so-called *anticipatory* principle which states that information systems should be flexible enough to include all possible events that may be generated by different combinations of information inputs. Unfortunately, the development of such anticipatory information tools is lagging far behind in environmental–economic modeling, although such adaptive information systems are clearly a prerequisite for effective planning.

2.4.4. Scale and aggregation problems

In designing an environmental–economic model that describes a complex environmental system, one is always confronted with the problem of choosing the relevant spatial scale (the *areal unit* problem) and of combining data from different geographical scales (the *aggregation* problem) (*see* Blommestein and Nijkamp, 1983).

Several authors have shown that the results of many analyses are *scale-specific* (*see*, among others, Carter, 1974; Clark and Avery; 1976, Duncan *et al.*, 1961). Such results have not only been found in factorial ecology (employing small census data units), but also in statistical correlation analysis and econometric modeling (*see*, among others, Alker, 1969; Hordijk, 1979; Lohmoeller *et al.*, 1983; Nijkamp *et al.*, 1983). Very often the outcomes of such analyses lead to making statements regarding individual behavior from aggregate analyses, so that false conclusions are likely (*see* Openshaw and Taylor, 1981).

Aggregation may – in general – pertain to various dimensions in economic research: individuals, firms, areal units, time periods, and so forth. It leads to a condensation of information and hence to a loss of detailed insight (*see* Orcutt *et al.*, 1968), but it may enhance the understanding of complex phenomena by structuring the data so as to focus the attention on their important general features.

As far as aggregation of areal units is concerned, various *regionalization principles* can be used, such as the homogeneity principle, the administrative principle, the modality principle, etc. In addition to an aggregation of *areal units*, one may also distinguish an aggregation of *models* or *equations* pertaining to areal units. In this regard, one may analyze the impact of a particular aggregation level of individual units upon the explanatory power of a model or relationship (*see also* Van Daal, 1980; Akdeniz and Milliken, 1975).

In general, one may expect that – whatever the spatial scale of a model – the so-called *additivity condition* is always satisfied, so that the model results at a certain spatial scale are in agreement with those at a higher spatial scale.

In conclusion, an aggregate analysis will often exhibit results that are not in agreement with behavioral relationships specified at a disaggregate level. Unfortunately, there has been a strong tendency in economic and geographical research to use fairly aggregate data, as such data are normally easier to obtain. So, conventional aggregation analysis is without any doubt still relevant for the specification of environmental–economic models (especially as far as the additivity condition is concerned), but specific elements related to the structure of the environmental system at hand (e.g., spatial interdependences) have to be taken into account as much as possible.

2.4.5. Causal structure of models

In addition to scale and aggregation problems in environmental–economic models, the causality structure of these models and the related econometric problems are also of major importance. In terms of the analysis of the structure of a model, the following measurement levels may be distinguished (*see also* Brouwer and Nijkamp, 1984):

(1) A *binary* relationship. This indicates whether a certain variable has an impact on another variable. If the direction of impacts is the only information available, graph theoretical methods are a useful tool to analyze the causal structure of such models (*see also* Roberts, 1978).

(2) A *qualitative* relationship. When the sign of the binary relation (viz. positive, negative, or zero) is known, indicating the qualitative direction of the impact of the one variable upon the other

one, qualitative calculus can be applied to operationalize such models (*see also* Greenberg and Maybee, 1981).

(3) An *ordinal* relationship. When the order of magnitude of impacts is indicated in terms of rank numbers, the relationship can be studied by means of ordinal econometrics.

(4) A *cardinal* relationship. When the causal relationship between variables is quantified by path coefficients, path analysis can be employed, based on an ordinary least squares estimation procedure to estimate model parameters (*see also* Leitner and Wohlschlägl, 1980).

In the recent past, a great many statistical and econometric tools have been designed to deal with data and variables measured at either of these four scales (*see* Nijkamp *et al.*, 1985).

The use of such *causality* principles means a significant progress in analyzing and identifying the key structure of complex and dynamic models for environmental economic systems.

2.4.6. The element of time

It has already been mentioned that the time scale in economic and environmental models may be different (e.g., different time horizons, discrete *versus* continuous dynamics). The problem of different time scales can be analyzed in two ways, viz. by means of the *discount principle* and by means of *discrete–continuous dynamic models*.

A discount rate is in general used in a policy or decision model in order to calculate the present value of a variable by transferring its future values into the present. The theoretical background of the traditional discount principle is very simple, as a discount rate serves to indicate the relative importance of a unit of money in the next year with respect to the current year. Conventional capital theory shows how such a discount rate can be linked to the marginal efficiency of capital in a free market system. In this way, the discount principle has often been used in cost–benefit analysis. The discount rate can also be used in a utility framework (for instance, in a dynamic programming or optimal control model) by indicating the decreasing importance of future values of the utility function. Under certain stringent assumptions, this discount rate can also be linked to the interest rate on capital.

These conventional views on discounting are based on the assumption that attributes (e.g., impacts) of decisions can be made commensurable by transforming them into a common monetary denominator or a common utility denominator. Given the externalities character of environmental problems it is generally impossible to find such a common denominator. This problem is once more serious, as in many situations the time horizon of economic decisions is different from that of environmental decisions. The problem of different time horizons can be reformulated as a problem of different discount rates in economic and environmental planning. Each time horizon corresponds to a certain discount rate, so that this problem can essentially be tackled by using different discount rates for different objectives of interest, so that then a coordination of models from different disciplines is not necessarily precluded. A similar conclusion can be drawn for spatial discount problems (caused by a spatial lag structure) or spatiotemporal lag structures. Thus, the *discount* principle may reconcile different time viewpoints in cross-disciplinary models in a way similar to thematic maps in mathematical space-time geography (cf. Polya, 1981).

The same conclusion can be drawn for discrete and continuous dynamic models. Discrete processes can be approximated by means of continuous models, though then also a (slight) adjustment via the discount rate is necessary. This holds also true for models characterized by bifurcations, singularities, or catastrophes (e.g., in a nonlinear dynamic setting).

Clearly, integration of economic and environmental issues in modeling requires a discounting for both economic and environmental objectives, as otherwise in a long-run context the module without a discount rate would always dominate the discounted module.

2.4.7. Conflict analysis

Conflicts may be the result of a multidimensional projection, as the latter operation reflects different components of a model which – due to functional linkages or policy interest – cannot be reconciled in terms of an evaluation of the outcomes of a choice made by decision makers. Conflicts (i.e., trade-offs to be made among diverging objectives) may even emerge within one component.

A separate optimization of objectives leads to infeasible solutions, so that conflict analysis is essentially dealing with reconciling diverging interests. In this respect, a *compromise* principle may be formulated which states that a "satisfier" solution can be achieved by aiming at minimizing the discrepancy between a certain ideal point solution (or reference solution) and the actual possibility frontier generated by the model (*see* Hafkamp, 1984; Nijkamp, 1980; Rietveld, 1980; Voogd, 1983). This method can easily be used in an interactive context and is then named the "displaced ideals method" or the "reference point method". Various applications have demonstrated the operational nature of the compromise principle in case of conflicting economic–environmental issues. In this regard, the recently developed multiple criteria methods and multiple objective programming methods offer a great potential for integrated economic–environmental planning models.

2.5. Summary and Conclusion

Can economic modeling contribute to providing more insight into environmental problems and into solution strategies for such problems? There are no doubt various limitations in the significance of environmental–economic models to solving environmental problems, caused *inter alia* by the following factors:

(1) Lack of data and lack of insight into interwoven economic–ecological processes.
(2) Lack of information on the economic aspects of environmental decay and environmental preservation.
(3) Lack of insight into the effectiveness of various policy instruments in economic–environmental management.

Despite many shortcomings in modeling activities, various positive consequences of environmental–economic modeling can be observed. Examples are:

(1) An adequate insight into the contributions of industrial sectors to air and water pollution.
(2) A reasonable insight into the sectoral effects of emission standards.
(3) An adequate amount of information on the macroeconomic consequences of antipollution policies.
(4) An increasing use of appropriate plan and project evaluation techniques in many public agencies.
(5) A continuing improvement of environmental policy instruments at a local and regional level for various environmental quality aspects.

In general, one may draw the conclusion that environmental–economic modeling has provided a great many stimuli for an improved environmental management by providing the tools for measuring, evaluating, and trading off environmental–economic aspects of the physical surroundings of mankind.

ECONOMIC-ECOLOGICAL MODELING
L.C. Braat/W.F.J. van Lierop, (Editors)
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CHAPTER 3

Ecological Modeling: Shortcomings and Perspectives

J.N.R. Jeffers

3.1. Introduction

The origins of ecological models lie in the mathematical representations of relatively simple interactions, such as those of predator/prey organisms, developed by Lotka and Volterra (Lotka, 1925; Volterra, 1926). These representations were principally analytical, in the sense that the mathematical formulations were capable of analytical solution. Models of this kind began to attract the attention of the more mathematically inclined ecologists shortly before World War II, but were of interest then to only a very small minority of biologists and ecologists. With the development of applied mathematics that took place under the influence of operational research during World War II, and the application of these new ideas in mathematics to a wide range of phenomena and practical issues, it was inevitable that at least some of the new schools of ecology that were formed in the post-World War II expansion of education and academic research would begin to explore the application of operational research techniques and statistical mathematics to ecological relationships. This extension was hastened by the emergence of the electronic computer, first in analog form, and then as the general purpose, programmable digital computer.

Progress might still have been slow, if the interest had been confined to academic research – characterized by a distinction between "models" as mathematical expressions of general laws and "simulations" as empirical equations describing the behavior of ecological processes. "Whereas a good simulation should include as much detail as possible, a good model should include as little as possible" (Maynard Smith, 1974).

It was, however, probably the International Biological Program (IBP) which gave the greatest stimulus to the development of ecological models. The need to understand and predict the biological productivity of a wide range of biomes – including tundra, grasslands, temperate forests, freshwater systems, and tropical forests – generated a need to define biological processes in mathematical terms. Much of the emphasis in IBP was on whole-system models, an ideal which was quickly found to be impractical, as was the method of collecting data first and modeling the processes subsequently, instead of defining the systems carefully first and then collecting the data to characterize the systems. Nevertheless, it was the IBP that brought mathematical models and systems analysis to the general attention of ecologists and biologists. The need to link ecological models to the economic and social sciences also emerged at much the same time, because it became necessary to consider biological productivity in relation to agricultural and forest policies, and to land management practices.

Models are formal expressions of the essential elements of a problem in either physical or mathematical terms

Figure 3.1. A definition of models.

Figure 3.1 contains the definition of models implied in this Chapter, i.e. as formal expressions of the essential elements of a problem in either physical or mathematical terms. Before looking more closely at the ecologist's experience of modeling as given in this definition, however, it is useful to consider why ecologists should feel that they need models of this kind.

The need lies in the complexity of *ecology as a science*. Ecologists are concerned with the interactions of organisms with each other, and with the interaction of organisms with their environment. These interactions are many-sided, and are dynamic, in the sense that they are time dependent and constantly changing. Furthermore, the interactions frequently have the feature which the engineer calls "feedback" i.e., the carrying back of some of the effects of a process to their source or to a preceding stage so as to strengthen or modify the effects. Such feedback will sometimes be positive, in the sense that the effects are increased, and sometimes negative, in the sense that the effects are decreased. The feedback may itself be complex, involving other positive and negative effects, with various results depending upon a series of environmental factors.

The complexity of ecosystems and of ecology is not, however, confined to the presence of multiple interactions in the relationships between organisms, and between organisms and their environment. Living organisms are themselves variable – indeed, variability is one of their essential biological characteristics. This variability may be expressed in terms of effects on other organisms, for example by competition or by predation, or it may be expressed in the response of the organisms, either collectively or singly, to environmental conditions. Such responses will be reflected in variable rates of growth and of reproduction, or even in variable ability to exist under markedly adverse conditions. When this characteristic is added to independent variations in environmental factors such as climate and habitat, ecological processes and ecological systems become difficult to investigate, to predict, and to control.

The ecologist, therefore, finds himself facing many difficult problems in the understanding of even relatively unmodified ecological systems. The traditional response of the scientist to such difficulties has been to focus attention on small subsets of the real problem. In ecology, for example, much academic research has been concentrated on the behavior of single organisms in simplified habitats, for example on flour beetles in bags of flour, or on Enchytraeid worms in selected media. Alternatively, the competition between two or three species, again in relatively simple habitats, has been studied extensively. In all of these approaches, an attempt is made to reduce the complexity to a level which is manageable by traditional methods of investigation and experimentation, and by eliminating as many as possible of the sources of variability. Even when everything possible has been done to simplify the system, the interrelationships often remain difficult to understand and to predict.

The principal goal of *applied ecology*, however, is to predict the effects of deliberate modification of complex ecological systems, and here a further dimension of variability and interaction is introduced. In some fields of applied ecology, for example, forestry and agriculture, simplification of the agricultural system may be achieved by considering the response of the crop species alone, but the ecologist frequently also needs to have information on the response of the whole system to modifications introduced by changes in management. For example, the effects of the crop species on the soil, and on organisms associated with the ecosystem on which the crop has been imposed, may be an important issue in deciding on the desirability of the proposed intervention. The extension of these ideas to the ecological effects of land use, where several alternative strategies for land use and environmental management are considered, is even more difficult. Research on the long-term management of natural or seminatural ecosystems, for example, or on the management of nature reserves to ensure the conservation of wildlife, requires the encompassing of the complexity and variability of the many species contributing to the stability or resilience of the ecosystem.

For all the above reasons – i.e., the inherent complexity of ecological relationships, the characteristic variability of living organisms, and the often counterintuitive effects of deliberate modification of ecosystems by man – the ecologist requires an orderly and logical organization of his research which goes beyond the sequential application of tests of hypotheses, although the "appeal to nature" invoked by the experimental method necessarily remains at the heart of this organization.

> Systems analysis is the orderly and logical organization of data and information into models, followed by the rigorous testing and exploration of these models necessary for their validation and improvement

Figure 3.2. A definition of systems analysis.

Applied systems analysis, as defined in *Figure 3.2*, provides one possible format for that organization, a format in which the experimentation is embedded in a conscious attempt to model the system, so that the complexity and the variability are retained in a form in which they are amenable to investigation and analysis. Models provide a way of thinking about complexity and of summarizing theories and assumptions which lie behind attempts to explain and describe that complexity. The language of mathematics, in its many forms, provides the only hope for the ecologist to test the incompatibility of the many existing theories. Verbal descriptions of this complexity are inadequate, partly because of the ambiguity of the words themselves, but also because human languages do not usually focus attention upon the essential elements of the problem, and do not provide a basis for the deductive logic which is alone capable of distinguishing between competing theories.

The advantages of *mathematical models* are that they are precise and abstract, that they transfer information in a logical way, and that they act as an unambiguous medium of communication. They are precise because they enable predictions to be made in such a way that these predictions can be checked against reality by experiment and by survey. They are abstract because the symbolic logic of mathematics extracts those elements, and only those elements, which are important to the deductive logic of the argument, thus eliminating all the extraneous meanings which may be attached to words. Mathematical models transfer information from the whole body of knowledge of the behavior of interrelationships to the particular problem being investigated, so that logically dependent arguments are derived without the necessity for all the past research to be repeated. Such models provide a valuable means of communication because of the unambiguity of the symbolic logic, a medium of communication which is largely unaffected by the normal barriers of human language.

The disadvantages of mathematical models lie in the apparent complexity of that symbolic logic, at least to the nonmathematician. In part, this is a necessary complexity – if the problem under investigation is complex, it is likely, but not necessary, that the mathematics needed to describe the problem will also be complex. There is also a certain opaqueness of mathematics, and many people have difficulty in translating from mathematical results to real life. Failure to interpret correctly the results of mathematical analysis is evident in many papers submitted to scientific journals.

3.2. A Review of Ecological Modeling Approaches

The ecologist's experience of mathematical models, over a wide range of different applications, has led to the recognition of families of mathematical models that can aid the selection of appropriate models for particular kinds of problems. *Figure 3.3* illustrates some of the more important families of models which have been used in ecology. The list is far from exhaustive, and the categories are also not mutually exclusive. The list is, however, sufficient to illustrate the mathematical

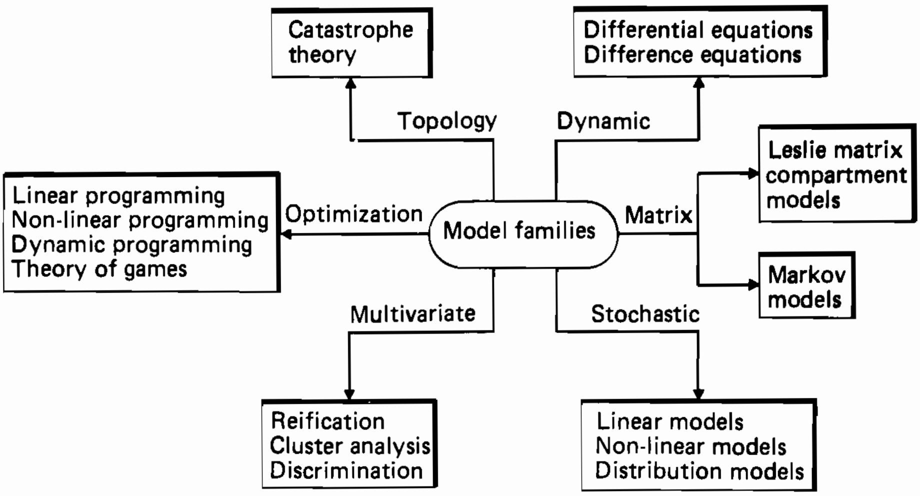

Figure 3.3. Various kinds of models used in ecology.

models applied to ecological problems and the principal characteristic properties of the various model families.

3.2.1. Dynamic models

Dynamic models consist of sets of differential or difference equations providing more or less complex mathematical descriptions of the processes or ecosystem functions being simulated. Characteristically, they provide deterministic and functional expressions for levels of various types which change at rates controlled by decision functions. These level equations may represent accumulations within the system of such variables as weight, numbers of organisms, and energy, and the rate equations govern the changes of the levels with time. The decision functions represent policies or rules, explicit or implicit, which are assumed to control the operation of a system.

The popularity of dynamic models arises from the great flexibility of the methods used to describe systems operation, including nonlinear responses of components to controlling variables, and both positive and negative feedback. This flexibility has some disadvantages, and it is, in any case, usually impossible to include equations for all the components of a system, as the simulation then rapidly becomes too complex. It is, therefore, necessary to obtain an abstraction based on judgment and on assumptions as to which of the many components are those that control the operation of a system.

Dynamic models have an intuitive appeal to many ecologists, especially if they have some reasonable mathematical background. The formulation of the model allows considerable freedom for constraints and assumptions, and allows for the introduction of the nonlinearity and feedback which are apparently characteristic of ecological systems. The ecologist focuses on processes, and is able to mirror or mimic the behavior of the system as he understands it, and gain some useful insight into the behavior of the system as the result of changes in the parameters and driving variables. Even where the values of parameters are unknown, relatively simple techniques exist to provide approximations for these parameters by sequential estimates. In particularly favorable cases, it may even be possible to test various hypotheses about parameters or functions. The implementation of dynamic models has been greatly helped by the existence of special-purpose modeling languages such as DYNAMO and CSMP.

A very large part of the ecological modeling which has so far been attempted falls within this category of model. In nearly all cases, the early stages of the investigation of a complex ecological problem will usually be attempted by dynamic models, thus concentrating attention on the basic relationships which are assumed to underlie the system, and defining the variables and subsystems that the investigator believes to be critical. In the later stages of investigation, it may often be preferable to switch the modeling effort to one of the other families of models.

3.2.2. Matrix models

Dynamic models offer almost complete freedom to the investigator for the expression of those elements considered to be essential to the understanding of the underlying relationships between those variables and entities that are identified in the description of the system. The model strives for reality – a recognizable analogy between the mathematics and the physical, chemical, or biological processes – sometimes at the expense of mathematical elegance or convenience. The price paid for this "reality" is frequently the necessity to multiply entities to account for relatively small variations in the behavior of the system, or difficulty in deriving unbiased valid estimates of the model's parameters. Matrix models, in contrast, represent one family of models in which "reality" is sacrificed to some extent in order to take advantage of the property of particular mathematical formulations. Deductive logic then enables the modeler to examine the consequences of his assumptions without the need for time-consuming experimentation on the model.

The most widely used form of matrix model in ecology is the Leslie matrix, a deterministic model which predicts the future age structure of a population of animals from the present known age structure and assumed rates of survival and fecundity. Mathematical analysis of such matrices determines the stable structure of the population without the necessity for time-consuming calculations. This example illustrates the basic reason for using a more restrictive formulation of the mathematics, in that relatively simple calculation reveals the principal properties of the model. A wide range of modifications of this model have been explored and have been described by Usher (1972).

Matrix models represent one family of models in which the 'realism' of the model is partly sacrificed in order to obtain the benefit of the mathematical formulation. The same formulation also imposes constraints upon the way in which the models can be used, but these constraints are balanced by the convenience of the computations and by the relative ease of establishing the values of the basic parameters. However, matrix models represent a relatively neglected family of models in ecology. Only a few research workers have published applications of such models, perhaps in part because of the unfamiliarity of biologists and ecologists with matrix algebra, although the differential and difference equations of dynamic models perhaps make an even greater demand on the mathematical ability of the modeler.

3.2.3. Markov models

Markov models are a hybrid between the matrix models described above and stochastic models described below. In these models, the basic format of the matrix is retained except that the matrix contains a series of transition probabilities from one state to another of the system at a specified timestep. In such a model, the future development of a system is determined by its present state and is independent of the way in which that state has been reached. Many ecological systems exhibit near Markov properties and Markov models have some useful properties, including:

(1) Algebraic analysis of the transition matrix itself determines the existence of sets of states which are transient, closed, or absorbing. Further analysis enables the transition matrix to be partitioned, and the submatrics investigated separately, thus simplifying the ecological system being studied.
(2) Analysis of the transition matrix provides estimates of the mean times to move from one state to another, and the mean length of stay in a particular state once it has been entered.
(3) Where closed or absorbing states exist, the probability of the system reaching those states, and the mean time to absorption, can be calculated.

3.2.4. Stochastic models

Stochastic models incorporate probability theory in mathematical modeling. The simplest examples of stochastic models include the use of statistical distribution theory to describe the variation of numbers, growth, or density of organisms in space and time. Experimental and survey designs are frequently used to determine the variation in different parts of ecological systems, and the linear additive model of the analysis of variance is used to estimate population variances from sample observations. Perhaps the most widely used of all the stochastic modeling procedures is that of regression analysis, a form of analysis which is frequently misused in an attempt to estimate the parameters of functional relationships, but which, with proper safeguards, can provide estimates of the variability of ecological processes and populations. Although stochastic models are quite widely used, they are frequently not recognized by ecologists as a form of ecological modeling. In part, this is because the traditional training of applied mathematicians does not include statistical mathematics and probability theory, with the result that only a limited range of mathematical techniques have been deployed in the solution of practical problems. Nevertheless, the rapid growth of statistical mathematics in biology and ecology, which followed the early publications by R. A. Fisher, has added a whole family of modeling techniques to the ecologist's armory (Fisher, 1925).

3.2.5. Multivariate models

Multivariate models represent a branch of statistical mathematics which has grown rapidly with the greater availability of the modern computer. The ability to investigate the relationships between large numbers of variables has led to the development of powerful techniques of ordination, discrimination, and canonical correlation. These techniques enable the ecologist to manipulate and analyze the very large numbers of variables characteristic of ecological problems. Most ecological systems, for example, contain large numbers of species whose numbers, density, or mass, vary in complex ways. Modeling of this complexity, and, particularly, the reduction of the complexity to the smallest possible number of independent dimensions has represented a fruitful source of research and ecological theory.

3.2.6. Optimization models

Of all the techniques that were developed during the Second World War, that of mathematical programming was perhaps the best known. The ability to find the optimum value of a defined objective function, within constraints defined by other equations or inequalities, stimulated a great deal of interest and application. The extension of the linear programming methods to nonlinear and dynamic programming, together with the associated theory of games, has found some applications in the broad field of ecology, especially where it has been necessary to relate ecology to economic and social factors. However, it has to be admitted that most optimization models are inadequate to capture the complexity of ecological systems and represent oversimplifications which may, in many instances, be extremely dangerous. Nevertheless, optimization models are quite widely used in ecology, especially by nonecologists wishing to manipulate ecological systems within defined economic and social constraints.

3.2.7. Topological and other models

In recent years, the topological model of catastrophe theory has attracted a good deal of attention, particularly among ecologists who can recognize in their systems the characteristic properties of catastrophe models, i.e., the presence of hysteresis, bimodality, divergence, and discontinuity. Relatively few valid applications of catastrophe theory have emerged from the literature, but the theory has a good deal of appeal, principally because its expression is highly visual, and because it often seems to provide some "explanation" of relatively complex phenomena. Other models such as network models, decision theory models, and, more recently, fuzzy set theory, have had some limited application within ecology, although there is as yet very little description of their use in the published literature.

It is not appropriate, in this chapter, to provide an extensive list of case studies for the various types of mathematical models that have been applied in ecology. However, there are several collections of case studies which provide a useful starting point for anyone wishing to follow up the application of mathematical models in ecology. Some of the principal sources of such references are given in *Table 3.1*.

This review of the available families of mathematical models has deliberately emphasized the very wide range of mathematical techniques available for the modeling of ecological systems. The extent of this range needs to be emphasized because it is surprising how limited a repertoire of methods has so far been exploited in practical applications. The mathematics exists to provide very exciting possibilities for the modeling of ecological processes at all levels, without losing the richness and complexity of ecological relationships and interactions. Truth is more likely to lie at the intersection of many models of different types than in the elaboration of only one type of model.

3.3. Success and Failure of Ecological Modeling Approaches

It has to be admitted that, to date, only limited success has been achieved in the use of ecological models. Early attempts to model ecological systems, especially in the International Biological Program, were overambitious, and tended to concentrate on whole-system models which rapidly became too big and too complex to serve as adequate simulations of the systems which they were intended to represent. More limited attempts at modeling have often been more successful, but have suffered from difficulties in being able to generalize from one point in time or space to some wider population or ecosystem.

Perhaps the most striking failure of mathematical modeling in ecology owes its origins to conspicuous failures to define adequately the system which is being modeled. Attempts to use mathematical modeling without embedding the models in an overall structure of systems analysis have led to the inadequate representation of those parts of ecological systems for which there is a mathematical description of the

Table 3.1. Case studies for ecological modeling.

Dynamic models
Hall and Day, 1977
Innis and O'Neill, 1979
Jørgensen, 1979
Ott, 1976
Patten, 1971, 1972, 1975, 1976
Shugart and O'Neill, 1979
Matrix models
Searle, 1966
Usher, 1972
Markov models
Usher, 1979
Stochastic and multivariate models
Daget, 1976
Green, 1979
Marriott, 1974
Optimization models
Van Dyne *et al.*, 1970
Topological models
Jones, 1975
Poston and Stewart, 1978

input, output, and state variables. Added to this difficulty has been the failure to observe the mathematical assumptions inherent in the use of any mathematical technique. This failure has been particularly characteristic of the use of statistical techniques, where there are frequently important assumptions which determine the validity of the procedures being used.

This failure to use an appropriate framework of systems analysis, and to pay sufficient regard to the inherent assumptions in the mathematical techniques that underlie the modeling of ecological processes, has frequently derived from an unfortunate characteristic of almost all ecological research, i.e., that of collecting data first and then attempting to model the collected data empirically. This method has emerged as a feature of all the principal initiatives in international ecological research, notably in the International Biological Program and in the Man and the Biosphere Program of UNESCO. Instead of defining the problem carefully, and the ecological system that is an inherent part of that problem, before collecting any data, there has been a compulsion to mount large experimental or survey programs for the purposes of obtaining empirical data. It has then been assumed that some ad hoc system of modeling will enable these data to be used to construct an appropriate and valid model. Systems analysts have continued to point out to ecologists the dangers and impossibilities of this procedure, but largely to no avail. The fact that the ways in which data are collected and the purposes for which they are collected may inhibit subsequent manipulation of the data cannot be stressed sufficiently strongly, but certainly has not yet had any major effect upon the ways in which ecologists, in developed and developing nations alike, manage their research programs.

As a result, it has come to be accepted that only "simple" models have any value in ecological research and management. This statement, however, represents a paradox. If, as has been argued in the introduction to this paper, the principal reason for using mathematical models in ecology is because of the complexity of ecological relationships, it follows that a model which is simple is almost certainly a caricature or distortion of those complex relationships. The "simple" model is unlikely to carry the complexity which the ecologist was at pains to explore in the first place. That caricature may, therefore, prove misleading, especially if the investigator then attempts to combine this simplified view of the ecological relationships with equally simplified views of economic and social processes.

3.4. Summary and Conclusion

The outcome of this attempt, over the last 40 years, to introduce mathematical models into ecology is perhaps discouraging. Ecologists have not yet achieved the success that they had hoped for in the introduction of mathematical models to ecological thinking. Too few ecologists know sufficient mathematics to work easily in this field, and the mathematicians themselves have come from too narrow a field of mathematics to exploit the power and flexibility of mathematics in the representation of the complexity of ecological processes. The outcome has been an undesirable one, i.e., of a relatively small range of mathematical models seeking an application within ecology, rather than a search for appropriate forms of mathematical representations of ecological processes as they are understood and revealed by systems analysis.

Whether the same situation exists in economics and in the social sciences, the paradigm of the linking of submodels of economics and social science with submodels of ecology has not been achieved. At best, a few economic or social variables have been incorporated into ecological models, or, conversely, a small number of ecological variables have been incorporated into economic and social models. The link between ecological submodels in their full and necessary complexity, and economic and social submodels, at their own level of complexity, has still to be made.

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CHAPTER 4

Integrated Economic–Ecological Modeling

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4.1. Introduction

In the previous chapters it has been shown that a great variety of modeling methods has been developed in economics as well as in ecology. As the underlying mathematical and statistical techniques used in both disciplines are generally the same, one would think that integration of economic and ecological models is essentially feasible. This assumption is further supported by the observation that the dynamics of economic and ecological systems can be described by similar equations (*see*, e.g., Odum, 1983). This suggests that interdisciplinary modeling advocated by some systems scientists and engineers has a definite theoretical basis. This would imply that in-case integration is not easily accomplished in practice, that either the mathematical and statistical possibilities were not fully explored, or that there are problems in other aspects of the models, such as data and temporal and spatial dimensions.

We employ two complementary definitions of *economic–ecological models*. In an *operational* sense, economic–ecological models are those that are capable of assessing the relevant impacts of the socioeconomic activities on ecosystems, as well as the relevant effects of the state and development of ecological systems on socioeconomic activity. In a *structural* sense, economic–ecological models are models in which both the economic and the ecological phenomena relevant to a particular problem, as well as the relationships between socioeconomic activities and ecological processes essential to the problem are included in an adequate manner.

In other words, adequate economic–ecological models consist of sets of variables or submodels which are accepted as monodisciplinarily adequate by economists and ecologists respectively. This may even refer to submodels which consist of internally unrelated sets of variables, if these loose sets are generally accepted as a clear characterization of key elements of the pertinent systems. Models in which the latter condition is not met, should, however, not be included in the class of economic–ecological models. Moreover, the modeled relationships between economic and ecological variables should not only reflect the structure or function of the real-world relationships but should allow for effective transfer of information from one submodel to the other.

The *objectives* of this chapter are:

(1) To identify possibilities to integrate economic and ecological modeling approaches to environmental and resource problems.
(2) To identify theoretical requirements and technical problems of the integration of economic and ecological theory and data in models.
(3) To prcsent an overview and evaluation of existing integrated economic–ecological models.
(4) To explore and propose solutions to technical problems.

In Section 4.2. we address the first objective. Each of the other objectives will be treated in a subsequent section. Clearly, in this chapter the focus is on the technical aspects of modeling. The practical side of modeling for resource and environmental management is treated in Part II of this book.

4.2. Possibilities to Integrate Economics and Ecology in Models

4.2.1. Introduction

There are many ways to build integrated economic–ecological models. For example, economic–ecological models can be constructed by the integration of existing models or technical model types. Given the great number of models, methods, and techniques developed in economics and in ecology and the experience with their successes and failures, it seems logical to explore the possibilities of integrating monodisciplinary modeling approaches into multidisciplinary models.

This approach is sometimes referred to as the "compartment modeling" approach. The main research question in this area is: which mathematical formats are available to transform the output of one monodisciplinary model into input with the characteristics required by the other one?

A second way to develop an economic–ecological model is rooted in so-called "general systems theory" (*see* e.g. Von Bertalanffy, 1968) and is often called ***holistic modeling***. Following this approach, the models are typically built as one consistent model, instead of being put together from separate, monodisciplinary submodels. To achieve consistency the modelers generally employ one single technique (simulation, dynamic programming) and they often use a single denominator for the variable quantities.

A third way would be to start from either an ecological or an economic model and expand this monodisciplinary model to such an extent that a multidisciplinary model is realized.

4.2.2. Models in environmental and resource economics

In Chapter 2, Nijkamp reviewed a number of environmental economic models and indicated several strong and weak points of these models. Here we shall briefly discuss models and techniques currently employed both in environmental and in resource economics, with the explicit purpose of identifying possibilities for matching them with the models and techniques used in environmental and resource ecology.

Two major approaches can be distinguished:

(1) Extending traditional cost–benefit analysis.
(2) Extending physical–economics models with resource inputs and waste output.

The objective of the first approach is to internalize environment and resources (as so-called "extra market effects") into the economic system so that they can be quantified in terms of money. A proxy for their value to the economy is the willingness to pay for them. These internalization strategies generally require very little information about the state of the environment as such, or about impacts of use. All that is needed is the present availability of resources. For the rest they rely on information from the producers and consumers about preferences and substitutability of resources for production.

In case cost–benefit analysis is extended with the time rate of nonrenewable resource use, still very little environmental input is required. Models listed by Hufschmidt and Hyman (1982) deal mostly with market price, substitutability, storage capacity, and depletion rate, which is the rate of extraction. Availability estimates are again the only environmental data used. Only when renewable resources are involved some ecological information is required, namely the rate of replenishment of stocks, for example the growth of harvestable populations, the development of fertile soils and the flow rate of freshwater streams for reservoir build-up. Finally, in multiobjective cost–benefit analyses environmental standards based on ecological and toxicological data may be included as constraints and criteria.

In environmental economics there is a line of work which concentrates on valuation of the economic impacts of pollution via monetary damage functions for physical damage (Opschoor, 1974). Physical ecological and direct human health effects have to be estimated as a basis for the monetary evaluation, so ecological data are in fact required. Much research has been devoted to finding the optimal way of pollution control (*see*, e.g., Pearce, 1976). Here again, environmental quality standards are necessary for implementation, so information from natural science research is required.

The second approach in environmental economics utilizes most explicitly resource input and pollutant output variables. This approach has been discussed extensively by Nijkamp (*see* Chapter 2). They include the materials-balance residuals management models and the extended economic input–output models.

In the IvM–IIASA survey of economic–ecological models (Braat and van Lierop, 1984; *see also* Appendix A) environmental economic models include descriptive energy–economy models (Slesser, 1982; Kaufmann and Hall, 1982) and a series of predictive "economics plus emissions (and emission control)" models (Bossel, 1981; Hanson, 1980; Bolzern and Fronza, 1982; Houweling, 1982; Muller, 1979; Orishimo, 1982). A model by Beddington and McAllister (1981) has been used to calculate the economic impacts of reopening herring fisheries on the North Sea. In addition three prescriptive models are included. Allen (1983) has developed a linear programming model for fuelwood plantation planning, Highton and Webb (1981) have calculated optimal allocations for energy production considering pollution abatement costs, and Werczberger (1974, 1976) has applied a mixed integer and a goal programming model to introduce air quality considerations into land use planning. Finally, two environmental economic models were found which combine several techniques. Hafkamp (1984) has developed an interregional economic model with emissions and emission dispersion. Multiobjective decision making procedures are subsequently used to work with the model. McKelvey (1982) has analyzed a multispecies fishery with a simple analytical model which is subsequently used for optimization of catch.

4.2.3. Models in environmental and resource ecology

As is the case in economic modeling, two lines of extending ecological modeling can be distinguished:

(1) Ecological evaluation models.
(2) Resource and pollution impact models.

One group of ecologists has concentrated on developing methods, techniques, and theory to valuate ecosystems, resource flows and environmental services in a manner analogous to the economist's ways to valuate capital goods, production factors, and products. These attempts in most cases originated from the perceived necessity to provide a matching quantification of ecological aspects in cost–benefit analyses of intended development projects. Three types of ecological valuation can be distinguished: (1) monetary, (2) energy, and (3) multidimensional.

In monetary evaluations the values of ecosystems and ecosystem functions are priced. A price is estimated by essentially economic means. In energy evaluations the whole project or system is evaluated in (embodied) energy terms to provide a common basis for cost–benefit analysis (Odum and Bailey, 1976; Odum, 1984; Costanza, 1982; Hannon, 1984; Lavine, 1984). The third method does not employ a single common denominator, but instead comparatively evaluates ecosystems using biological indicators such as diversity of species, number of rare species, naturalness, etc., and subsequently ranks them, for example from "too valuable to lose" to "not so valuable" (i.e., can be developed for economic purposes). These methods have received much attention in the United Kingdom for nature conservation management (*see* Ratcliffe, 1976) and in the Netherlands for land use planning (*see* Van der Ploeg and Vlijm, 1978; Braat *et al.*, 1979; Van der Ploeg, 1985).

The first approach requires socioeconomic information. Energy evaluation requires, in most techniques, some information on energy prices and if the economy is reevaluated in energy terms, an extensive understanding and knowledge of economics is required. In the ecological evaluations no economic input is required.

The second line of extending ecological analysis is constituted by the ecological models which incorporate variables to indicate exploitation and pollution impacts. A great number of pollution models have been developed over the last decade. They include general pollution impact models, eutrophication models, and ecotoxicology models, dealing with, e.g., heavy metals, PCBs, and radiation. They cover all sorts of terrestrial and aquatic ecosystems, lake models being the most abundant (*see* Jørgensen, 1979; Van Steenkiste, 1978; Rinaldi, 1982). The same references include also a variety of publications on resource ecology, notably both abiotic resources (water, soils) and biotic resources (agriculture, fisheries, and forestry).

The IvM–IIASA survey of economic–ecological models (*see* Appendix A) produced a small number of extended ecological models. A set of dose–response equations describing the relationship between groundwater extraction and vegetation has been presented by Reijnen and Van Wiertz (1981). Four models pertain to water quality of lakes (Benndorf and Recknagel, 1982; Leonov, 1982; Leonov and Toth, 1981; Virtanen, 1981). These models do not include any economics. The three remaining models are applied in agricultural situations and do include cost and profit factors related to ecological production factors (Sutherst *et al.*, 1979; Swaney *et al.*, 1981; Wilkerson *et al.*, 1981). Point (1983) has developed an essentially ecological optimization model which minimizes costs in nitrogen emission management. An extensive set of ecological constraints is combined with an economic objective function. These ecological resource and environment models and analyses do not involve economic modeling. They include input or output flows, the source and destination of which, respectively, is implied to be the socioeconomic system.

4.2.4. Conclusions

Looking back at the models described in this section and at those described by Nijkamp and Jeffers in Chapters 2 and 3, respectively, we conclude that as far as *contents* are concerned the Input–Output and Material Balance models could very well be matched with those ecological models that study harvestable populations (fish, trees) and with those that model the impacts of pollutant inputs. For outdoor recreation modeling, where both resource use and pollution and disturbance inputs to ecosystems are at hand, the same conclusion holds.

As to the *format*, it is hard to judge whether individual models can be integrated. Within the scope of the IvM–IIASA survey project (*see* Appendix A), it was impossible to test empirically the possibility of coupling the models which were received. In theory, however, problems of temporal, spatial, and mathematical format differences can be solved if data are available to support technical conversions and adaptations. We shall therefore look at these technical problems in the next section.

4.3. Theoretical Requirements and Technical Problems

4.3.1. Model building procedure

In theory, economic–ecological modeling involves the same principles and technical rules as standard monodisciplinary modeling. Additional hypotheses need to be formulated, however, about the relationships between economic activities and ecological components and processes. The spatial and temporal dimensions of variables may differ widely between economic and ecological models which concern a similar or even the same problem. One frequently occurring reason is that the data have been collected at different scales. In some cases, data may differ to such an extent that "traditional" statistical techniques for dose–response analyses can not be applied.

It is essentially dependent on the definition of the problem whether there are temporal and spatial discrepancies between the monodisciplinary submodels. However, where it comes to data the picture is somewhat different. In many countries various *national* economic statistics are available. At the local level, on the contrary, ecological data may be more abundant. Spatial and temporal discrepancies and data problems together determine to a great extent whether a simple or a complicated mathematical structure has to be developed.

Another aspect of modeling which may differ widely between economic and ecological systems, both in type and extent is uncertainty. Especially in analyses of long-term developments and impacts, the degree of uncertainty is a major determinant of the outcome. Uncertainty may be due to:

(1) Stochastic properties of the system components.
(2) Lack of knowledge, i.e., about system state and processes.
(3) Problems of data measurement and interpretation.
(4) Lack of control on various input factors.
(5) Limited duration of operationality of control systems.
(6) Changing perspectives, moral standards, and values.

Uncertainty is considered a part of the spatial, temporal, and data aspects. Consequently, the discussion of uncertainty is included in those of the other aspects.

4.3.2. Temporal aspects

In integrating economic and ecological models, one needs to pay attention to at least three aspects of time:

(1) Turnover time.
(2) Temporal dynamics.
(3) Temporal development horizon.

Each biotic species has its own average lifetime, as has each type of capital asset. From these lifetimes one can derive the length of time it takes to replace all individual organisms of a population or assets in an particular system, counting from any point in time. This is called the ***turnover time***. It can be measured as the ratio of throughput to content (Odum, 1971a). For example, phytoplankton life is measured in days and human lives extend to an average of 70 years, while tropical rain forests and cities have turnover times of hundreds of years. Knowledge of the turnover times of system components is important for the modeler in case a dynamic model is developed for simulation. The time step of the simulation should fit the turnover time of the variable, otherwise forecasts that function as input to another variable may not reflect the real world flow per unit time, and cause unrealistic results.

Furthermore, turnover time indicates the period over which data concerning the development of the modeled system should ideally be available. In addition, such data would enable researchers to define accurately the ***temporal dynamics*** of the system, i.e., the shape of the curves indicating the development of its components. Regrettably, too often information on turnover time is not available and too often the temporal dynamics are derived from limited sets of data incidentally available for a specific time of measurement or randomly collected for that period.

Another aspect of time which may cause frictions in multidisciplinary modeling is the ***temporal development horizon***, which may differ between economic and ecological variables. An example of this is the acid rain case. Emission control measures may be implemented and effective in a few years. The development of damage in forest may however continue for decades, owing to irreversible changes in forest soils. Knowledge of these development horizons is necessary to adequately assess the total of effects of both negative and positive influence on natural systems. It should furthermore define the time period set for predictions.

4.3.3. Spatial aspects

Next to time, spatial dimensions need to be dealt with carefully in connecting economic and ecological variables. Again three aspects are distinguished:

(1) Spatial scale.
(2) Spatial dynamics.
(3) Spatial development horizon.

In analogy with turnover time each system component and the system itself have their own range of spatial dimensions, also called the ***spatial scale***. Knowledge about the spatial scale is important as it defines the area (or volume) over which data ideally should be available for an accurate analysis. The boundaries of the economic system considered in a resource problem analysis may not a priori be the same as those of the pertinent ecological system.

Especially if ***spatial dynamics*** are to be modeled (i.e., regular development patterns, random, or lumped development and dispersion over space) one needs to be careful with the spatial scale of the dynamic variables. Detailed information will enable the analyst to formulate realistic spatial development curves. Often, such information is not available for all the relevant systems components. This may cause serious credibility problems (especially in lumped parameter models).

Space may cause still another kind of problem in multidisciplinary modeling, namely, when the ***spatial development horizon*** differs between submodels. This refers to the area or volume over which impacts and developments extend, and the analysis of causal factors should extend, as well as to the area (or volume) over which the model can be validated. In analogy with temporal development the full extent of problem impacts may not be grasped if "a priori" chosen system boundaries did not match the spatial development horizon.

4.3.4. Data

The crucial role of data in modeling is well known to modelers and model users, and has already been indicated several times in this book. We distinguish three aspects of data which deserve special attention in economic-ecological modeling:

(1) Type of data.
(2) Level of aggregation.
(3) Quantity of relevant data.

The data available in multidisciplinary research may vary in ***type***: qualitative or quantitative data. Often the modeler has to work with a mix of these. If data types differ between submodels, conversions may be required which are often hard to test. In case qualitative or mixed data are the only data available, traditional statistical and econometrical techniques may not be sufficient.

Differences may also exist in the ***level of aggregation***, i.e., the data available may differ as to their temporal and spatial resolution or refer partly to individual (micro) behavior and partly to aggregate (macro) phenomena. Again one needs to be cautious with such data sets when used in determining dose–response relationships.

Further discrepancies may arise between desired and available ***quantities of data***. Statistical/econometrical parameter estimation techniques require usually a minimal number of data points to produce significant results. Differences between economic and ecological variables and submodels in type, level of aggregation, or quantity of the data cause problems in drawing inferences from the integrated modeling exercise.

4.3.5. Mathematical structure

We distinguish two kinds of problems with respect to the mathematical structure aspect of linkages between economics and ecology in models:

(1) In case two or more existing monodisciplinary models are to be connected, the problem consists of finding a mathematical format which resolves the format differences of the information which must be exchanged. For example, if an economic deterministic, differential equations model needs to be linked to an ecological probabilistic set of difference equations, a series of technical translation problems arise. Next to the technical linkage problems, there are the interpretation problems caused by differences in model specifications. For example, the predictions derived from an empirical model should be interpreted under the constraints defined by the theory of statistics and by the data set. This does not apply to the predictions derived from a theoretical model. Here, the extent to which the model predictions, obtained by simulation experiments, agree with observed data guide the interpretation. Combining these two model types may lead to erroneous interpretations.

(2) In case an integrated, holistic model is to be developed, the problem consists of finding adequate representations of the respective systems and combining them in a single, coherent, and operational format. In this case, we do not need to worry about differences in model specifications, but concentrate on the selection of a compromise format which is considered adequate enough for both system types. For example, expressing ecological processes in monetary units in order to match an economic cost variable is not considered adequate by most ecologists.

From the material researched it is not clear whether there are preferred mathematical structures among economic and ecological modelers, although the IvM–IIASA (*see* Appendix A) survey shows a slight preference for dynamic, nonlinear, deterministic approaches.

Basically, the problems are due to the general problem of lack of relevant data, with which translations of mathematical structures can be substantiated. Additional data acquisition is often required to connect submodels or integrate variables in a holistic equation. Quite often, however, this is not possible due to practical constraints. In those cases, quick and cheap technical solutions are to be employed. Some of these will be discussed in Section 4.5.

4.4. Integrated Economic–Ecological Models

4.4.1. Model structure

The definition of economic–ecological models can be made more meaningful by looking at existing economic–ecological models. The internal structure of a submodel can be defined, among others, by the relationships between the variables it contains. Only two types of relationships are distinguished here:

(1) Submodels consisting of a single variable or of a set of isolated variables (simple submodels).
(2) Submodels consisting of a fully, or partially related set of variables (complex submodels).

A single variable or a set of several unrelated variables is rarely referred to as a submodel. But, for reasons of contrast, we do so here. The complexity of the class of "complex" submodels may vary widely, due to the high level of aggregation in this typology (*see Figure 4.1*).

The integrative structure can be defined by the types of the relationships present between (variables of) the economic and ecological submodels. Three types can be distinguished:

(1) A one-way relationship in which the economic submodel drives the ecological submodels.
(2) A one-way relationship in which the ecological submodel drives the economic submodels.
(3) A two-way relationship, i.e., interdependent submodels.

In the first case the ecological submodel is linked to the economic submodel via one or several dependent ecological variables, the values of which are determined by independent (at least from ecological variables) economic variables. In the second case the situation is reversed. Interdependent submodels may be linked through sets of dependent variable–independent variable relationships and through mutually dependent variables (*see Figure 4.2*).

By combining these aspects of the model structure, a typology of economic–ecological models can be developed. *Figure 4.3* summarizes the resulting 12 model structure types. Of course, in large-scale analyses one may find combinations of several types of submodels and

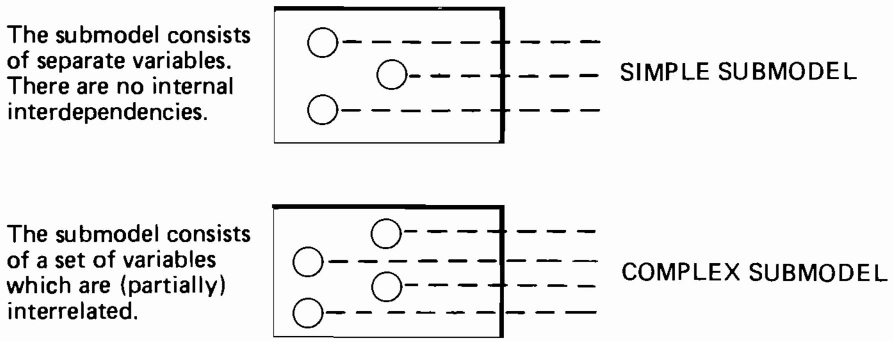

Figure 4.1. Internal relationships.

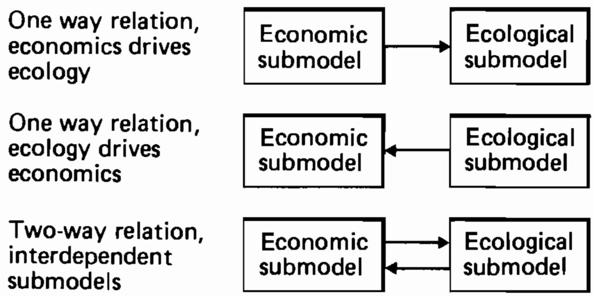

Figure 4.2. Relationships between submodels.

several types of economic–ecological relationships. Furthermore, submodels describing still other disciplinary systems may be included.

In the IvM–IIASA survey on economic–ecological models (*see* Appendix A) this theoretical classification was used. The class of simple models stands for highly aggregated, lumped parameter models, and two-variable regression models. Examples of these models can be found in Chapter 5, this volume, and in Charles (1982). One-on-one variable regressions produce the simple models of type 1 and 3.

4.4.3. Model characteristics

Completely dynamic models are predominant in the survey sample of economic–ecological models. We found about as many optimization (prescriptive) models as simulation (predictive) models and combinations of these two types. The ecological submodels are mostly built for simulation experiments, a majority of the economic ones use

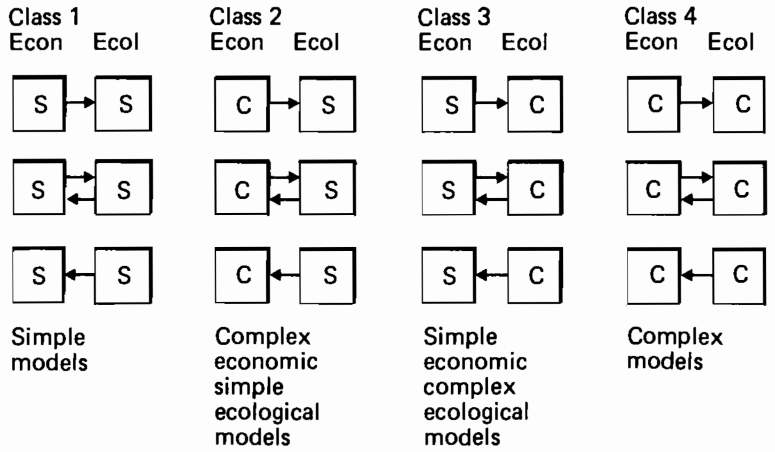

Figure 4.3. Classes of economic–ecological models.

optimization techniques (62.5%). As to spatial aspects, the survey results indicate that the local and regional scale of modeling is most common in this multidisciplinary field. A closer look at the submodels reveals that in about 65% of the models the geographical scales are the same, either local, regional, national, or global. In other models, the submodels may represent the system at more than one spatial level, but in almost all cases they have at least one level in common.

Examples of descriptive economic–ecological models are: a quantitative description of Gotland's economy and natural systems in energy units (Jansson and Zucchetto, 1978) and an energy analysis of Gotland's agriculture (Zucchetto and Jansson, 1979). Lavine and Butler (1982) describe a regression model of inputs to the economic process, including natural energies, and economic output. Costanza and Neill (1981) developed an input–output model for the biosphere, used to estimate direct and indirect solar energy costs of various commodities. Pearce (1976) presents a static model in which ecological impacts are integrated into the standard economic model. All these models do explicitly include economic and ecological variables, sometimes all expressed in a single metric.

Predictive models are most numerous in the survey. Most of the models are regional scale simulation models dealing with land use, fisheries, outdoor recreation, and water management (Arntzen *et al.*, 1981; Boynton *et al.*, 1977; Christianson, 1982; Limburg, 1982; Steinitz *et al.*, 1976; Tai, 1979). Brown (1977) has modeled Vietnam as an economic–ecological system, and predicted the extent of disordering effects of war on cities and ecosystems. All these models do contain a somewhat balanced description of both the economic and ecological system, some quite complex, some very aggregated (e.g., Christianson, 1982).

We found the following *prescriptive* economic–ecological models. Zucchetto *et al.* (1980) developed an input–output model of Gotland, used for optimization of economic and ecological resources in a rural design. D'Arge and Kogiku (1973) developed an optimal control model in which consumption and waste generation can be regulated. Bogardi *et al.* (1982) use a multiobjective compromise programming technique to solve a multipurpose regional aquifer management problem (*see* Chapter 9, this volume). And finally, Charles (1983a,b) uses a dynamic programming approach for optimal investment planning in fisheries. These optimization models all include both economic and ecological aspects, either as objective functions or both.

Lonergan (1981) *combines* simulation and optimization techniques. Duckstein *et al.* (1982) combine a predictive model with an evaluation technique for a nutrient loading control problem. All other models listed combine predictive or descriptive techniques with prescriptive (optimization) techniques. Clark *et al.* (1979), Clark and Kirkwood (1979), Clark and Lamberson (1982) and Silvert (1977) use these models for fisheries problems, Bennett and Bowden (1976) for an agricultural problem, Bresser (1981), Nachtnebel *et al.* (1982) and Bordet (1981) for water quality and water resource problems. Both Ikeda (1980) and Spofford *et al.* (1976) combine an ecological simulation submodel with an economic optimization (programming) submodel.

4.5. A Procedure for Economic–Ecological Model Design

4.5.1. Introduction

The essential problem for policy analysts and academic modelers involved in policy oriented modeling studies is: what kind of model is most appropriate and effective for the policy issue at hand. In this section we introduce a procedure for model design, which summarizes the information on the various types of models used in environmental and resource management, as presented in Chapters 1 through 4. The procedure directs the search in very general terms only. We distinguish two levels in the process of designing or, if models are available, selecting the most appropriate model. The first level is aimed at determining the general content and operational technique of the model. The content is derived from the characteristics of the policy issue, the general operational technique from the intended use of the model, which depends on the phase in the policymaking process. The second level deals with the technical aspects of multidisciplinary modeling.

4.5.2. Level 1: Model content and operational technique

The *first step* in any model design or selection process in a policy or management context should be the analysis of the policy issue, to ascertain that the model portrays the essential characteristics of the issue. *Figure 4.4* indicates the model types that are considered adequate models for certain types of policy issues. For example, issues referring to ecological impacts of resource use cannot be modeled adequately by models in which the ecological submodels are independent, driving submodels. The straightforward choice would be a model consisting of complex ecological and relatively simple economic, driving submodels. If feedbacks are relevant because one suspects that the impacts have an influence on the causing factor, then the two-way type in model class 2 is an adequate choice. If a simple indicator of ecological impacts is preferred because the concern is more on the causal side, i.e., the economic subsystem, or if a simple ecological indicator is the only possibility datawise, environmental–economic models are appropriate structures. Finally, in case the policy objective regards both the impacted system and the causing system, complex economic–ecological models, with or without feedbacks are the logical choice. In this way *Figure 4.4* provides an overview of types of model structures which are considered "adequate" and "inadequate" for different types of policy issues. All the empty cells represent model structures which are not the most appropriate ones for the policy issues at hand.

The *second step* is included to determine the operational technique which fits the intended use of the model in the policymaking process. *Figure 1.5* summarizes this second step. Only a general indication of the model technique is obtained in this way. For example, from this figure one can see that some optimization routine (a prescriptive model) is to be employed for the plan design phase (= devise the best solution to a problem). One cannot conclude, however, whether it needs to be a linear, nonlinear, or a dynamic programming model. This is rather a matter of how adequate a model is required, than of intended use.

4.5.3. Level 2: Technical integration of economics and ecology

Basically, one can integrate mathematical models in two ways:

(1) *"Interpretative" integration.* A user interprets the output of one model and translates it mentally and manually into input data for the other model. We have not found any analyzable description

	Environmental economics and resource economics models (chapter 2)			Environmental Biology and resource ecology models (chapter 3)			Integrated economic-ecological models (chapter 4)		
	EN → EL	EN ⇆ EL	EN ← EL	EN → EL	EN ⇆ EL	EN ← EL	EN → EL	EN ⇆ EL	EN ← EL
Ecological policy issues (*Fig. 1.2*)	IM – IN	IM – IN$_+$	–	IM	IM$_+$	–	IM	IM$_+$	–
Economic - policy issues (*Fig. 1.3*)	–	IM$_+$	IM	–	IM – IN$_+$	IM – IN	–	IM$_+$	IM
Economic - ecological policy issues (*Fig. 1.4*)	–	INT–IM–IN$_+$	–	–	INT–IM–IN$_+$	–	–	Fu–INT$_+$	–

EN → EL = Economics drives ecology
EN ← EL = ecology drives Economics
EN ⇆ EL = dual relation.

IM = impact model.
IN = indicator model.
IM – IN = impact–indicator model.

INT – IM – IN = integrated impact and indicator model.
FU – INT = fully integrated model.
\+ = feedbacks are introduced.
– = models with this type of relations between economics and ecology are not likely to be available for the policy issues in this cell.

Figure 4.4. Likely model structures for economic and ecological policy

of this process and therefore refrain from further discussion of this way of integrated modeling.

(2) *"Mechanical", "formal" or "mathematical" integration.* In quantitative models, relationships take the form of a regression equation, a time-indifferent coefficient, a time-dependent mathematical function, or a threshold value (which leads to conditional responses of the effect variables).

As mentioned before, we distinguish two forms of integrated economic–ecological modeling:

(1) Integration of two or more monodisciplinary submodels.
(2) Integrated, holistic modeling of an economic–ecological system.

In the first case the modeler has different models as starting material and has to assess the differences before deciding how to connect them. In the second case the modeler only has a problem description as starting material, for which he has to define the variables which must then be integrated in a model.

In practice, the difference between connecting submodels or variables (as in a holistic model) is marginal. Integration of submodels implies defining and quantifying relationships between variables in different submodels. *Empirical models* involve relationships for which the quantitative specification is derived by means of statistical/econometric analysis. The most common method is some form of regression analysis, either by maximum likelihood or least squares estimation (Dobson, 1983). If the data set and the variables do not fit the requirements of the statistical estimation techniques, *theoretical models* offer a possibility to obtain quantified relationships. By comparison of model simulation results with observed data, parameter values can be estimated. In both cases the model needs to be tested against data not used for parameter calibration.

Before integrating variables from different submodels in a single equation, the modeler must assess the differences in various model aspects (*see* Section 4.3). If there are differences in *temporal* or *spatial* aspects, then the most common solutions are aggregation and disaggregation. Both techniques are regularly used to facilitate multidisciplinary connections. For example, initial variable and parameter values are recalculated for a level at which connecting is relatively easy. The actual connection is sometimes done by means of model specifications in which the independent variables have a dummy character. Although this connection technique is quite simple and straightforward, problems still exist:

(1) It may be difficult to get information about the distribution of multidisciplinary relations at various levels of an integrated system.
(2) Dummy variables neglect part of the information they represent.

If the *mathematical structures* differ, one can sometimes adapt one submodel to fit the other (e.g., linearize equations, turn stochastic equations into deterministic ones and dynamize static submodels), or reformulate both submodels (partly) to establish the basis for connections and realize an operational model. Some creativity and a lot of mathematical proficiency is required to complete such operations successfully.

All the transformations and adaptations will only lead to adequate models if the new, integrated model can be tested against a sufficient data set. If such a data set is not available the model may only function as a conceptual tool, which integrates knowledge and hypotheses, and which calls for testing.

Generally the *data* on which the different submodels are based differ in type and quantity. Not always is quantitative information available. Several techniques have been developed to deal with these situations (qualitative and incomplete data sets), for example, path models (Jorskog, 1977), scaling analysis (Nijkamp, 1979), graph theory (Ponsard, 1966), and disaggregate choice analysis (McFadden, 1984). Disaggregate choice analysis can be used directly, i.e., independently. The expected results need however have a discrete character. Other soft data methods which are used independently, often suffer from interpretation problems, like many scaling methods, or they can only be applied if all data are translated to the level of the data with the least quantitative character. This results in loss of information. Most of the methods can be used more satisfactorily when they are applied to derive quantitative indicators as improved alternatives of using dummies. Scaling methods are good in this respect.

If, however, straightforward observation, experimentation, and data analysis do not provide a basis for an empirically based relationship, technical manipulation of variables, data, and equations may be required to quantify the relationship. Two examples are given.

Multidisciplinary relationships are regularly modeled by "intermediars", i.e., intermediate variables or even complete mathematical models. They may function as a technical bridge between two variables for which a direct relationship can not be established, or provide an aggregation/disaggregation mechanism. An example of the first is the case where the output of an economic model is measured in a pollution index, combining several pollutants, for which a relationship has been established (e.g., by simple regression analysis) with some ecological variable(s). The output of an economic model is also sometimes used as input for an intermediar pollution model which then defines the input for an ecological model. For example, a specific spatial distribution model may function as a third component in an integrated economic–ecological approach.

The second example regards conversion of dimensions of the variables. The relationship between two variables of different dimensions may require a simple constant conversion coefficient, a linear or a complex non-linear equation. Many economic studies have attempted to valuate ecological aspects in terms of money, and in several ecological approaches economic aspects have been modeled in terms of energy, mass flow etc. In both types of conversions specific characteristics, and sometimes essential ones, are lost.

4.6. Concluding Remarks

Economic–ecological models differ from the 'extended' monodisciplinary models developed in such disciplines as environmental and resource economics and ecology, in that they include adequate representations of both the economic and the ecological components and processes involved in the problem they are developed for.

There are definite possibilities to integrate existing economic and ecological models of environmental and resource problems. The most successful attempts have been in connecting so-called physical–economics models [e.g., material-balance models (Kneese *et al.*, 1969)] with ecological "flow" models, designed to assess resource extraction and pollutant input impacts (*see* Ikeda, 1980; Spofford *et al.*, 1976; Arntzen *et al.*, 1981).

Integration of economics and ecology in a holistic model has been shown to be entirely feasible. Both empirical models (Isard, 1972) and theoretically derived models (e.g., Brown, 1977; Boynton *et al.*, 1977) have been applied in resource and environmental analysis. The extended economic cost–benefit models and the so-called ecological–evaluation models do not provide good material for integrated economic–ecological modeling.

Although the integration of economics and ecology in mathematical models proves to be feasible, it is still an operation which generally requires a series of technical adaptations of the models or additional data manipulations with which the integrated model is to be developed. Temporal scales have to be synchronized, spatial scales must be matched through (dis)aggregation, and sometimes whole sets of equations need to be reformulated to make the integrated model run.

Uncertainty pervades all aspects of modeling, and becomes a major obstacle in those exercises which concern long-term projections and impact analysis. If data are available in the right form and quantity, the technical adaptations are not only easier to implement, but then the consequences of integration on the modeling results (predictive or prescriptive output) may even be tested. This, however, appears to be an everlasting modeler's dream. Explicit studies which evaluate integrated modeling in this respect have not been found among the studies surveyed.

The selection of existing economic–ecological models discussed indicates, and not more than that, which types of technical structure and characteristics are to be expected in these models. Furthermore, economic–ecological models turn out to be used for all types of environmental and resource policy issues, even when the objective in policy-making or management would warrant extended monodisciplinary models only.

Much has been presented in the first four chapters about models for environmental and resource management. Most of it has been theoretical, sometimes adorned with empirical examples and references to experience in the modeling world. In the next nine chapters the practical aspects of environmental and resource modeling will be presented. It is there, that actual models, applied in the real world, are to be found.

PART II

The Practice of Environmental and Resource Modeling

Introduction to Part II

L.C. Braat

W.F.J. van Lierop

Part II contains nine chapters, each of them presenting descriptions and comparative evaluations of models and model applications in a particular field. The authors have selected a number of models or modeling approaches which they consider to be more-or-less representative of the variety present in the fields discussed. The models are described in three ways. A brief verbal description of the model structure precedes the more formal representation by means of diagrams. The diagramming style differs between the chapters. For those readers who prefer mathematical formulations, most of the models have been described in such a format. The evaluation of the effectiveness of the models in policy analysis and environmental and resource management is usually introduced with a short overview of the results. Evaluation criteria differ with modeling purpose, field of application, and type of problem.

Colin Clark, in Chapter 5, opens the series with a review of fisheries models. In Chapter 6, Goran Ågren discusses the models used in forestry. A selection of models used in agriculture is presented by Zsolt Harnos in Chapter 7.

Istvan Bogardi introduces the reader into the field of water resources modeling in Chapter 8. In this chapter the quantity of water (of various qualities) is the focus. The subject matter of Pete Loucks' Chapter 9, on the contrary, is the quality aspect of water bodies. Models for outdoor recreation are discussed by Floris van der Ploeg in Chapter 10.

In Chapter 11, Ferenc Tóth describes and evaluates a number of models which have only one thing in common; they address multiple resource use. Saburo Ikeda provides the reader with a sample of models for regional systems policy in Chapter 12.

The final chapter of Part II, Chapter 13, written by Howard Odum, addresses national, international, and global systems models. Obviously, such a wide range of systems cannot be dealt with fully in a few pages; therefore this chapter has a structure that differs from the other eight chapters.

In terms of the relationships between socioeconomic and ecological systems as depicted in *Figure 1.1*, Chapters 5, 6, 7, 8, and 11 primarily address the resource flows from natural environments to socio-economic processes. The pollution flows and disturbance of human origin to the environment are addressed in Chapters 9 and 10, respectively. Resource flows are often at stake in outdoor recreation management (Chapter 10) though. Chapters 12 and 13 take a different approach. They address the total system of environment and economy, and their relationships, at different spatial scales.

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CHAPTER 5

Fisheries as Renewable Resources

C.W. Clark

5.1. Introduction

Biological modeling has long been an indispensable ingredient of commercial fisheries management. The earliest models, developed in the 1950s (Beverton and Holt, 1957; Ricker, 1954; Schaefer, 1954) were designed for the purpose of estimating the *maximum sustainable yield* (MSY) for fisheries based on individual species and stocks of fish. Thus MSY was explicitly recognized as the fundamental policy objective in these models, which still provide the basis for operational management of most commercial fisheries.

Economic fishery models, also developed during the 1950s (Gordon, 1954) were concerned with the objective of *maximizing net economic yield*, and with identifying policy instruments that could be used to achieve this objective. It was firmly demonstrated, on both theoretical and empirical grounds, that the unregulated "open-access" (or common-property) fishery would tend toward a *bionomic equilibrium* in which net economic yield (or "rent") was entirely dissipated. For highly priced species, this bionomic equilibrium would involve severe depletion of the fish stock, with physical yields far below MSY. Thus the "overfishing problem" was seen as a combination of biological and economic phenomena; it had been discussed by many earlier writers and was rediscovered by Hardin (1968), who saw it as an instance of the "tragedy of the commons".

Relying on an implicitly static economic model, the early economic studies suggested that individual property rights, in the form of private ownership of the entire fishery, would automatically ensure both resource conservation and economic efficiency (Scott, 1955; Crutchfield and Zellner, 1962). Alternatively, the state could ensure conservation and efficiency by assuming ownership and charging fishermen an appropriate user fee or royalty. Neither of these policy recommendations has proved to be feasible in practice.

The belief that private resource owners would rationally conserve their resource base conflicts with the often observed behavior of farmers, lumbermen, and other owners of resource stocks. The central role of *time discounting* in resource conservation was clearly established by the agricultural economist, S.V. Ciriacy-Wantrup (1968). A "bioeconomic" model developed by Clark (1973, 1976) summarized the issue in mathematically simple and rigorous form. Private owners of renewable resource stocks will opt to deplete those resources if revenues cover costs, and if the private discount rate exceeds the biological growth rate of the resource. Government intervention or assistance (e.g., in the form of low-interest loans) may thus be necessary to ensure resource conservation, even among private owners. Government intervention is essential in the case of common-property resources.

A bioeconomic model of the Antarctic whaling industry (*see* Section 5.1) indicates that the current levels of depletion in this now inactive fishery are roughly compatible with either paradigm, private owner profit maximization, or common-property bionomic equilibrium. The model emphasizes the difficulty of managing international resources in a conservation mode.

As noted above, past fishery management policy has been largely based on the singleminded objective of achieving MSY. This has traditionally been accomplished by means of Total Allowable Catch quotas (TACs), or by means of regulated fishery openings and closures, gear restrictions, etc. Although such methods have often proved successful in maintaining high yields and preventing overexploitation (with notable exceptions among the pelagic species such as herrings and anchoveta), the economic consequences have been extremely disappointing. Actual bankruptcy of fishing enterprises operating on fully protected stocks have become commonplace, in fact almost universal, since the advent of national jurisdiction within 200-mile coastal zones. Quite spectacular government subsidies have been employed, but have not always succeeded in maintaining economic viability of the fishing industry. Interest in achieving economic as well as purely biological objectives has thus increased.

It is not necessarily obvious, however, which types of regulatory instruments are both feasible and potentially successful in achieving the desired economic objectives. In Section 5.2 we therefore describe a general model of fishery regulation designed to address this issue. The model is deterministic, and is restricted to the case of a single-species fishery. The main result is a demonstration that, under these conditions, *allocated transferable vessel quotas* are equivalent to royalties (or sole ownership) in terms of economic efficiency. This result, long known in welfare economics, may be of considerable practical importance in fisheries management. Allocated quota systems are currently being introduced into several commercial fisheries in New Zealand, Iceland, Canada, and elsewhere.

The limitations of deterministic fishery models have been addressed by numerous authors (e.g., Beddington and May, 1977; Reed, 1979; Ludwig and Walters, 1982). A stochastic model due to Reed is described in Section 5.3. Reed (1979) establishes the optimality of constant-escapement harvest policies for a stochastic single-species, non-age-structured fishery model with known parameters. Ludwig and Walters (1982), on the other hand, show that a "probing", or experimental management strategy may be optimal in cases where parameter estimates involve high uncertainty.

For lack of space, we will not discuss multispecies or ecosystem fishery models in this chapter. Such models are not presently utilized to any extent in fisheries management (even though many fisheries are in fact multispecies in nature), primarily because the lack of sufficient data has precluded the validation and testing of complex fishery models. There is also a lack of agreement as to appropriate objectives for multispecies fishery management, MSY being essentially meaningless in this context (May *et al.*, 1979).

5.2. Antarctic Whaling

5.2.1. Introduction

The depletion of stocks of large baleen whales (particularly blue, fin, and humpback whales) in the Antarctic is well known. The reluctance of active whaling nations to agree on conservation measures under the auspices of the International Whaling Commission (IWC) has been variously attributed to stupidity, greed, and institutional failure. The model described in this section attempts to depict the exploitation policy that would have been followed by a sole owner of the resource, assuming normal business practice, i.e., maximization of the discounted present value of net long-term yield. The model incorporates both variable and fixed (capital) costs of whaling, the latter being especially significant for this industry.

During its heyday, Antarctic whaling was among the most profitable of marine resource industries, with total profits of billions of dollars. Current revenues are infinitesimal by comparison, with all species except the small Minke whale presently under an IWC moratorium. *Potential* profitability remains high, however, due to the demand for whale meat in Japan, Norway, and the Soviet Union. The present moratorium may thus be fragile. The institutional problems of whale conservation in international waters cannot be considered to have yet been resolved.

5.2.2. Model structure

The Antarctic whaling model (*Figure 5.1.*) contains two state variables, X_t = whale stock (Blue Whale Units) at time t, and K_t = capital stock in terms of fleet capacity (Factory Fleet Units). The whale stock is increased via recruitment, and decreased by harvesting:

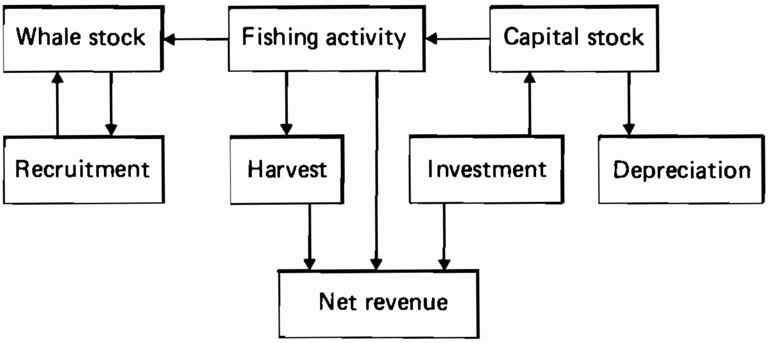

Figure 5.1. Structure of the whale model.

$$\mathrm{d}\frac{X_t}{\mathrm{d}t} = G(X_t) - H_t \tag{5.1}$$

The natural growth function $G(X_t)$ is specified in logistic form:

$$G(X_t) = rX_t(1 - X_t/\bar{X}) \qquad (r, \bar{X} = \text{constants}) \tag{5.2}$$

The harvest rate H_t is related to X_t and to effort E_t (Factory Fleet Units) by the usual formula:

$$H_t = qX_tE_t \qquad (q = \text{constant}) \tag{5.3}$$

Effort is constrained by fleet capacity K_t:

$$0 \leq E_t \leq K_t \tag{5.4}$$

Fleet capacity depreciates at rate γ, and may be increased by investment at rate I_t:

$$\frac{\mathrm{d}K_t}{\mathrm{d}t} = I_t - \gamma K_t \qquad (\gamma = \text{constant}) \tag{5.5}$$

Investment is assumed to be irreversible:

$$0 \leq I_t \tag{5.6}$$

The sole owner schedules investment I_t and effort E_t so as to maximize the present value of net revenues:

$$\underset{E_t, I_t}{\text{maximize}} \int_0^\infty e^{-\delta t} (pH_t - c_1 E_t - c_2 I_t)\, dt \tag{5.7}$$

where δ = discount rate, p = price per BWU, c_1 = variable cost of effort, and c_2 = cost of investment.

5.2.3. Evaluation

A rigorous analytic solution is given by Clark *et al.* (1979), who demonstrate that:

(1) The optimal policy ultimately reaches a long-term equilibrium ("optimum sustained yield"), with all variables X_t, K_t, E_t, I_t constant.

(2) If the initial stock X_0 exceeds the long-term equilibrium, then the fishery proceeds through an initial cycle of (apparent) overcapacity, overexploitation, depreciation, and rehabilitation, before eventually reaching the equilibrium (*Figure 5.2*).

The parameter values shown in *Table 5.1* were estimated from a variety of sources, and pertain to the Antarctic whaling industry *ca.* 1980 (Clark and Lamberson, 1982). The trajectories shown in *Figure 5.2* pertain to these values and to an initially unexploited stock, $X_0 = \bar{X}$.

Sensitivity to the parameter values is analyzed in the reference (see *also* Charles, 1983a). Because of the low biotic potential of whale stocks (r = 0.05), the discount rate δ plays a dominant role in determining the optimal long-run equilibrium, X^* = 115|000 BWU (i.e., 29% of the unexploited stock level). The current Antarctic whale biomass is thought to be about 90|000 BWU; the present IWC rules would maintain a complete moratorium until X = 160|000 BWU, and would reach a long-run equilibrium at about 240|000 BWU. (However, the IWC rules require separate management of each breeding stock, a detail not included in our model.)

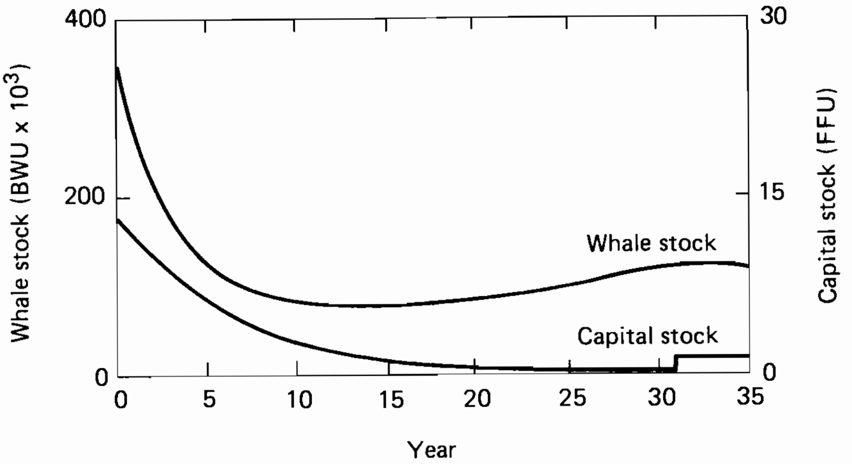

Figure 5.2. Antarctic whaling: optimal dynamics.

Table 5.1. Parameter values for the whaling model.

Parameter	*Interpretation*	*Estimated value*
r	Intrinsic growth rate	0.05 per annum
$\bar{X}$	Carrying capacity	400 000 BWU
q	Catchability coefficient	0.026 per FFU year
γ	Depreciation rate	0.15 per annum
p	Price of whales	\$7 000 per BWU
c_1	Variable cost of effort	\$1.0 $\times 10^7$ per FFU year
c_2	Fixed cost of capacity	\$2.0 $\times 10^7$ per FFU
δ	Discount rate	0.10 per annum

The main conclusion derived from this model is that, at least in the case of whales, "normal" profit motives and conservation motives may be in sharp conflict, owing to the discounting of future values. This phenomenon is most severe for slowly growing resource stocks, but appears to be all but omnipresent in resource economics. The need for government policies designed to reduce the anticonservationist policies of both private and public resource users thus becomes apparent – a theme that has long characterized the conservationist movement.

5.3. Fishery Regulation

5.3.1. Introduction

The vast majority of the world's marine fishery resources are exploited competitively. Fish stocks in international waters, and transboundary stocks, are exploited by fishermen subject to separate national juristiction, if any. Control of depletion can be difficult if not impossible in such circumstances. Doubtlessly much of the impetus for 200-mile economic zones stemmed from this fact.

Fish stocks within national boundaries are obviously more amenable to protection. However, well-protected fish stocks can still support an impoverished fishing industry – and will inevitably do so unless specific measures are taken to prevent this outcome. In this section we describe briefly a model of fishery exploitation under regulation. The model provides qualitative predictions of the biological and economic effectiveness of various types of management policy, including total catch quotas, license limitation, fiscal policy (taxes, fees, royalties), and quasi-property rights (allocated quotas). The model is more fully described by Clark (1980).

5.3.2. Model structure

The general structure of a regulated fishery is illustrated in *Figure 5.3*. Fishermen exploit the stock under normal economic motives, which are influenced by the set of regulations, which we assume rigorously enforced. Regulations may change over time, according to the biological and economic situation.

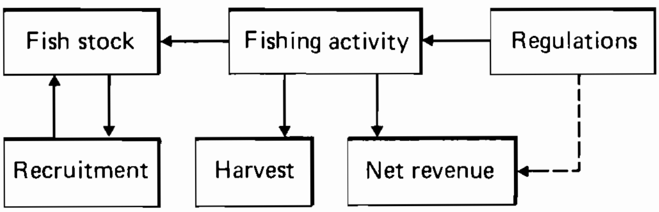

Figure 5.3. Structure of the fishery regulation model.

Mathematically, stock dynamics are again modeled by the equation:

$$\frac{dX_t}{dt} = G(X_t) - H_t \tag{5.8}$$

where now

$$H_t = \sum_{i=1}^{N_t} qE_{it}X_t \tag{5.9}$$

with N_t = number of fishing units (e.g., vessels) operating at time t and E_{it} = standardized effort of the ith unit. Net revenue flow per unit is given by:

$$\pi_{it} = pqE_{it}X_t - c_i(E_{it}) \tag{5.10}$$

where the effort–cost functions $c_i(E_{it})$ may differ, reflecting differences in vessel efficiency. The cost functions are assumed fixed for all time, and the c_i are concave, $c_i'' \geq 0$. For simplicity fixed costs are ignored (this is a serious limitation).

In the absence of regulation the competing fisherman (fishing unit owner) determines his effort level E_{it} so as to maximize π_{it}. Thus:

$$c_i'(E_{it}) = pqX_t \tag{5.11}$$

provided this yields $\pi_{it} > 0$; otherwise $E_{it} = 0$.

Equations (5.8)–(5.11) determine a unique solution X_t, E_{it}, N_t. Since total effort $E_T = \sum_1^{N_t} E_{it}$ is an increasing function of X, this solution converges to a uniquely determined equilibrium $\bar{X}, \bar{E}_i, \bar{N}$. [This holds if, for example, $G(X)/X$ is decreasing.] Net economic yield $\bar{\pi}_T = \sum \bar{\pi}_i$ will generally be *positive* at this equilibrium, but suboptimal (*see* below). This is *not* the Gordon bionomic equilibrium, which assumes open access, i.e., no restriction on vessel numbers for each cost category.

The first prediction of the model, then, is that the dissipation of economic rent characterized by bionomic equilibrium could be *partially* relieved by means of license limitation. But the model itself suggests that this prediction is suspect, since it hinges on the assumption of unchangeable cost functions c_i. In practice, vessel owners may be able to improve efficiency (i.e., to decrease variable cost) through capital improvements. There will be a tendency to do so, as long as the potential increase in individual yield π_i exceeds capital cost (amortized). Consequently the trend toward bionomic dissipation persists under license limitation. This trend has frequently been observed, and is referred to as "capital stuffing". It can take many perverse forms.

A socially optimal exploitation policy would maximize net economic input derived from the resource. If we ignore any divergence between private and social costs, this objective can be expressed as follows:

$$\text{maximize} \int_0^{\infty} e^{-\delta t} \left(\sum_1^{N_t} \pi_{it}\right) \mathrm{d}t \tag{5.12}$$

subject to the model equations (5.8)–(5.10). Standard optimization theory leads to the necessary conditions:

$$c_i'(E_{it}) = (p - \mu_t)X_t \tag{5.13}$$

where μ_t is a "shadow price" associated with the stock variable X_t, satisfying the adjoint differential equation:

$$\frac{\mathrm{d}\mu_t}{\mathrm{d}t} = [\delta - G'(X_t)]\mu_t - (p - \mu_t)q\sum E_{it} \tag{5.14}$$

The new system of equations again has an equilibrium solution X^*, E_i^*, N^*, μ^*, and it can be shown that:

$$\mu^* > 0, X^* > \bar{X}, N^* < \bar{N}, E_i^* < \bar{E}_i \, (i \le \bar{N}), \pi_T^* > \bar{\pi}_T \tag{5.15}$$

Optimal sustained yield thus involves a higher fish biomass X, a smaller active fleet N, lower effort levels E_i, and greater economic yield π_T, than the competitive equilibrium.

By comparison of equations (5.11) and (5.13) we see that the competitive fishery can in theory (i.e., under the given assumptions) be forced into the optimal mode by means of a single *tax on catch* (royalty) equal to the shadow price μ_t. This well-known principle of welfare economics is often invoked in policy recommendations for fisheries and other environmental resources. The distributional implication of such taxes is, of course, that economic rent then accrues to the state rather than to the fishing industry.

In welfare economics, quotas play a dual role to taxes. To model this, assume that each fisherman has a personal quota Q_{it}:

$$H_{it} = qE_{it}X_t \leq Q_{it} \tag{5.16}$$

Assume quotas to be freely transferable in any portion, and suppose that a perfect market develops for quota transfers. Let $m = m_t$ denote the clearing price on this market. The equation:

$$\frac{\partial \pi_i}{\partial H_i} \Big|_{H_i = Q_i} = m \tag{5.17}$$

then determines the ith fisherman's demand function $H_i = D_i(X, m)$ for quota units (reason: if $m < \partial \pi_i / \partial H_i$ the fisherman can increase his net income by purchasing an additional quota unit, and vice versa).

The market clearing conditions are therefore:

$$H_i = D_i = Q_i \tag{5.18}$$

$$c_i'(E_i) = (p - m)qX \tag{5.19}$$

Equation (5.19) follows immediately from (5.17). The total quota demand is:

$$D_i(X, m) = \sum Q_i = \bar{Q} \tag{5.20}$$

where $\bar{Q}$ denotes the total quota. The market price m depends inversely on $\bar{Q}$. If $\bar{Q}$ is set optimally, the quota price will be equal to the shadow price μ, and hence also to the optimizing catch tax.

This establishes the equivalence, *in terms of economic efficiency and incentives*, of taxes and allocated transferable quotas – under the model assumptions. The *distributional* implications of the two approaches, however, are opposite, since economic rents accrue to (original) quota holders. By combining quotas and taxes, the management authority can achieve any desired partitioning of economic rent between the industry and the state.

5.3.3. Evaluation

The model is very restrictive in its assumptions. Some of its limitations are discussed in the original reference (Clark 1980). Stochastic effects are discussed by Andersen (1982) and by Clark (1985a); the equivalence of taxes and quotas breaks down for stochastic models.

In spite of its limitations, the model does clearly indicate the likely effects of various management methods. License limitation will lead to "capital stuffing", even if combined with total (unallocated) catch quotas (which are analyzed in the reference). Taxes or allocated quotas both have the potential for inducing efficient fish harvesting. Explicit application to a given fishery obviously requires careful attention to the specific details; the need for strict enforcement of the tax or quota system is paramount.

5.4. Stochastic Models

5.4.1. Introduction

Deterministic fishery models are unrealistic in two related senses. First, fish stocks undergo random fluctuations over both space and time. Secondly, fishery systems involve significant levels of uncertainty. Stochastic models and decision–theoretic methods can be used to study the implications of these phenomena for fishery management.

5.4.2. Model structure

We consider the following stochastic stock-recruitment model (Reed, 1979):

$$X_{k+1} = Z_k G(S_k) \tag{5.21}$$

$$S_k = X_k - H_k \tag{5.22}$$

where X_k denotes recruitment biomass in year k, S_k denotes escapement following harvest H_k $(0 \le H_k \le S_k)$ and where the $Z_k \ge 0$ are i.i.d. random variables with mean $\bar{Z}_k = 1$ and probability density $f(z)$. This model is a stochastic, discrete-time analog of equations (5.1) and (5.8). The corresponding optimization objective is:

$$\underset{H_k}{\text{maximize}}\ E\left\{\sum_{k=0}^{\infty} \alpha^k \pi(X_k, H_k)\right\} \tag{5.23}$$

where $\alpha \varepsilon (0,1)$ is the discount factor, $\pi(X_k, H_k)$ is net economic revenue in year k, and the expectation is taken over the random sequence $\{Z_k\}$.

Under appropriate assumptions, Reed (1979) establishes that the optimal harvest policy is a constant-target escapement policy of the form:

$$H_k = \max(0, X_k - S^*) \tag{5.24}$$

If

$$\pi(X, H) = \int_{X-H}^{X} Q(x)\mathrm{d}x \tag{5.25}$$

then (under an additional condition on S^*) S^* is characterized by the condition (*see also* Sobel, 1982):

$$S^* \text{ maximizes } \alpha E_z \{Q[zG(S)]\} - Q(S) \tag{5.26}$$

Constant-escapement catch policies are in fact often employed in fisheries management (e.g., the Pacific salmon fisheries). Their use requires "on-line" monitoring of the fishery in order to ensure that the escapement is actually achieved. Such monitoring is expensive, and is thus only practical for valuable species caught in relatively confined areas.

For most fisheries estimates of stock abundance are subject to significant uncertainty. Annual catch quotas, usually determined on the basis of 'best estimates' of current stock levels and productivity, are thus also subject to error. Overestimated catch quotas can lead to overfishing, whereas underestimates cause unnecessary losses to the industry.

In order to model stock uncertainty, Clark and Kirkwood (unpublished) modified Reed's model by assuming a Bayesian prior distribution for annual recruitment X_k. Under this assumption, constant-target-escapement policies are no longer optimal. The effect of stock uncertainty on catch quotas has been investigated; for moderate levels of uncertainty (coefficient of variation less than 50%, say), optimal quotas are reduced, but for high uncertainty quotas may in fact be increased relative to the certainty-equivalent model.

5.4.3. Evaluation

The treatment of uncertainty in economics is a difficult and challenging problem. Although uncertainty is recognized as a major factor in commercial fisheries, present management practice tends to be restricted to deterministic paradigms. It appears likely that several fishery collapses over the past two or three decades might have been prevented by means of a more careful consideration of the influence of random fluctuations and uncertainties. The same applies to fisheries that become seriously depressed when anticipated markets or price levels fail to materialize. A bias toward optimism pertaining to both biological and economic prospects has been an observable characteristic in many fisheries. The use of a decision-theoretic approach to fisheries management seems overdue, but will require the development of appropriate new models.

Further discussion of stochastic fishery modeling appears in the papers of Beddington and May (1977), Ludwig (1979), Mendelssohn (1980), Andersen (1982), Ludwig and Walters (1982), Charles (1983b), and Clark (1985a,b).

5.5. Overall Evaluation

The general objective of the type of modeling described in this chapter relates to policy aspects of fishery management. The tradition in fisheries has long been one of ignoring economic implications and concentrating on biological yield. This tradition is not necessarily unreasonable, inasmuch as the introduction of economics inevitably entails distributional and hence also political questions. It would be very convenient if the scientific and social problems of fishery management could be treated separately. Unfortunately this has not proved to be feasible. An ecological-economic theory of fishery management therefore seems essential.

The models described in this chapter are of a general rather than a specific nature. Their purpose is to provide insights and qualitative predictions, rather than to generate quantitative estimates of quotas and the like. For example, the result (Section 5.3) that allocated quotas, variously called quantitative rights, quasi-property rights, enterprise allocations, etc., provide some realistic hope for resolving the common-property dilemma, is nowadays being taken seriously by management authorities. In fact, biological and social aspects can be fairly well separated using this approach, since the total quotas can be determined largely on biological grounds. (Exceptions may occur if market opportunities are limited, and for multispecies fisheries.)

It is by no means true (as has sometimes been asserted) that practical implementation of such models, or theories, requires the development and validation of a specific model for each and every application. Specific biological models *are* needed, in order to determine total catch quotas, and some estimate of optimal fleet size is also required. A transferable allocated quota system will then allow the fishery to rationalize its own operations without further intervention. (Management by taxes, on the other hand, would require detailed economic modeling on a real-time basis.)

Modeling is expensive, especially so for specific and complex systems models. Management systems that rely on a minimum of modeling expenditure are obviously desirable. The benefit–cost ratio for simple analytic models often seems vastly greater than that for full-scale computer simulation.

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CHAPTER 6

Models for Forestry*

G.I. Ågren

6.1. Introduction

Forest models are very diverse in their structure. Yet, in their relation to ecological or economic aspects they show a typical bimodality. The models are either oriented towards the forest as an economic resource for society with very little concern for the ecology of forests, or the forest is viewed as an ecological system that eventually will produce some economic benefits. In the cases where economic and ecologic issues are discussed within the same model, any feedback between the ecological and economic subsystems is generally mediated via a manager or model user setting different parameter values. Of the models discussed in this chapter there is only one, the spruce budworm model, which integrates economy and ecology in a mathematical structure, all the others being strongly biased toward economy or ecology.

In a short chapter like this, several approaches must be excluded. To some extent the choice of models to be presented reflects the interest of the author but is mainly a result of selected sampling from the literature. A class of models I have deliberately excluded is yield tables (or more sophisticated models), which mainly have merchantable timber products as outputs. I have done so because these models are generally statistically based with very little ecological theory and do not address the economic problems of what to do with the timber or at which rate to harvest. For example, a number of such models can be found in Fries (1974). Forest succession models is another class of

*This work was supported by the Swedish Natural Science Research Council and the Swedish Council for Forestry and Agricultural Research.

models that I have excluded (*see* e.g. Shugart and West, 1980). These models would not have been inappropriate to discuss here but their generally very long time perspective, hundreds of years, puts them outside the scope of economic policy.

The five models I finally decided to include in this discussion are therefore not a complete coverage of the field but represent some of the state of the art and display as well the great heterogeneity prevailing in this area.

6.2. Nutrient Flux Density Model

6.2.1. Introduction

Circulation of nutrients is a subject dealt with extensively in ecological research (e.g., Clark and Rosswall, 1981). Economically, it is interesting because of the possibilities of enhancing the nutrient cycling through fertilization with subsequent increased yields. The nutrient flux density model, developed mainly by G.I. Ågren (Ingestad *et al.*, 1981), is an extremely simple representation of a coniferous forest stand. The model was constructed for the purpose of capturing the most important aspects of the nitrogen dynamics, other nutrients being neglected. The emphasis in the objectives of the model lay in the basic understanding of the interaction between nitrogen turnover and biomass development in a forest. Although the model initially was intended for ecological understanding it soon became apparent that the model had great potentials for practical applications in fertilization experiments (Willén, 1983) and evaluations of impacts of acid precipitation (Ågren, 1983; Ågren and Kauppi, 1983).

6.2.2. Model structure

In this model, three state variables and six parameters forming a system of nonlinear ordinary differential equations were considered enough to attain the objectives. The state variables are needle biomass, nitrogen in the needle biomass, and nitrogen in the needle litter ("soil"), respectively (*Figure 6.1*). Other life forms than the dominant tree species are incorporated in the parameters. The nonlinearity in the model is a negative feedback from the needle biomass to the production rate of new needles. The external world can influence the model behavior by controlling the rate of nitrogen flow (fertilization rate) into the nitrogen needle-litter pool.

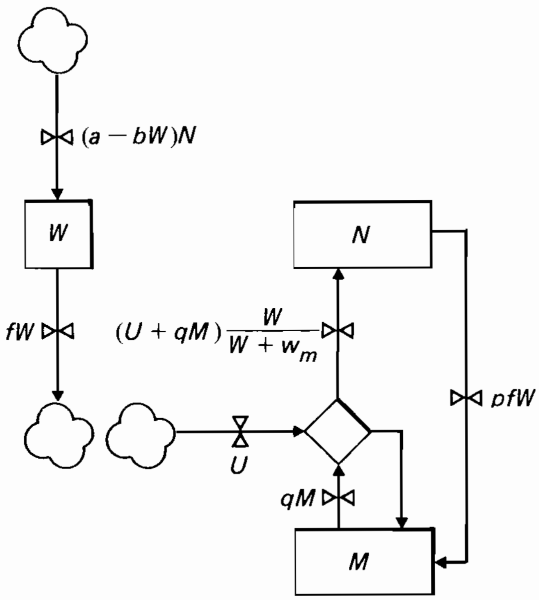

Figure 6.1. Flow chart of the nitrogen flux density model. *W* and w_m are carbon amounts in tree needles and ground flora leaves, respectively. *N* and *M* are nitrogen amounts in tree needles and "soil", respectively. *U* is external nitrogen inputs; lower-case letters denote parameters.

6.2.3. Model evaluation

The model was initially applied in interpreting a fertilization trial (Ingestad *et al.*, 1981) in an academic environment. The increased yields in this experiment have, however, attracted the attention of the Swedish forest industry and one of the major forest companies is now performing a large-scale experiment based on the model predictions (Willén, 1983). With the long response times of coniferous forest it is still too early to draw any conclusions about the economy of large-scale continuous fertilizations of forests or potential environmental hazards due to nitrate leaching.

Another application of the model has been an analysis of impacts of acid precipitation. One expected consequence of acid precipitation is that high sulphur depositions should cause damage to soil microorganisms resulting in decreased mineralization rates of nutrients. The model analysis indicates that coniferous forests with a high circulation of nutrients within the tree are well buffered against perturbations of this type (Ågren, 1983). The other major component in acid precipitation, nitrogenous compounds, should on the other hand have a stimulating effect on forest growth up to some critical level where the forest becomes saturated. Beyond this point further depositions are likely to result in damage. Preliminary results from a study within the framework of the IIASA project on transboundary air pollution (Ågren and Kauppi, 1983) point at urgent needs of reduced pollution levels in the most exposed areas of Europe.

With the very simple structure of the model, the use of it is inexpensive. Most data required by the model can be obtained from the scientific literature – they represent general properties of the tree species. In principle, only initial values of the state variables and estimates of mineralization rates of soil nitrogen must be obtained uniquely for a specific situation. Many of the properties of the model can be derived analytically and where computer simulations are required, simulation times are only a few seconds.

6.3. FORCYTE

6.3.1. Introduction

With a perspective of increasing world demand for forest products and a history of overexploitation of forest in Canada, a Canadian team (J.P. Kimmins and K.A. Scoullar) set out to construct a model to investigate the long-term consequences of increased biomass harvesting and possible countermeasures by management actions, e.g., fertilization. The model was to be general in its structure so that it could be applied to a variety of forest conditions and management treatments. A feedback between nutrient availability and plant growth should be included so that the model would respond to site degradation or improvement. It was further desired that inventory-type data should be the required form of input data, avoiding detailed process descriptions which require large amounts of scientific work to derive. The work has resulted in a series of models named FORCYTE (FORest CYcling Trend Evaluator), the current version of which is no. 11 (Kimmins and Scoullar, 1982). With each new version additional attributes have been added and the flexibility of the model has been increased.

6.3.2. Model structure

The driving functions for plant growth consist of site-specific volume/age equations of the Chapman–Richards type but with a feedback from nutrient availability that adjusts the current growth, up or down, to match nutrient availability. The growth equation simulates stem development to which age-specific ratios are applied to yield development of other organs. Litterfall depends upon site fertility and tree stocking density. The decay rates of litter vary both in time, increasing initially and decreasing as the material approaches humus conditions, and with litter type. Parallel to biomass development is a flow of nutrients, where an "available soil nutrient pool" serves as a switch between mineralization of soil organic matter, inputs from precipitation, canopy leaching, soil weathering, slope seepage, and fertilization and outputs through immobilization, soil leaching, and uptake by plants. Minor vegetation forms are treated in the same way as the trees but in less detail. The model can handle both coniferous and deciduous species (a maximum of three different species) as well as tree size distributions. Nutrients included are nitrogen and an optional two others. Several management interventions are possible in the model: planting, thinning, harvesting, and fertilization (*Figure 6.2*).

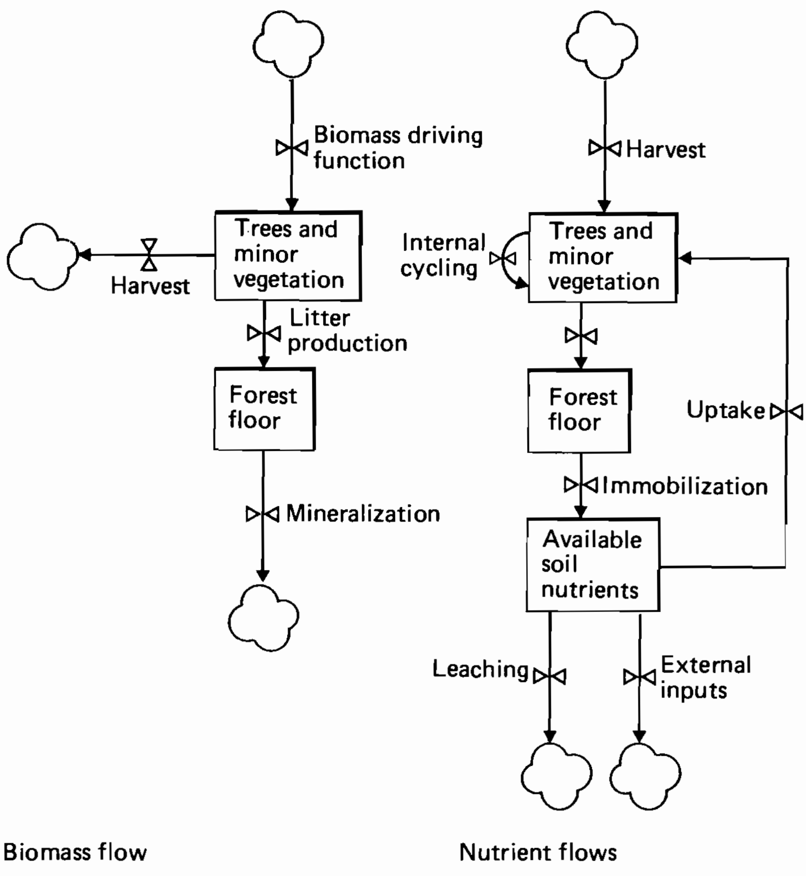

Figure 6.2. **Simplified flow chart of FORCYTE showing the principal structural components and major controlling variables.**

6.3.3. Model evaluation

A series of field projects have been initiated for validating FORCYTE. These include biomass accumulation and nutrient dynamics in an age series of Douglas-fir on sites of different productivities. Different perturbation experiments have also been conducted: slashburning experiments to study aspects of nutrient losses, herbicide spraying of brush vegetation as a means of changing competitive interactions, and changes in nutrient losses from the forest floor following clear cutting. Preliminary analyses with FORCYTE indicate that short rotations combined with whole-tree harvesting can drastically reduce yields in later harvests (Kimmins *et al.*, 1981).

With the evolution of the model the objectives have also broadened so that later versions (10) include also economic analyses of various management interventions. Energy budgeting is another feature that has been introduced. A special development of the model will be the extension from single stands to regional levels where it can be used as a management simulator.

The model leads to a fairly large computer program, version 7 extending over 3527 lines of FORTRAN code. It likewise requires large amounts of information about the biological processes. Version 7 needs about 120 parameter values and 30 tables as inputs. Later versions with additional tree species and nutrients, as well as more detailed process descriptions, will have further increases in the number of these values.

6.4. DYNAST–MB

6.4.1. Introduction

The problem for which this model was designed (S.G. Boyce 1977, 1978) was to order and interrelate multiple benefits from a forest. DYNAST–MB (DYNamically Analytic Silviculture Technique – Multiple Benefits) is a cybernetic system in that it is guided towards a goal by feedback processes (management actions). It is based on a computer model of a managed forest where harvest of timber is regulated to guide the forest towards a steady state with the desired configuration of benefits. Four categories of benefits are considered: timber production yield, suitability of the forest for wildlife (several species of game and nongame species), esthetic values, and sediment flow. For each particular benefit an index, range 0 to 1, expressing the current status of the forest is calculated.

6.4.2. Model Structure

The forest is divided into seven habitats characterized by the average sizes of the trees: seedling habitat, sapling habitat, 6-inch pole habitat, 8-inch pole habitat, 10-inch pole habitat, mature timber habitat, and old-growth habitat. The forest develops through a linear flow through the different habitats, each habitat type having its characteristic time constant. Superimposed on the internal dynamics of the forest are management actions which determine the fractions of forest in mature and old-growth habitats to harvest. In the management actions lie also a decision on the sizes of openings to be created when removing timber. Indices are then calculated by weighing the fractions of the forest in the different habitat types and the number of openings and their sizes (*Figure 6.3*). Indices considered are:

(1) Potential timber yield: timber yield when timber production is favored over all other benefits.

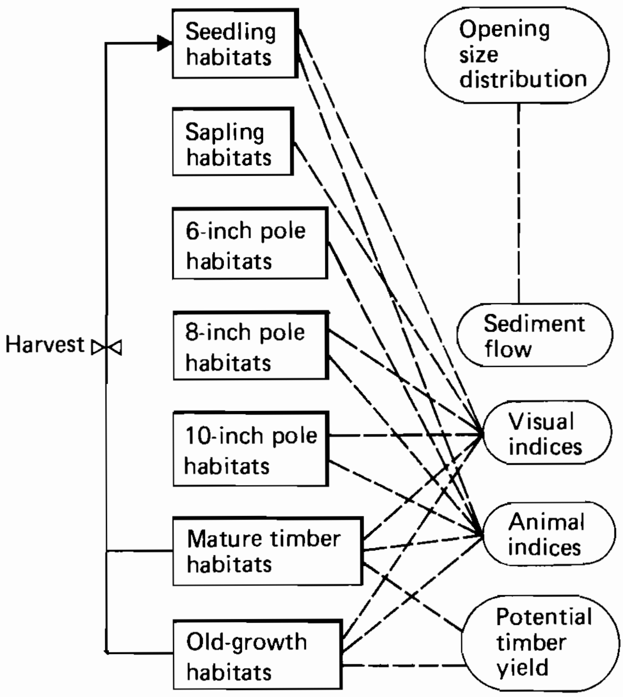

Figure 6.3. Flow chart of DYNAST–MB showing the relationship between the habitat state variables and the outputs in form of various indices.

(2) Sediment flow: a function of the density of openings.
(3) Visual indices: estimate the visual aspect of the forest. They depend upon the opening size distribution and the balance between habitat types.
(4) Animal species indices: squirrel, deer, and black bear are included, each species with its favoured combination of habitats.
(5) Bird indices: four songbird species, four woodpecker species, turkey, and grouse are considered, each having its own preferred habitat.

6.4.3. Model evaluation

The model has been evaluated by running it over a series of management regimes ranging from 100 to 50% of maximal timber production and as a special case no management at all. When decreasing timber production from its maximal value all other benefits improve initially, basically owing to the increase in old-growth habitat which is absent under maximal timber production. With further decreases in timber production rates the indices begin to diverge. For example, some wildlife species are disfavored (e.g., turkey) whereas other indices (e.g., ugliness) continue to improve.

The size of the model is relatively small, requiring only 150 lines in the DYNAMO language. On the other hand, using over 200 variables and parameters (some of which are expressed as tables) the model demands considerable amounts of information as inputs.

6.5. Spruce–Budworm Model

6.5.1. Introduction

Extensive insecticide spraying in Canadian forests proved efficient in minimizing tree mortality but maintained the insect (spruce budworm)–forest (balsam fir and white spruce) in a state of instability with respect to external perturbations. A systems analysis attack on the problem of defining alternative policies for this complex problem in time and space which should achieve specific objectives and yet be robust was initiated by Holling, Jones, and Clark (1977). It was also seen as an experience in the state of the art of systems analysis and not only an academic exercise in constructing a model of the spruce–budworm system. A number of indicators were produced, enabling managers and policymakers to evaluate consequences of different management actions.

6.5.2. Model structure

Since spatial patterns were considered as key features, the forest region was subdivided into smaller sites. For each site identical models were used, identical except for initial conditions and external driving variables (weather). The site model (*Figure 6.4*) has two principal elements: a budworm survival model and a forest response model. The forest is described by the fractions of the forest in 75 age-classes. In addition, the amounts of foliage divided into new (preferred food by the spruce budworm) and old is specified. The forest develops by discrete transitions between age-classes, except from the oldest one which serves as an accumulator. Not all trees are transfered to the next age-class but some die for "natural" reasons or due to insect attacks which are measured by the level of defoliation. Trees can also be removed by harvesting. Foliage dynamics is described in a similar way.

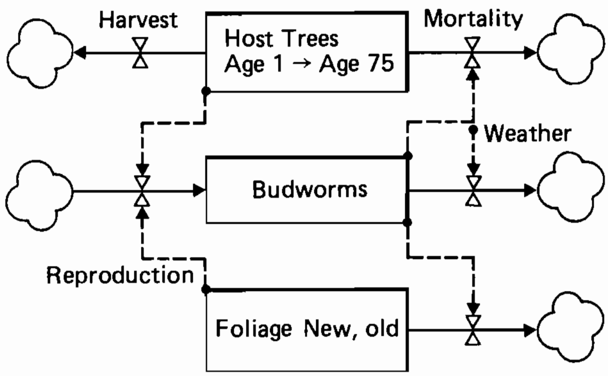

Figure 6.4. Simplified flow chart of the spruce budworm site model showing the interactions between the spruce, the budworms, and the external variable, the weather.

The budworms are described by a simplified life history: eggs, small larvae, large larvae, pupae, adults. Survival of small and large larvae as well as fecundity depends on the quantity and quality of foliage. The other stages in the development of the insect are independent of the forest condition. Competition between insects influences both their survival and the amount of foliage eaten per individual. An important trigger of outbreaks is the randomly fluctuating weather conditions which are simulated as mainly affecting the large larvae survival. Finally, dispersal of the budworm between sites occurs when females deposit eggs outside her site (Jones, 1977).

6.5.3. Model evaluation

The model produces temporal–spatial maps of tree volumes and egg densities. These have been used for analyzing a series of management objectives. Since objectives are both ambiguous and contradictory in complex management situations, a series of different formulations of objectives have been reached with regard to their consequences. Some of the objectives are (Holling, 1978): retain historical management approaches, maximize long-term profits to logging industry, minimize budworm density, eliminate all human interventions.

Rather than evaluating the model against quantitative data it was considered as more essential that the qualitative patterns in time and space could be reproduced. The model has been successful in predicting the correct intervals between outbreaks (temporal pattern). It also gives the observed wave-like spread of an outbreak from an initial source. In its first application the model was used in the province of New Brunswick, Canada. Later it was demonstrated that only by changing weather conditions and initial conditions of the forest could the outbreak patterns of Ontario (longer intervals between outbreak episodes) and Newfoundland (outbreaks requiring dispersal from the interior as triggers) be derived.

Economic analysis with the model has been performed using dynamic programming on a simplified system (Winkler, 1975). These results are extreme ones and do not represent "optimal" solutions because they have not been derived with consideration to, for example, forest industry capacity. They do, however, serve as a base for further exploration of the problem. Another approach to the problem has been through the introduction of stochastic dominance (Thompson *et al.*, 1979), which shows that it is possible to rank at least some of the policy options unambiguously even when outcomes only form a statistical distribution.

The temporal and spatial distributions aimed at have led to the requirement of a large number of variables, in fact 20 935. However, this large number essentially comes from the division of the region into 265 subregions and the trees into 75 age-classes. The number of qualitatively different variables is much smaller, only 5. The number of parameters required to describe the system is around 40 plus three arrays of parameters with a value for each age-class of the trees.

6.6. Gippsland Plantation Model

6.6.1. Introduction

Plantations represent an important part of the capital cost in plantation-pulp mill enterprises. Managing forest in terms of silvicultural methods (thinning, fertilizing, weed control, etc.) must therefore be balanced against alternative ways of obtaining wood. The Gippsland plantation model (Dargavel, 1978) is an application of linear programming to solve this optimization problem for the utilization of *Pinus radiata* plantations by an Australian forest company with a planning horizon of 25 years.

6.6.2. Model structure

The plantations consisted of some 40 000 ha which could be subdivided into 188 relatively homogeneous stands representing a variety of growth conditions. Intensive data collection had made it possible to develop regression equations for height, basal area, and volume over time for all site conditions of interest (Turner *et al.*, 1977). The possible management actions considered were:

(1) Fertilization which added to both height and basal area increment over some fixed duration of response.
(2) Weedicide application which partly was incorporated in fertilizer response and partly resulted in improved planting survival.
(3) Tree breeding increasing basal area at age 10 by up to 20%. Breeding was assumed to be done in conjunction with fertilization, thus making the two actions additive.
(4) A series of thinning and clear-felling options.

Evaluation of all these possible combinations of actions was not feasible, a maximum of about 10 000 combinations for all the 188 stands combined was set, leading to a selection of 20–140 (average 54) strategies to be chosen as the main alternatives for each stand. The optimal was the one maximizing corporate profit. However, the best alternative for any one stand could not be chosen independently because of the need to meet overall objectives and share overall resources. The constraints on the solution were that:

(1) Wood supply had to satisfy the capacities of present and planned mills and yet not exceed the possibilities of increasing their capacities.
(2) The cost of the forest operation for each selected combination of strategies must not exceed the budget available each year.
(3) The standing stock at the end of the planning period must represent a sustainable yield.

6.6.3. Model evaluation

The validity of the model has been established by having local people react to the strategies suggested as well as having external consultants reviewing major parts of the model. All uncertainties were not removed in these steps, being inherent either in the plantations or the markets or the company's knowledge of them.

The model has demonstrated that the forecast of future demands can significantly change the thinning method and clear-felling age – reduced or delayed demands increase the amount of clear-felling in the immediate future but reduce thinnings. Increasing the budget for forest operations not only increased the intensity and extension of these but led also to changes in the methods used of the same type as noted above.

A model of the type described here requires a very large data base. A team of seven persons spent five months gathering data, constructing functions and building computer models. The computer model was also very time consuming, requiring up to 8 hours of computer time (IBM 370/135) to obtain an optimal solution. Revisions with respect to constraints were more rapid, taking 2–6 hours.

6.7. Overall Evaluation

The common feature of the models presented in this paper is the explicit use of time. This is quite natural as a prominent property of a forest is change over time – a "static" forest would not be of much interest in a model. The five models cover a wide range of purposes, from being intended mainly as scientific tools for analyses (nutrient flux density model) to dealing with general policy issues (DYNAST–MB, FORCYTE) or scrutinizing specific problems (spruce budworm model, Gippsland plantation model). In all cases, the ecological submodel (the forest) is formulated as a simulation model. The economic submodel represent a manager trying to manipulate the ecological system toward some present goals. In only one model (Gippsland plantation model) is this optimization routine integrated in the model. In the other models the optimization has either been included in certain applications (spruce budworm model), is done off-line or is only presented as a possibility.

The time horizons and time steps in the models are those natural to a forest, a rotation period and a year respectively, although in absolute terms the rotation period can vary between 25 and 200 years depending on the geographical location of the particular forest. Geographically, the models have a natural local base, a stand. Three of the models operate with several stands simultaneously although the interaction between the stands is via the economic submodel in two of them (DYNAST–MB, Gippsland plantation model) and only the spruce budworm model has connections, and critical ones, between the geographical subunits in the ecological submodel. Forest models with a wider geographical coverage (e.g., Lönnstedt, 1983a,b) tend to become entirely dominated by the economic issues.

Two philosophies among the model constructers can be discerned: simplicity or completeness. The first school, represented by the nutrient flux density model and the spruce budworm model, tries to find a minimum representation for a given problem. In general, these models are derived from basic principles or some clearly stated hypotheses about the system properties. The second school, to which the other three models belong, approaches the problem more inductively, gathering a large data base from which some, often statistical, representation of the system can be made. Also the attitudes toward model testing differ, to some extent because of the difficulties in performing validation tests on these types of models. In no case is an explicit comparison between model output and "real world" data performed. Only with the spruce budworm model has it yet been possible to compare qualitatively the model behaviour with observations. In other cases, data collection and experimentation are going on to provide a basis for model evaluation. However, the general attitude seems to be that if the basic ideas, functions, and parameter values are correct, then the combination of these will, also, yield results that are, if not entirely true, at least reasonable.

The exposé of the five models in the previous sections clearly demonstrates that there exists no base from which to start in constructing forest models. I think this is something that we will have to live with for a long time. Although much progress has been made in the understanding of forest ecosystems over the last few years there is no unified picture allowing the appropriate model to be selected unambiguously in a given situation. We must expect to see a multitude of models also in the future. The art of model building in the domain of the ecology–economy is still young, the oldest reference in the reference list dates back only 10 years.

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CHAPTER 7

Agricultural Models

Z. Harnos

7.1. Introduction

Models are built for agriculture and included or related fields, such as plant production, animal husbandry, soil sciences, water management, and so forth, for very different purposes. Selection of a particular model is usually determined by the problem investigated.

A considerable part of the models describes the interaction between production and ecological conditions. This is obvious, for environmental conditions have an important effect on the level of agriculture. Depending on how far the possibilities of utilization of ecological conditions are connected to the investigation of economic problems, there is a shift in stress from the ecological to the economic side.

The impact of ecological conditions on the production is not yet very well known. At the same time, however, they have a more and more important role in agricultural planning. So-called biomass programs were initiated in many countries of the world. Their aim is not only to determine the conditions to increase the production, but to secure the balance of the production and environment as well. The ecological character of the models selected for this chapter is justified by the above considerations. The three models are, at the same time, good complements to one another.

The first model examines the possibilities of plant production setting out from an ecological basis. The second one describes how to adjust production to ecological conditions taking into account the dynamic impacts of production on the environment. Results of the first model could serve as inputs to the second. The third model is related to pest management making possible joint analysis of ecological and economic parameters. Such types of models can be used in planning the long-range structure of plant production very well.

Methodologies of the models applied in agriculture follow the nature of the problems examined and since the problems are heterogeneous, the methodologies are very diversified. Recently, dynamic, control-type models have become increasingly general to describe these phenomena. This is reflected in the models to be described. We can say about their methodology, that the first model uses simulation, the second one multipurpose optimization, while the third one can be considered as a control model.

7.2. A Physical Crop Production Model and its Environmental Feedback

7.2.1. General

The aim of this study is to describe analytically the relations among productivity of plant production, the natural environment, and the applied agrotechnology.

The model provides the following possibilities:

(1) The prediction of expected yield of certain crops with the knowledge of the environmental parameters.

(2) With the long-term monitoring of the production and environmental factors, it is possible to check the impacts of production on the environment and based on that it is possible to decide whether to continue the applied agrotechnology or to change it.

The model-system was prepared by N. Konijn at IIASA and used for the Nitra, Czechoslovakia, case study.

7.2.2. Model structure

The model-system consists of five parts:

(1) It determines the possible dry-matter production on a purely geographical basis for each decade for C3 and C4 crop types. Dry matter production can be determined on the basis of geographical latitude, the number of sunny and clouded days and the radiation.

(2) It modifies the potential dry-matter production according to the water-supply. In the model, the dry-matter production including water constraint depends basically on the potential dry-matter production and evapotranspiration. To calculate the latter one is a more complicated task, since it is related, beside the crop-type, to the soil type, soil moisture content, rainfall, irrigation,

capillary rise, run-off, and drainage. Here, irrigation tends to modify the effects of bare natural conditions. Distribution of dry-matter among different parts of a plant can be determined depending on the stage of development.

(3) Beside the water constraint, agrotechnology affects directly the development of the plant only via the nutrient supply. Nutrient supply is possible from two sources:

(a) Fertilizers.
(b) Decomposition of organic matters in the soil.

The latter source is not included in the model. The development of the plant can be determined from the parameters described above.

(4) The model-system describes the environmental impact via soil erosion. Erosion is described by the Universal Soil Loss Equation (USLE) which takes into account, beside the noncontrollable natural parameters, e.g., length of the slope, steepness, type of plant, and agrotechnology implied, and the organic matter content of the soil via the soil characteristics. It is possible to have direct or indirect effect, that is feedback, on the environmental conditions via selection of the agrotechnology and supply of nutrients.

(5) The resource adjustment model keeps track of the soil organic matter content, the soil moisture characteristics and the available nutrients. Erosion affects the organic matter content of the soil, and indirectly the moisture content of the soil as well.

Instead of a detailed mathematical description of the model (*see Figure 7.1*), only its main relationships are presented.

The potential dry-matter production can be calculated by the following relation:

$$\text{pdm}_{j,\Delta t} = f\,[j,\Delta t,k,m,\text{rad},\text{tab1}(j,k,\Delta t,m)] \tag{7.1}$$

where j = type of plant (C3 or C4), Δt = 10-day period of the year, k = latitude in grads, m = clear or overcast day, rad = radiation, and tabl = the dates given in the tables.

The dry-matter production including water constraint is:

$$\text{wdm}_{l,c,\Delta t} = \text{pdm}_{j,\Delta t}\left[\frac{E_{\delta,l,c}}{Ep_{\Delta t,l,c}} + \frac{E_{\Delta t-\delta,l,c}}{Ep_{\Delta t,l,c}}\right] \tag{7.2}$$

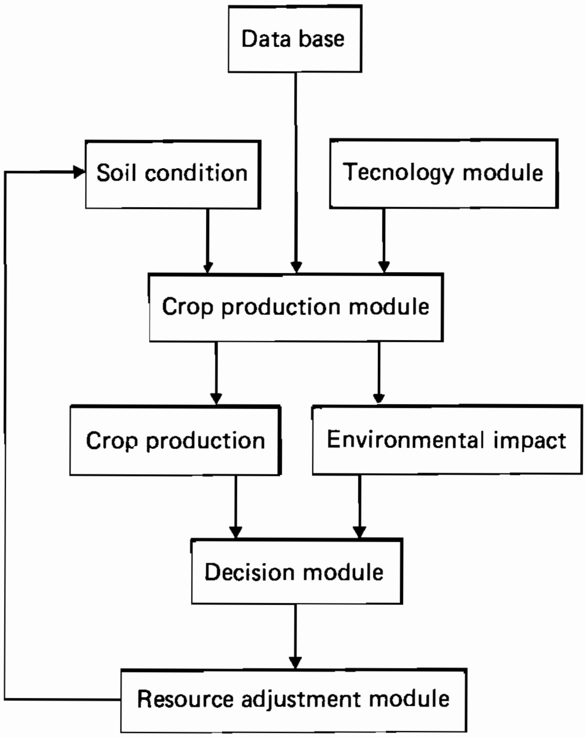

Figure 7.1. The general model structure.

$$0 \leq \delta < \Delta t \tag{7.3}$$

where, l = land class, c = crop, and E = actual and Ep = potential evapotranspiration. In order to determine E, equations concerning the water balance and the evapotranspiration must be established.

To determine the availability of water for plants we use the water balance in the following form:

$$S_{t+\Delta t,l,c} = S_{t,l,c} + P_{\Delta t,l,c} + I_{\Delta t,l,c}, + C_{\Delta t,l,c} - E_{\Delta t,l,c} - R_{\Delta t,l,c} - D_{\Delta t,l,c} \tag{7.4}$$

$$(t = 1,36; \quad l = 1,l; \quad c = l,c)$$

where S = soil moisture content of the root zone; P = rainfall, I = irrigation, C = capillary rise, E = evapotranspiration, R = runoff, D = drainage (in cm), l = land class, c = crop, t = time period.

The water balance determines the soil moisture content at the end of the time period t knowing the initial moisture content and the terms specifically for the time period concerned.

The evapotranspiration from a free water surface can be approximated by the so-called Penman formula:

$$E_{ws} = [\Delta (R_n - G)/L + \gamma (e_s) - e_a) f(u)]/(\Delta + \gamma) \tag{7.5}$$

where E_{ws} = evaporation from a free water surface, R_n = net radiation, G = soil heat flux, Δ = rate of change of the saturation vapor pressure with temperature, γ = psychrometric coefficient, e_s = saturation vapor pressure, e_a = actual vapor pressure, $f(u)$ = wind speed function, L = latent heat of vaporization.

The relation between the nutrient supply and the plant can be described by:

$$Y_N = Y_N^0 + \alpha_{C,N}\, \beta_{C,N}\, V_N \tag{7.6}$$

with Y = marketable yield, Y^0 = yield without fertilizer application, V = amount of fertilizer applied, α = uptake coefficient, β = efficiency coefficient, C = crop or crop variety, and N = kind of fertilizer.

At each yield level we assume relative uptake between nutrients described by $U_P / U_N = C1$ and $U_K / U_N = d2$, where U is the uptake and the nutrients are given as subscripts.

The effect of water erosion is described by the Universal Soil Loss Equation (USLE):

$$A_{l,c,\Delta t} = R_{l,\Delta t} K_{l,\Delta t} L_{l\,\Delta t} S_{l,\Delta t} C_{c,\Delta t} P_{l,c,\Delta t} \tag{7.7}$$

where, A = soil loss, R = rainfall erosivity factor, K = soil erodibility factor, L = slope length factor, S = slope grade factor, C = crop and management factor, P = practice support factor, and l,c, and Δt = land class, kind of crop, and time period, respectively. The rate of change of organic matter decay is a function of the amount of material in the particular fraction:

$$\mathrm{d}fr_j / \mathrm{d}t = -kfr_j \tag{7.8}$$

with fr = amount in fraction, k = coefficient of decay, and j = the fraction.

The coefficient of decay will, however, change with time due to the change in heterogeneity of each of the fractions.

$$dk_j/dt = -qk_j \tag{7.9}$$

7.2.3. Model evaluation

The shown physical crop production model has the same structure as the physical crop production model developed by the Centre for World Food Studies (1980). Since this latter one has been transferred to many other places, we can assume that it is widely used at least on a research level. The equation concerning the water erosion (USLE) used in the model system is widely known. It is discussed in detail in *CREAMS*, and the accompanying package makes its practical utilization much easier. It is a great merit of the study that it includes the physical crop module and the environmental impacts of the production in a single system, increasing in this way the utilizability of the results.

Presumably, the practical usefulness of the results provided by the model-system depends largely on the:

(1) Homogeneity of the land.
(2) Reliability of the starting data base.

Homogeneity can be ensured for relatively small areas only. If the model is to be used for forecasting purposes then the stochastic nature of the weather must be taken into account. Only trends can be concluded from the average climatic effects. Useful results for this purpose which can promote application of environment-protective agrotechnologies. The need for a detailed data base makes the utilization of the model difficult. Collection and processing of reliable data assume serious intellectual and budget potential.

The application of the model for any geographical area assumes the check of its validity. Therefore the utilization of the model is time-consuming and expensive, at least initially.

7.3. Methodology for Investigating of Long-Term Consequences of Technological Development in Hungarian Agriculture

7.3.1. General

The aim of this study is to describe short- and long-term effects in the utilization of fertile soil and to promote elaboration of land-use policies ensuring the long-term sustainable soil use. The role of ecological aspects is more important in the model-system than that of economic considerations.

The major objective of the study to give answer the following questions:

(1) What are the production potentials of the existing soil resources and how can these be increased and utilized?
(2) How to increase the productivity and efficiency of Hungarian agricultural production by using more rational combinations of existing technological alternatives?
(3) What are the long-term consequences of the several land use policies?

The model system to be presented synthesizes the methodologies for land-use of the different projects:

(1) The agroecological potential of the Hungarian agriculture in 2000.
(2) The long-term possibilities for utilization of materials of biological origin (biomass).
(3) The impact of the agricultural technologies on the production and environment.

Two of these projects were directed by the Hungarian Academy of Sciences and the third one was the Hungarian Case Study in the Food and Agricultural Program at IIASA. Results of these models were used in the first nationwide study to work out long-term economic development policies and to reveal possibilities of more intensive utilization of agroecological conditions.

7.3.2. Model structure

The mathematical model of the system is a multiobjective linear control-problem describing the interaction between soil utilization and changes in the condition of the production site. Dynamic changes in the conditions of the production sites can take place in two directions:

(1) Fertility is decreasing if inappropriate agrotechnology is applied/erosion, acidification and so forth.
(2) Conditions of the production site can be improved by amelioration or by application of ameliorative agrotechnology.

Relations between control and state variables are shown in *Figure 7.2*. Amelioration and the level of agrotechnology applied are controlled by control conditions, that is by investments. Behavior of the system is basically controlled by the conditions prescribed for the structure and quantity of the production. The most important relationships are shown by *Figure 7.3*.

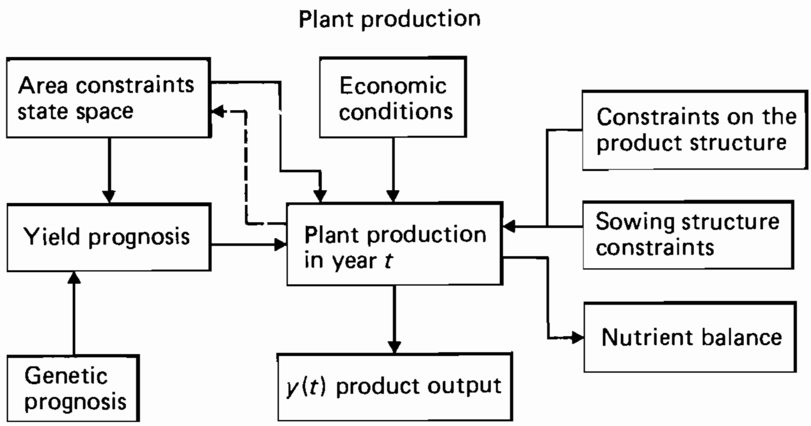

Figure 7.2. Plant production.

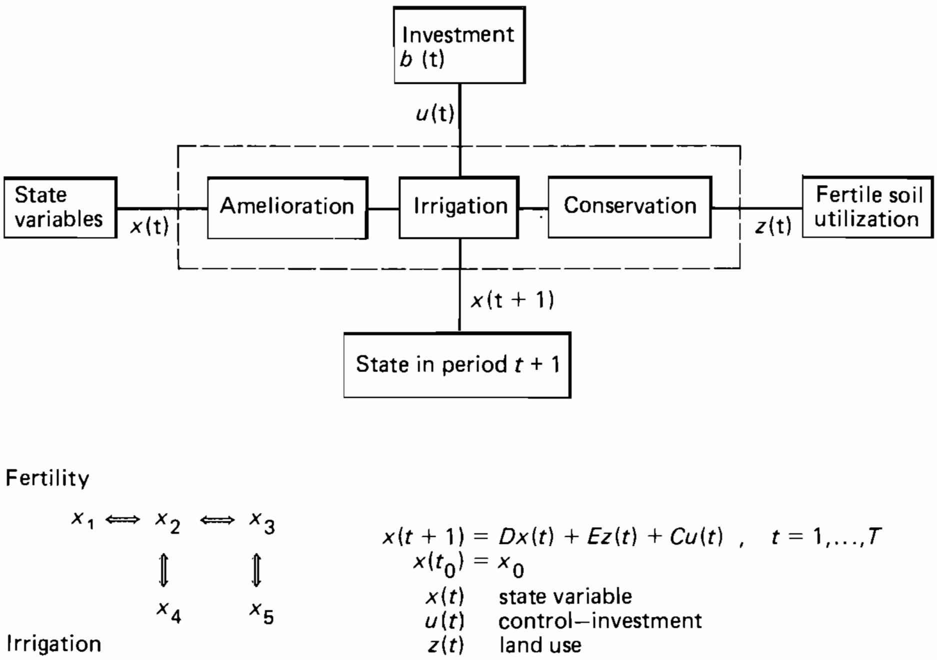

Figure 7.3. **Relations between control and state variables.**

The level and the product mix of plant production in one period are controlled by:

(1) Conditions in the production site.
(2) Yields of the species grown.
(3) Nutrient supply.
(4) Constraints for the production structure and sowing structure.

The agrotechnology applied and amelioration exert direct effects on the state of production site thus its fertility will change as a result.

Capital used as control is divided into three parts: investments in agrotechnology and amelioration and nutrient supply. Irrigation is included in amelioration and agrotechnology. This type of capital allocation provides the possibility to contrast the effects of increasing the short-term benefit by increasing the yields via higher nutrient supply and of the long-term consequencies of lack of amelioration and intensive soil utilization.

The goal function used in the model aims at increasing the fertility of the soil. Reference curves were determined on the basis of assumed genetic development of average yields which is a logistic curve. The aim is to follow these curves in certain sense.

The *state equation* of the control problem is:

$$x(t+1) = Dx(t) + Ez(t) + Cu(t) \tag{7.10}$$

$$x(t_0) = x_0 \qquad t \in \left[t_0, t_1, \ldots, T\right] \tag{7.11}$$

where $x(t)$ state variables represent the state of soils, $u(t)$ control variables represent the resources to be distributed, and $z(t)$ system variables represent the sowing structure; D, E, and C are constant matrices.

Control conditions are described by a system of linear inequalities:

$$Bu(t) \leq u_0(t) \tag{7.12}$$

$$u(t) \geq 0 \qquad t \in \left[t_0, t_1, \ldots, T\right] \tag{7.13}$$

Operation of the system is described by three different types of inequalities:

$$x(t) \leq Fz(t) \qquad t \in \left[t_0, t_1, \ldots, T\right] \tag{7.14}$$

represent the relation between available land area and the sowing structure;

$$Az(t) \leq b_1(t) \qquad t \in \left[t_0, t_1, \ldots, T\right] \tag{7.15}$$

represent the conditions for sowing structure; and

$$H(z, u)(t) \leq b_2(t) \qquad t \in \left[t_0, t_1, \ldots, T\right] \tag{7.16}$$

represent the relations between yields and nutrient supply.

The ***transfer*** between inputs and outputs is given by:

$$y(t) = G(t)z(t) \tag{7.17}$$

where $G(t)$ is constructed from the average yields of different plants depending on time and the type of production site.

Production conditions are given by:

$$y_0(t) \leq y(t) \leq y_1(t) \tag{7.18}$$

The difference from the reference curve is measured by the so-called generalized Tschebishev-distance, the analytic form of which is:

$$f(y) = \min_{t,n}\left\{y_n(t) - y_n^*(t), \sum_n \sum_t \left[y_n(t) - y_n^*(t)\right]\right\} \tag{7.19}$$

where $Y_n^*(t)$ represents coordinates of the reference curve. The task is to determine:

$$\left[x(t), u(t), z(t), y(t)\right] \qquad t \in \left[t_0, t_1, \ldots, T\right] \tag{7.20}$$

which satisfies the above conditions and minimizes the $f(y)$ function on the set of feasible solutions. The problem can be solved by linear programming.

7.3.3. Model evaluation

The modification of conditions and goals provided the possibility to work out a large number of development versions in the projects mentioned. Results of the model have justified several hypotheses of different experts.

There is an enormous amount of work involved in construction of the data base necessary for the model (soil assessment, preparation of genetic prognosis, forecasting average yields, determination of possibilities, and necessities of amelioration and irrigation, and so forth). Therefore, this type of model can only be connected to projects which have either the appropriate data base or sufficient resources to construct it. This, however, requires a considerable amount of time and effort (1.5–2 years).

7.4. Multiseasonal Management of an Agricultural Pest

7.4.1. General

This study is concerned with the multiseasonal crop-pest management problem. The main result is that the timing of the application of pesticide can be used to control buildup of resistance and that the intensity of the application can be used to control the crop yield. These results make possible to establish optimal production–protection policies. The model system was developed by M. Mangel and R.E. Plant for the cotton-spider system in the San Joaquin Valley of California.

7.4.2. The structure of the model system

The basic hypothesis is the following: cotton is an annual crop grown every other year, frequently in rotation with wheat, a crop that does not support mites. Although some mites overwinter in the field, mites also immigrate to the field at some rate throughout the entire season, coming from external sources such as fruit orchards and weed patches.

These external sources of the mites appear to be much more important than the overwintering mites already in the field, for it is assumed at the start of the season the agricultural field has negligibly few pest units.

The model system is based on the submodel for the pest and crop dynamics. The goal of the pest model is, given the fractions of resistant and susceptible pests in the population at year n and the spraying strategy in year n, to find the respective fractions in year $n+1$. Population dynamics and relatively simple genetics are included in the pest submodel.

The submodel for the crop dynamics has the following goal: given the pest populations at the start of year n, and a spraying strategy in year n, what is the yield of crop in year n? Using this submodel we can also determine the optimal spraying strategy within a single season.

The three primary parameters in the application of a pesticide in a given season are the number of applications, the timing of the applications, and the intensity of the applications.

The model system is built up step by step. The first one is the so-called age-independent model, in which pesticide susceptibility and crop consumption are independent of age. In this model it is assumed that all pests are susceptible to the pesticide, the pest population does not reach its carrying capacity before the end of the season, and as a consequence, the growth rate of the pest population is independent of the value C, the crop.

The second model incorporates age dependence in both susceptibility to the pesticide and consumption of the crop. The resistant and the susceptible pests are divided into two groups: young and old.

The mathematical description contains the dynamic model of the crop–pest system. A multiseasonal economic optimization problem is to choose a spraying strategy over N seasons to maximize the profit from the crop harvest.

Here only the simplest of the family of models presented in this chapter will be described. This simplification is possible since the conclusion was that very simple age and genetic structure in the model gives results which are qualitatively the same and quantitatively close to those obtained using a more complex model.

The model comprises a set of differential equations describing the dynamics of the pest population:

$$\frac{dy_k}{dt} = \rho_k a_k + \mu_k(n)I(t) - \gamma y_k - \left(\frac{\omega_y s(t;n)}{e_{yk} + s(t;n)} + \bar{\sigma}\right) y_k \quad (7.21)$$

$$\frac{da_k}{dt} = \gamma y_k - \left(\frac{\omega\, ubas(t;n)}{e_{ak} + s(t;n)} + \bar{\nu}\right) a_k \quad k = R, S \quad (7.22)$$

$$y_k(0,n) = a_k(0,n) = 0 \quad (7.23)$$

The parameters $I(t)$ are the immigration rate, ρ_k is the birth rate, and γ measures the turnover rate from young to adult.

The equation for the crop is:

$$\frac{\mathrm{d}c}{\mathrm{d}t} = r_c c - \nu = \left[x_R + x_S\right] \tag{7.24}$$

$$c(0,n) = C_0 \tag{7.25}$$

where $y_k(t;n)$ and $a_k(t;n)$ are the young and adult pest populations, which are divided into resistant and susceptible subpopulations:

$$x_k(t;n) = y_k(t;n) + a_k(t;n) \tag{7.26}$$

c is a measure of the crop, the continuous variable t, $0 \le t \le T$, represents intraseasonal time, the discrete variable n, $1 \le n \le N$, represents seasonal time.

$$\sigma(t) = \frac{\omega_y s(t)}{e_y + s(t)} + \bar{\sigma} \tag{7.27}$$

$$\nu(t) = \frac{\omega_a s(t)}{e_a + s(t)} + \bar{\nu} \tag{7.28}$$

The parameters ω_y and ω_a measure the maximal effect of the pesticide on the population. The parameters e_y and e_a measure the necessary dose to obtain a given pest kill ratio. The parameters $\bar{\sigma}$ and $\bar{\nu}$ represent natural mortality.

$$\mu_k(n) = x_k(T;n-1)/[(x_R T;n-1) + x_S(T;n-1)] \tag{7.29}$$

$$\mu_k(1) = p_0 \tag{7.30}$$

$$I(t) = I_0(1 - I_c t/T) \tag{7.31}$$

The function $S(t;n)$ has the form:

$$S(t;n) = \begin{cases} \eta\,; t_s \le t \le t_s + \delta \\ 0\,; \text{otherwise} \end{cases} \tag{7.32}$$

for a single dose of pesticide applied at time t_s. The control variables for this problem are the variables t_S and η.

The multiseason economic optimization problem is to choose a spraying strategy $\eta[(n),t_s(n)],n = 1, \ldots, N$ to maximize the profit function J, subject to the appropriate dynamics as given.

The profit function is:

$$J = \sum_{n=1}^{N} \alpha^{n-1}\left[c(T;n) - c_p\,\eta(n)\right]$$

where $c(T;n)$ is the crop biomass in year n at the end of the season, $\alpha = (1 + \gamma)^{-1}$ where γ is the discount rate, and c_p is the relative cost per unit of pesticide. This problem can be solved by the method of dynamic programming.

7.4.3. Model evaluation

The principle of the model presented has been used by many authors. Therefore, the results can probably provide a good description of the phenomenon and the model can probably provide an appropriate strategy. There is no indication of a concrete application of this particular model in the chapter. Therefore the efficiency of application cannot actually be determined. To judge the application in a wider range we quote Mangel and Plant (1982): "We believe that this theory will be applicable to a wide variety of crop–pest systems." The validity of the model must of course be checked in each application.

It is a precondition to the application of the model to determine the parameters of the model and to have the appropriate mathematical software for the numerical solution of the problem. To determine the parameters and to check the validity of the model requires presumably a considerable amount of time and inputs.

7.5. Comparative Evaluation

The simultaneous study of ecological and economic aspects is a natural requirement in a number of areas nowadays. All this is motivated by the fact that the resources which earlier have been considered as renewable do not necessarily regenerate, and often their renewal is not complete.

To provide for the conditions of renewal normally involves serious economic implications, their effect does usually not manifest in direct benefits, and the benefits cannot be measured in monetary value.

This perception gives explanation to the fact that, in contrast with the earlier decision-making practice, only economic criteria were taken into account, now, in addition to economic considerations, often the expected ecological effects are also reckoned with in decision making, e.g., in long-term planning. Mathematical modeling is aimed at helping the work of the decision makers, hence the increasingly frequent occurrence in the recent years of economic models including ecological relationships as well.

The decisions are usually motivated by economic considerations only. This is the reason why most of the traditional agricultural models are of economic character. These models comprise the optimization of the organization of production, the problems of storage, transportation, buffer stocks as well as the determination of optimal nutrient supply or feeding and so on. The applied methodology is relatively rich, although the use of the traditional methods of operation research is overwhelming. These kinds of models proved to be useful and are widely applied. In the sequel I shall deal with some problems related to ecological economic models.

Let us consider the behavior of the agricultural system ecologically. The production within a given period is determined by:

(1) The natural conditions: soil, hydrological conditions, the weather during the period.
(2) The applied technology: cultivation, nutrient supply, pest control, species, etc.

The productivity of the soil changes relatively slowly, it can be considered as a partially renewable resource. The weather can only be handled as a stochastic phenomenon. Hydrologic conditions are determined partly by the soil and partly by the weather.

Production, in turn, affects the soil and the hydrologic conditions, but the effects are delayed. Factors increasing productivity like amelioration or irrigation require huge, slowly recovering investments, deterioration, like acidification, salinization, erosion of the soil, is usually very slow as well. The costs of preventive interventions may consume the profits in the short term, therefore in practice, the whole problem is neglected. On the other hand, delaying these interventions may lead to much greater losses in the long run. At the same time, the presently available methodology to describe the problem is not suitable either.

To sum up, we can establish that there is:

(1) A direct effect of the environment on production.
(2) An indirect effect of production on the environment.

This set of relationships is not yet known enough, therefore an exclusively economic description cannot reflect reality clearly.

Owing to the prevailing time delay, an adequate representation of the ecological aspects, obviously, requires the use of stochastic and dynamic models. As the system is very complex and the computable models are very simple, these phenomena can be studied only in a strongly simplified form.

Another problem related to ecological effects is that often those causing the effects and those suffering from it are different, and therefore there is no interest to prevent the damages. An important example of this is the case of the application of chemical fertilizers, or the disposal of liquid manure which lead to the nitrification of ground waters and to the eutrophication of surface waters. Similarly, the responsibility of the industry for the damages caused in agriculture by acid rains can hardly be determined. The problem is worsened by the fact that the effects often exceed national boundaries, hence only international cooperation could produce a solution. This is to say that:

(1) The damage of the environment may occur far from the causing source.
(2) The effects manifest slowly, with long time delays.
(3) The damage to the environment cannot be evaluated in monetary terms, it cannot be compared with protection costs.

As a consequence the ecological models describing the above phenomena cannot be extended to include economic considerations, nevertheless we very well know that decisions are motivated in most cases by direct economic interests.

Ecological problems in agriculture are usually investigated within the framework of individual disciplines, like soil science, hydrology agrometeorology, genetic, etc. – often in isolation. The aim of the study is the description of the phenomena with little interest in economic aspects. This type of model is exemplified by Konijn (1984), who describes the relationships between the productivity in plant production, the natural environment, and the applied technology.

There are several examples for the linkage of individual agricultural processes by way of ecological and economical considerations. This type of models and investigations is of great use, the knowledge obtained that way is a necessary precondition of the study of the system in its totality. At the same time, due to the necessarily extensive simplifications results can be utilized under restricted conditions, as in a model system elaborated by Mangel and Plant (1982).

Another type of ecological–economical model is represented by the macro level ecological models of the agriculture. Typical questions addressed by these models are:

(1) If a certain prospective economic goal is given, what kind of production and investment policy is to be pursued?
(2) What are the consequences of other policies?

This is exemplified by the model described by Csáki *et al.*. (1982). The model presented by them served as a basis for a number of studies recently carried out in Hungary. The stress in the model is laid upon the ecological relationships. There are, at the same time, certain economic considerations as well, like the allocation of limited resources, but decisions are made according to long-term, non-economic principles. The main reason for this is that the long-term effects of the different policies are not known, or even if so, methods for the consistent economic evaluation of these effects are not available. The purpose of the study is the simultaneous representation in one model of short- and long-term interests in the agricultural use of the land. In concrete terms, this means the search for economically feasible land use policies ensuring the maintenance of a high productivity of the soil, thus providing for a sustainable land use. The study addressed the following questions:

(1) What is the production potential of the soils; what are the ways to increase it?
(2) How can a more rational use of resources and an appropriate selection of technologies contribute to the increase of the productivity of agriculture?
(3) What are the ecological consequences of the different land use policies?

Economic conditions and goals were given in the form of scenarios.

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CHAPTER 8

Water Resources Models

I. Bogardi

8.1. Introduction

The purpose of this chapter is to review three examples of water resources models.

Economic–ecological models related to water resources can be discussed in the context of discrete systems models consisting of the following five elements: input, state, state transition function, output, and output function (Booth, 1967; Wymore, 1976; Bogardi and Duckstein, 1978).

Accordingly, water resource related economic–ecological models can be defined as water resources models with economic and ecological (environmental) components of the input and/or state vectors (*Figure 8.1*).

Thus model elements are defined as follows for time $t = 0,1,2,\ldots$:

(1) The *input* $I(t)$ may comprise:

(a) Environmental elements such as natural factors (wind, rainfall, temperature), physical properties of the area (soil, topography, mineral resources), ecological factors (pollution limits, transpiration demand, etc.).

(b) Economic and social elements such as noncontrollable resources (capital, machine, manpower), requirements (water, mineral, industrial products).

(c) Possible control actions, that is decisions related to resource development (capacity increase, dam construction, hydropower, etc.) and ecology (pollution treatment, effluent charges, artificial recharge).

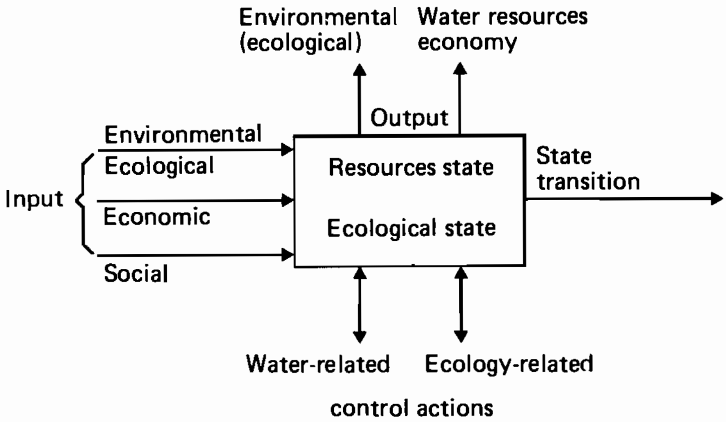

Figure 8.1. General model for water resources.

(2) The *state* $S(t)$ may include the resources state such as the level of irrigation development or the amount of stored water and the ecological state such as the pollution of air, water, and land.

(3) The *state transition function* calculates the state at time $(t+1)$ as a function of state and input at time t:

$$S(t+1) = \varphi(S(t), I(t)) \tag{8.1}$$

As an example a resources state in $(t+1)$ can be calculated from the existing capacity in t and inputs such as capacity increase in t. Typical environmental state transition functions are the mass and chemical balance equations. An example is the partial differential equation describing karstic water movement in the regional aquifer. Note that the numerical solution of such an equation is generally calculated for discrete time steps as in equation (8.1).

(4) The *output* $R(t)$ may have elements of the state vector, especially the environmental state, say air or water pollution, karstic water level or flow. The output for the resources economy may be represented by the typical decision criteria, such as losses, costs, benefit, social indicators. In case of this output, total-period outputs are commonly defined such as discounted costs or benefits. On the other hand, it is often not easy to interpret total-period environmental outputs since there is no ground, for instance, to summarize water flows for the whole period and consider this sum as a derived environmental output. You may consider separately each environment state $P(t)$, select the maximum, or use conceptions of fuzzy control.

(5) The *output function* Ψ calculates the output vector as function of $S(t)$ and $I(t)$:

$$R(t) = \Psi[S(t), I(t)] \tag{8.2}$$

The economic outputs may be calculated by the help of discounted costs and benefits. The environmental output function corresponds to the environmental state transition function.

Note that the first three elements of this model, that is, $I(t)$, $S(t)$ and Φ are sufficient for a dynamic description of the economic–ecological system considered.

Given the above model elements one may select one or more outputs to be optimized and consider the rest as constraints.

These model elements permit the construction of different known groups of systems models (Bogardi, 1982):

(1) Descriptive systems models do not contain decisions among input elements.
(2) Decision models contain decisions among input elements.
(3) Forecasting models calculate output function in different points by simulation or analytic means, and ranking of the systems is possible according to these calculated values.
(4) Optimization models consider the output function as an objective function and seek maximum or minimum.
(5) The model is multiobjective (multicriteria) if the output to be optimized has more than one element such as economic, social, environmental or political criteria.
(6) Static models consider a single and selected time period: state transition functions cannot be interpreted.
(7) Dynamic models consider several time periods: state transition functions are interpreted.
(8) Stochastic models consider stochastic state and/or input.
(9) Discrete modeling techniques such as dynamic programming, integer programming, simulation are used, if discrete values of state and input are considered.
(10) In other cases continuous models such as linear programming can be used.

It is not claimed that these models are among the best in the field. In fact, as all such selection, this one also reflects a small and biased sample. However, there are still three aspects why these models may be regarded as representative:

(1) Each of them aims at crucial common problems (water and agriculture, aquifer management, regional conflict resolution).
(2) Each of them has been applied to real-life decision making.
(3) The definition of economic-ecological models can be fully followed in each of them.

There is an abundant amount of literature on the theoretical aspects and the application of mathematical models in water resources. Examples of comprehensive books on the subject include the works of Buras (1972), Biswas (1976), Hamies (1977), and Loucks (1980). Most models of water-related systems may be considered as economic–ecological models according to the definition given in this chapter.

8.2. Irrigation Scheduling

8.2.1. Introduction

The following group of models, and, specifically the model of Bras and Cordova (1981) refers to the maximization of agricultural economic benefit – the economic aspect, and to the ecology of the interaction among plant, soil, and water, as controlled by irrigation scheduling.

A common feature of these models is that water-limited crop production is assumed. Though this assumption does not hold in many cases, especially in developing countries, where nutrient management or other production factors may be the limiting factors, recent droughts all over the world (e.g., in 1983) highlighted the demand for highly controlled agricultural water management. On the other hand, the joint modeling of agricultural economics and ecology has posed methodological problems even in the simplest, purely water limiting case. We mention two examples. The first is that simulation of the plant–soil–water interaction generally considers deterministic meteorological input (rainfall, temperature), which in reality exhibits unneglectable random variation. This random variation naturally influences water-limited crop yield, being thus also a random variable. Realistic decision models of agricultural water management such as irrigation scheduling should be thus stochastic models. However, only simplified stochastic models of plant ecology can now be coupled with the economic model. Another problem is the very complex process of plant, soil, water interaction requiring an often absent interdisciplinary analysis. In fact, the specific model of Bras and Cordova as shown next endeavors to take both aspects into consideration and to stay still real-life oriented. Among similar models available the one of Schmidt and Plate (1983) is mentioned as a typical realistic example.

8.2.2. Model structure

A growing season for a given crop is considered, and divided into a finite number of stages. In each stage a decision can be made on the amount or irrigation, fertilizer application, tillage operation, etc. Every decision has an influence on the growth of crop, and thus on the final yield. Stochastic, uncontrollably meteorological elements would also contribute to the actual magnitude of the crop yield. Water-limited agriculture means that the plant is not able to satisfy for the growing season its full evapotranspiration demand ET_0, but only a part of it – actual evapotranspiration ET. The magnitude of ET_0 is governed by uncontrollable (for this type of model!) meteorological elements, while ET is controlled by the moisture content of the root zone. This latter one can be modified by irrigation, tillage operation, soil conservation. Then the crop yield is proportional to the ratio ET/ET_0. The moisture content of the root zone changes as influenced by rainfall events, runoff, infiltration, capillary uptake from the groundwater, deep percolation, actual evapotranspiration, and irrigation. On the other hand, irrigation return flow by incremental surface runoff and deep percolation, may contribute to the pollution of groundwater and surface water because nutrients are dissolved in both runoff and deep percolation. These amounts of nutrients are, in turn, governed by fertilizer application and tillage operation. Economic objectives may be to maximize expected agricultural net benefit, or minimize its variance. On the other hand, the environmental objectives may refer to the minimalization of surface water and groundwater pollution.

The specific model of Bras and Cordova seeks to maximize expected net benefit, and does not regard environmental objectives. *Figure 8.2* shows the elements of the simplified model for a stage t in the growing season:

(1) *Input* includes:

(a) Stochastic rainfall events.
(b) Expected potential evapotranspiration ET_0.
(c) Type of crop, irrigation area, soil, and groundwater data.
(d) Cost function of irrigation, price of crop.
(e) Decision: the amount of irrigation water.

(2) The *state* consists of:

(a) Root zone average moisture content ϑ.
(b) Available amount of water supply for the remaining stages v_t.

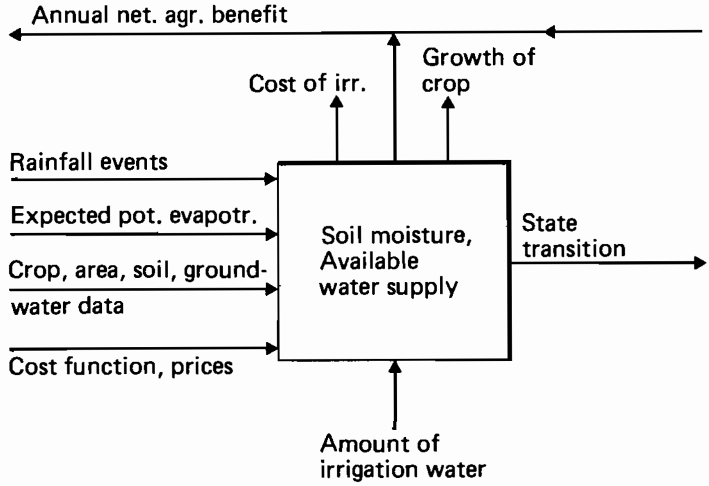

Figure 8.2. Illustration of a simplified model for a stage in the growing season.

(3) The *state transition function* has two parts. The change of moisture content in the root zone can be calculated as:

$$\frac{\mathrm{d}}{\mathrm{d}t} = I_t + f_t + C_p - P - ET \qquad (8.3)$$

where I_t and f_t are the rates of irrigation and infiltration, C_p is the capillary uptake, and P is the percolation rate to the groundwater.

Depending on how members of equation (8.3) are calculated several models are available such as those of Skaggs (1980) and Hanks and Hill (1980). However, most of these models cannot be used in a stochastic framework, that is, infiltration events triggered by stochastic rainfall events. Among the few examples that can be used in such a framework is the model of Cordova and Bras (1981), based on earlier results of Eagleson (1978). This model assumes instantaneous events of storms and irrigation, and calculates infiltration volume per event, using an approximated solution of the classical Philip equation (1957). The other three factors, C_p, P and ET, are the function of the moisture content and other parameters (soil, groundwater level, crop). By the help of this model, statistics (mean and variance) of Θ for the next stage can be analytically calculated, or the whole distribution simulated.

The other part of the state transition function refers to the available supply:

$$v(t+1) = v(t) - I(t) \tag{8.4}$$

(4) The *output* has also two parts. Within-the-stage output includes the contribution $Y(t)$ of irrigation $I(t)$ to the growth of the crop, and the cost of irrigation. Output for the whole growing season may be the net agricultural benefit, B. Note that both $Y_t(t)$ and B are considered as random variables due to the random rainfall contribution. Further outputs such as water-loss to groundwater (P) or groundwater pollution can be also considered in the same framework.

(5) The *output function* to be maximized, calculates the expected agricultural net benefit $E(B)$.

$$E(B) = E\left[\sum_{t+1}^{T} R(t)\right] - P_c \tag{8.5}$$

where T = the number of stages, P_c = production costs apart from irrigation costs, and:

$$R(t) = p \cdot Y_t(t) - C_t[I(t)] \tag{8.6}$$

where p = unit price of crop, C_t = irrigation costs. The total yield:

$$Y = \sum_{t=1}^{T} Y_t(t) \tag{8.7}$$

can be calculated by several "crop" models mostly as a function of actual evapotranspiration during the stages:

$$Y = Y_m H(ET_{ot}, ET_t) \qquad t = 1,\ldots,T \tag{8.8}$$

where the function H may represent additive or multiplicative crop models (Blank, 1975; Hanks and Hill, 1980) and Y_m is the maximum yield.

If an additive model is used as in Bras and Cordova (1981) and equations (8.7) and (8.8) are substituted into equation (8.5) the final objective function will contain the expectation of additive terms of ET_{ot}, ET_t, and C_t. In this case stochastic dynamic programming can be used in order to find the irrigation decisions: $I_1, I_2, \ldots, I_t, \ldots, I_T$ maximizing the expected agricultural net benefit.

8.2.3. Model evaluation

The decision-aid resulting from the use of the scheduling model shown is a number of tables. Each table indicates the optimal irrigation application as a function of a limited water supply and of the soil moisture content for a particular stage of the irrigation season. Some numerical results after Bras and Cordova (1981) are given in *Figure 8.3*, where annual net benefits for stochastic and deterministic control is represented by an operating policy of irrigating in each stage up to field capacity. The annual net benefits were calculated by simulating 27 years of hourly rainfall data for corn. The conclusion is that the expected net benefit increases and its variability is reduced when using stochastic control.

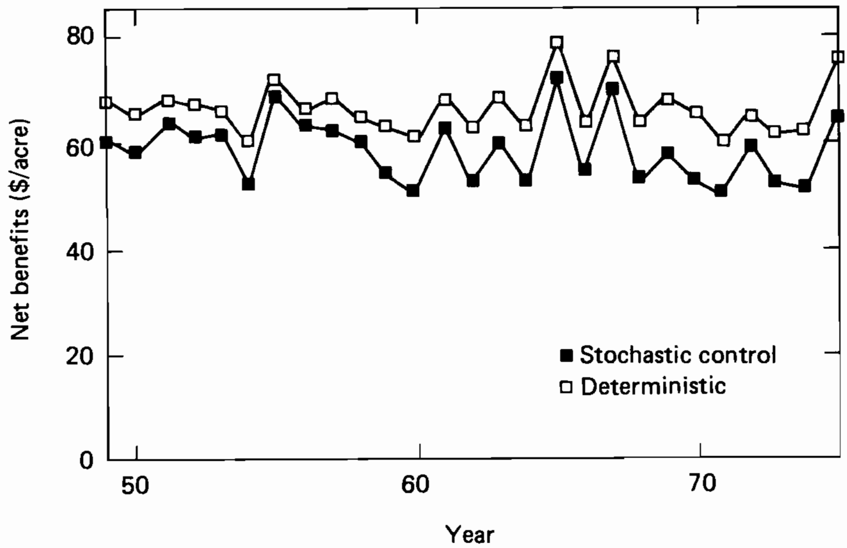

Figure 8.3. Yearly benefits of irrigation policy.

As for any stochastic DP model, it is not possible to develop a program package for general use. However, the principles and elements of the model are common; thus a programmer familiar with DP is always able to prepare a computer program reflecting the specific features of the very application.

8.3. Aquifer Management

8.3.1. Introduction

The policy issue concerns the conjunctive planning and operation of water and mineral resources extraction in a given region under a dual set of objectives; namely, an economic set and an environmental one.

The economic objective may be characterized by indicators such as present worth of cost, net benefit, and benefit–cost ratio.

Over a region of 10 000 km^2 in Hungary, large-scale surface and underground deposits of bauxite are to be extracted at several existing and planned mine sites which are under the water hazard caused by a regional karstic aquifer. To remedy this situation, one may either lower locally the piezometric level or decrease local transmissivity by grouting, or use a combination of both measures.

Given the ore processing capacity and contracted quantities over a 20 year horizon, the economic objective is to allocate production rates to existing and potential mines so as to minimize total discounted cost.

The environmental objective is to maintain the karstic water system in a state no worse than the present one. Possible actions to fulfil the environmental objectives are the control of mining withdrawals, grouting, or artificial recharge.

The environmental state may correspond to the flow and quality of springs, thermal baths, and water wells in the region. There is, however, substantial controversy about the numerical values of flow and quality parameters that represent a "sound regional environment". To circumvent this difficulty, a fuzzy set approach is used (Bellman and Zadeh, 1970; Zadeh *et al.*, 1975).

Development, solution, and application of the model can be found in detail in Bogardi *et al.* (1983).

8.3.2. Model structure

The planning horizon is divided into T discrete stages, and a two-tier analysis is performed at each stage. In the first tier, the $(N+1)$ elements of the objective function vector are defined as one overall cost function, and one environmental objective function at each of N relevant control sites. In the second tier, the N environmental objective functions combine into one single fuzzy set membership function.

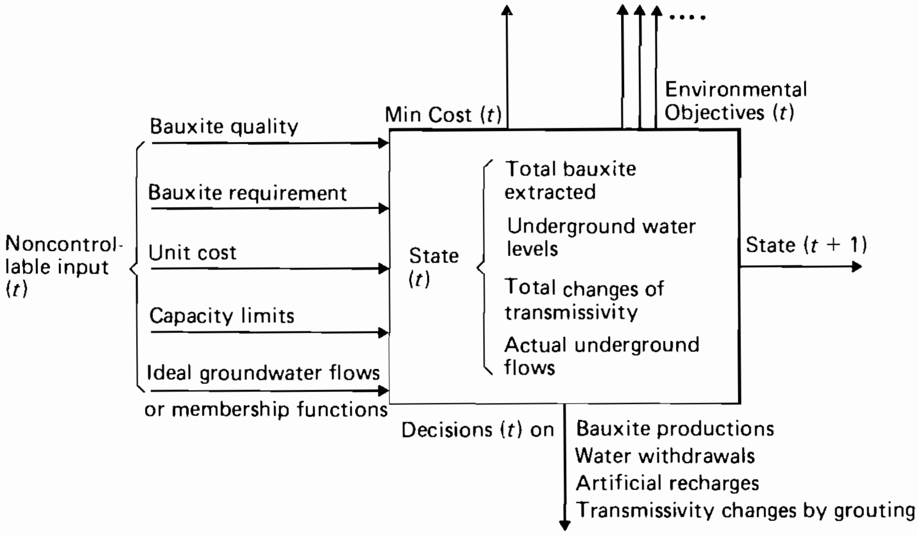

Figure 8.4. **Aquifer management model.**

Elements of the model as illustrated in *Figure 8.4* are not introduced.

For a planning horizon of T years, the model includes: M mines: $i = 1,\ldots,M$; K recharge sites: $k = 1,\ldots,K$; J grouting sites: $j = 1,\ldots,J$; N control points (springs, wells): $n = 1,\ldots,N$.

(1) The *input* comprises: $x^*(i)$: total ore available at mine i. *Unit costs* as $u(m,i,t)$, mining production; $u(q,i,t)$, water control; $u(r,k,t)$, artificial recharge; $u(g,j,t)$, reduction of transmissivity; $d(t)$, discount factor. *Decision matrices* $X = [x(i,t)]$, annual bauxite extraction; $A = [q(i,t)]$, annual dewatering; $R = [r(k,t)]$, annual recharge; $G = [g(j,t)]$, relative transmissivity change by grouting; a number of capacity limits, requirements, and mineral quantities.

(2) The *state* includes: $X(i,t)$, total bauxite extracted until t; $Z(i,t)$, piezometric level of aquifer; $G(j,t)$, total relative change of transmissivity until t; $H(n,t)$, spring flow at control point n.

(3) The *state transition function*. The state transition function has four components corresponding to the four components of the state. The first component describes the bauxite mining:

$$X(i,t+1) = X(i,t) + x(i,t+1) \tag{8.9}$$

The second and third components of the state transition function correspond, respectively, to piezometric level and spring flows, stemming from the aquifer. Both can be calculated by numerical solution of a partial differential equation system (Szilágyi *et al.*, 1978). Finally, the fourth component of the state transition function describes transmissivity change as:

$$G(j,t+1) = G(j,t) + g(j,t+1) \tag{8.10}$$

(4) The *output* to be minimized consists of $N+1$ elements. One is the total discounted cost and the other N are the environmental objectives expressed in a fuzzy way. That is, the N environmental objectives are expressed by their respective membership function, which are defined as follows: the flow of a spring $H(n,t)$ will be a full member of the fuzzy set of sound environmental condition, with grade 1.0, if experts consider this flow $H(n,t)$ to be excellent. On the other end of the scale, the membership grade is zero if $H(n,t)$ is unsatisfactory (*Figure 8.5*). In addition, there are several other outputs (total bauxite extracted, rates of production, withdrawal, recharge, grouting) which are kept as constraints.

(5) The *output function vector* to be minimized has thus also $N+1$ elements. The economic objective:

$$F(X,O) = \sum_{t=1}^{T} d(t) \left\{ \sum_{i=1}^{M} [u(m,i,t)x(i,t) + u(q,i,t)q(i,t) + \sum_{k=1}^{K} [u(r,k,t)r(k,t)] + \sum_{j=1}^{J} u(g,j,t)g(j,t) \right\} \tag{8.11}$$

The environmental output functions, the membership functions are piecewise linear, with given thresholds $E_0(n,t)$ and $E_1(n,t)$:

$$\mu(n,t) = \begin{cases} 0 & \text{if } H(n,t) < E_0(n,t) \\ \dfrac{H(n,t) - E_0(n,t)}{E_1(n,t) - E_0(n,t)} & \text{if } E_0(n,t) \le H(n,t) < E_1(n,t) \\ 1 & \text{if } E_1(n,t) < H(n,t) \end{cases} \tag{8.12}$$

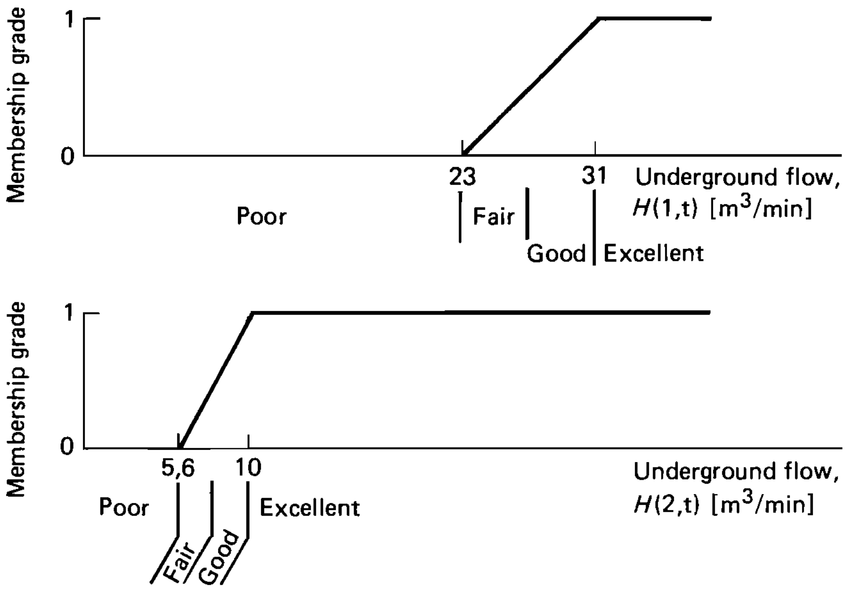

Figure 8.5. Results of aquifer analyses.

Let the set of environmentally feasible solutions (EFS) be defined as:

$$(EFS) = \prod_{n=1}^{N} \prod_{t=1}^{T} [\text{feasible set of } H(n,t)] \tag{8.13}$$

The corresponding membership function is:

$$\mu(EFS) = \min_{n=1,\ldots,N} \min_{t=1,\ldots,T} \mu(n,t) \tag{8.14}$$

The multiobjective optimization problem has now been reduced to two objectives:

Min $F(X,Q)$ as defined in equation (8.11) (8.15)

Max $\mu(EFS)$ as defined in equations (8.13) and (8.14)

The constraints consist of physical limits (capacity, ore availability), production requirements (quantity and quality), and the state transition function.

8.3.3. Model evaluation

In its original form the model is a nonlinear dynamic multiobjective problem. For the real-life application the model has been linearized, so that a two-objective dynamic linear program is obtained.

The use of compromise programming (Zeleny, 1973; Duckstein and Opricovic, 1980) then leads to a standard linear dynamic programming formulation.

The results presented herewith correspond to a simplified version of the case study which has actually contributed to finding bauxite extraction scheme in the Bakony region of Hungary.

The planning horizon of T = 20 years is divided into four stages, t = 1,2,3,4. There are: M = 3 mines, including mine 3 under water hazard; K = 1 recharge site; J = 1 grouting site; N = 2 spring site. Regional hydrological input and aquifer properties are taken after Schmieder (1970). Total quantity and average quality of the bauxite resource are given in *Table 8.1*; bauxite requirements, capacity limits, and ideal groundwater flow are given in *Table 8.2*.

Table 8.1. Amount and quality of bauxite available.

Amount (10^3 tonnes)	Al_2O_3	S_iO_2
3000	53.7	4.9
7300	49.8	9.1
8100	52.1	5.8

Table 8.2. Requirements and capacity limits.

Stage t:	*1*	*2*	*3*	*4*
Bauxite requirements (10^3t)				
Amount	2500	3800	4400	4700
Module	7.2	7.0	6.8	7.0
Mining capacity limits (10^3t) :				
Mine 1	1000	1600	1600	1600
Mine 2	900	2000	2900	2900
Mine 3	1800	2400	3200	3200
Minewater withdrawal capacity limits (Mine 1 in m^3/min)	310	350	400	400
Recharge capacity limits (m^3/min)	10	30	50	50
Grouting capacity limits in relative transmissivity	0.4	0.4	0.4	0.4

Table 8.3. Solution without grouting, with weights (1,0).

Stage t:	*1*	*2*	*3*	*4*
$x(1,t)$	1000	1600	400	0
$x(2,t)$	872	1975	2325	2128
$x(3,t)$	628	225	1675	2572
$q(3,t)$	0	57	400	400
$r(1,t)$	0	0	0	0
$H(1,t)$	36	33	19	19
$H(2,t)$	15	13.6	5	5

Membership grade: 0
Total discounted cost: 3058×10^6 ft.

Table 8.4. Solution without grouting, with weights (0.5, 0.5).

Stage t:	*1*	*2*	*3*	*4*
$x(1,t)$	1000	1600	400	0
$x(2,t)$	900	2000	2272	2128
$x(3,t)$	600	200	1728	2572
$q(3,t)$	0	220	295	366
$r(1,t)$	0	0	26	50
$H(1,t)$	36	27	27	27
$H(2,t)$	15	9.5	8.6	7.8

Membership grade: 0.49.
Total discounted costs: 3132×10^6 ft.

Table 8.5. Solution without grouting with weights (0.0, 1.0) and (0.2, 0.8).

Stage t:	*1*	*2*	*3*	*4*
$x(1,t)$	1000	1600	400	0
$x(2,t)$	795	2000	2377	2128
$x(3,t)$	705	200	1623	2572
$q(3,t)$	129	217	275	275
$r(l,t)$	0	30	50	50
$H(l,t)$	31	31	31	31
$H(2,t)$	11.3	10.7	10.3	10.3

Membership grade: 1.0.
Total discounted costs: 3218×10^6 ft.

Further data and the results are given in Bogárdi *et al.* (1983); here three cases are illustrated.

Table 8.3 shows the results of purely economic optimization, without grouting: the total discounted cost is minimum, but environmental flows become unacceptable low at stages $t = 3$ and 4, hence the membership grade is zero.

Weights refer to the terms in the L_p -metric of compromise programming.

With equal weighting, in *Table 8.4*, environmental flows are only fair to good, and overall membership grade is 0.49. Further change in weights favoring environmental flows; for example, the set (0.2, 0.8) leads to a membership grade of 1.0 in *Table 8.5*.

The use of the original nonlinear model requires detailed data, mathematical skill for the algorithm and programming, as well as considerable computer time. On the other hand, the linearized model applies generally available data and LP. Thus a trade-off can be found between the accuracy requirement and available resources such as data, analyst, computer. Our experience has shown that the linearized model satisfies the conditions of real-life application.

8.4. Marchfeld Region Conflict Resolution

8.4.1. Introduction

The purpose of this section is to describe the evaluation of long-range development plans for the Marchfeld region in Austria. This area is situated in the east of Austria and is the Northern part of the Viennese basin. The region has 57 000 inhabitants and an area of approximately 950 km^2. Agricultural production is highly developed and is an important part of the regional economy.

Water for irrigation is supplied by exploiting an extended aquifer which also serves to satisfy industrial and domestic water requirements. The increasing utilization of the main regional water resource has led to subsequent depletion to such an extent that the future supply is highly endangered.

Environmental aspects are of importance since a high load of agricultural, industrial, and domestic waste water is impairing the quality of both surface and groundwater. In recent decades, forested areas have suffered from agricultural activities, river regulation, drainage and lowering of groundwater table. The forests bordering the river reaches (so-called "Auwald") of the March and Danube are especially endangered.

A two-step approach is selected to assist in regional water resource planning. The first step is a preliminary screening of a discrete number of alternatives by the application of a comprehensive cost–effectiveness approach. For the subsequent approach, a dynamic systems model for both management and physical aspects is developed around the set of preferred alternatives. This systems approach aims at determining such a development of the activities that a compromise among conflicting objectives is achieved throughout the planning horizon. The first approach can be found in Nachtnebel *et al.* (1981), while the dynamic model is described in Nachtnebel *et al.* (1982, 1983).

8.4.2. Model structure

The dynamic model consists of the following elements:

(1) *Input*

These variables $I(t)$ modifying members of the state set of the system are separated into uncontrollable variables such as growth coefficient of population, development of industrial production, freshwater requirements, economic, and hydrologic variables; and decision variables related to the activities: d_1, increment in irrigation canal capacity; d_2, land use; d_3, increment in waste water treatment capacity; d_4, recirculation systems in the industrial sector.

(2) *State*

The following fundamental state variables $S_i(t)$ are defined: S_1, irrigation capacity of the existing canal network; S_2, average depth of groundwater table; S_3, average nitrate concentration in groundwater; S_4, average BOD concentration in surface water (BOD stands here for the 5-day biochemical oxygen demand).

(3) *State Transition Functions*

For each state variable a corresponding state transition function can be developed using simple mass balance equations for the groundwater system, the nitrate concentrations and the BOD load. As an example, the nitrate concentration rates may be expressed as:

$$S_3(t+1) = \varphi[S_1(t), S_2(t), S_3(t), d_1(t), d_2(t), d_3(t), NI, NO] \quad (8.16)$$

NI and *NO* stand for natural nitrogen input and output.

(4) *Output*

The output consists of five elements: R_1, costs of industrial water supply; R_2, net benefit from agricultural production: R_3, environmental consequences related to the depth of the groundwater table; R_4, environmental consequences related to groundwater pollution (nitrate concentration); R_5, environmental consequences related to surface water quality (BOD load).

(5) *Output Functions*

The output function $R_1(t)$ is the sum of industrial water supply costs C plus economic losses L caused by water shortage:

$$R_1(t) = C\, W_1(t) + L[r(t) - W_1(t)] \tag{8.17}$$

r is the industrial freshwater requirement and W_1 is the available amount. On the basis of data given in Nachtnebel *et al.* (1981) the output function R_2 can also be calculated, which involves gross benefits and costs of the agricultural production.

Environmental consequences $R_3(t)$, $R_4(t)$, and $R_5(t)$ are expressed by their respective membership function. In our problem they indicate the degree of relevancy of a certain environmental indicator for a sound environment.

8.4.3. Model evaluation

A large number of possible development alternatives is generated dependent on the sequence and combination of decisions at various time steps. The objectives are:

(1) The sum of discounted industrial water supply costs should be minimized: $\min \sum_{t=0}^{T} D_t R_1(t)$

(2) The sum of discounted agricultural net benefits should be maximized: $\max \sum_{t=0}^{T} D_t R_2(t)$

(3) Each set of membership functions $R_3(t)$, $R_4(t)$, and $R_5(t)$ should be maximized: $\max \sum_{t=0}^{T} R_3(t)$, etc.

Because no set of decisions d_i optimizes simultaneously all the objectives defined above, a multiobjective decision technique is necessary to select a compromise set.

Since simple balance equations and loss functions L are applied to describe the system, the state transition function and the output functions can be linearized. In that case linear compromise programming can be applied as shown in Section 8.3.

8.5. Comparative Evaluation of the Models

Table 8.6 evaluates the three economic–ecological water resources models.

Table 8.6. Model evaluation and characterization.

	Models		
Evaluation criteria	*Irrigation scheduling*	*Aquifer management*	*Marchfeld conflict resolution*
Type of problem	Operation	Planning	Planning
Scale of problem	Local	Regional	Regional
Type of model	Decision Optimization Dynamic Stochastic Discrete	Decision Optimization Multiobjective Dynamic Continuous	Decision Optimization Multiobjective Dynamic Discrete
Level of application	Year-to-year basis	Actual decision-aid	Interagency recommendation
Cost of model development	Relatively high	Low	Low
Model availability	No general form	Some elements routinely used	Some elements routinely used

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CHAPTER 9

Water Quality – Economic Modeling

D.P. Loucks

9.1. Introduction

Much of what we who live do, including what we need to do to stay alive, consumes products and generates by-products. Some of these by-products are of little or no use, and hence are considered as wastes to be discarded. They are discarded into our environment. Many liquid forms of these discarded waste products are discharged into water bodies such as streams, rivers, lakes, and estuaries, and eventually into groundwater aquifers, seas and oceans. This can result, and has resulted, in reduced benefits from alternative uses of these water bodies. Some waterborne wastes may also enter food chains and hence end up, temporarily, in living organisms. This can lead, and has led, to potential or actual health problems as well. Concern has grown over the ecological, economic, and public health problems resulting from the disposal of waste products into the environment. When these problems become serious, as they can when too much of certain types of liquid wastes are discharged into too small a receiving water body, they motivate the need to identify policies that can balance the cost of reducing the problems with the damage those problems cause. The implementation of any policy to reduce the imbalance is made difficult by the fact that those who discharge wastes are not usually those who suffer the damages. Furthermore, major components of the damages are difficult to quantify.

Since the mid-1920s, mathematical models have been used to help predict the effects of waste discharges on the quality of water bodies.

For various types of receiving waters, there now exists a wide range of mathematical modeling approaches for water quality impact prediction (Orlob, 1983). These modeling approaches range from those used to study relatively simple single-constituent, conservative or first-order decay, fully-mixed, steady-state, zero- or one-dimensional systems that receive wastes from only point sources, to those developed to help understand two- and three-dimensional dynamic–multiconstituent systems that include aquatic ecosystem organisms and account for non-point as well as point source waste loadings. Interestingly, as modelers attempt to capture more realism and detail by building more complex models, the results are not always more reliable. Comprehensiveness and complexity have not necessarily led to increased accuracy. Our current ability to model aquatic water quality systems far exceeds our knowledge about such systems.

In spite of our relative ignorance about water quality constituent interactions between each other, and with the physical hydrologic system, water quality management decisions are made. The cost of these decisions, often involving wastewater treatment, land use restrictions, l;ow-flow augmentation, and the like, can be substantial. Hence in the early 1960s, some modelers began to look for ways of defining cost-effective solutions to complex water quality management problems. This involves linking water quality prediction models to models that incorporate management alternatives and their costs. Early approaches, and many of those in general use today, have attempted to define minimum cost solutions for meeting predefined water quality standards, sometimes considering equity or cost–distribution issues.

Since the early 1960s, water quality management models have become increasingly sophisticated and complex. There have even been serious attempts by skilled analysts to quantify water quality benefits or damages and to couple fairly detailed water quality prediction models to models of regional development and other economic and waste production activities (Dorfman *et al.*, 1972; Kneese and Bower, 1979). Perhaps because of the difficulties, and costs, in producing, calibrating, and verifying such comprehensive models, let alone making them understandable and useful to those involved in political decision making, there has developed a prevailing view that, at least for management, relatively simple models are preferred.

This short chapter will limit its focus to water quality–economic management models: models designed to identify particular feasible combinations of economic and water quality variable values. Entire books have been devoted to this subject (Biswas, 1981; Dorfman *et al.*, 1972; Orlob, 1983; Rinaldi *et al.*, 1979; Thomann, 1972). The particular models chosen for comparison in this short chapter are only a small sampling of the many models available for various types of water bodies.

Any cursory review of the literature in this subject will show that a wide range of water quality constituent interactions, water bodies, and management alternatives have been modeled using a wide range of modeling techniques. Most models are based on conservation of mass and energy. A few are based on statistical regression methods. In all cases, there exists considerable ecologic and economic uncertainty, even beyond that which a few have tried to define through the use of stochastic models. This uncertainty results from a deficiency of data (affecting model calibration and verification) and a deficiency in knowledge (affecting model structure). Both economic and water quality variables can be incorporated within a single model, to permit the simultaneous consideration of both types of variables, or the economic submodel can be separated from the water quality prediction submodel, requiring iterative sequential alternative solutions of each submodel.

The majority of water quality–economic models have been designed for planning, as opposed to design and operation. Planning models are usually relatively simplified steady-state models that permit the examination of a wide variety of water quality control options. For these analyses, most planning models are deterministic, and assume some "design" or "fixed" environmental and hydrologic conditions. This is the case for all but one of the models selected for comparison in this chapter.

This chapter will continue with a brief enumeration of some of the common policy issues addressed by water quality–economic models. Following this, several models that have been used for planning the management of water quality in rivers, lakes and estuaries will be reviewed and compared. The chapter concludes with some comments concerning model effectiveness in practice and the need for further research both in water quality prediction and in water quality management planning.

9.2. Water Quality – Economic Policy Issues

Those involved in activities that adversely impact regional water quality are often required to identify measures that can reduce these adverse impacts. Any evaluation of alternative water quality improvement measures involves estimates of total economic costs and their distribution among those who pay for such measures. Costs and their distribution are pervasive policy issues. Most water quality–economic models have been designed to identify cost-effective control alternatives that satisfy distributional and water quality standards. Alternatively they can be used to indicate the increase in water quality possible given the amount and distribution of available funds that can be allocated to water quality improvement.

The difficulties in implementing cost-effective solutions are those of monitoring, enforcement, and administration. Deciding who is to pay how much is not a trivial exercise. It is thus usually expedient to require the same for each potential polluter, e.g., imposing the same level of waste treatment for everyone, regardless of the extra cost. The reasons for such a policy are obvious. The merits are still being debated.

Other policy issues include the establishment of water quality standards themselves, and how they should be defined. The issue is not only one of selecting the type and extent of various constituent concentrations that are permissible in a water body, but also defining the allowable extent and duration of failure. Other policy issues include the establishment of appropriate water quality control measures, for both point and nonpoint waste sources; the estimation of water quality improvement benefits, and especially identifying who benefits; and model reliability in situations where data are meager with respect to future waste loadings, hydrology, water quality impacts, costs, interest rates, and pollution control technology.

Some continue to debate the proper role of water quality–economic models in the policymaking process. This too is a policy issue. If such models are to contribute more effectively to any debate over what to do or what policy to implement, increased attention will have to be given to the interaction and communication between modelers, their models, and those who can benefit from the information obtained. Improved human–model–computer interaction can facilitate the effective use of water quality–economic models for policy exploration, analysis, and synthesis.

9.3. Model Descriptions and Evaluations

In the following sections, several models will be presented that represent some typical approaches to water quality–economic modeling in streams and rivers, lakes and estuaries. Research is continuing towards extending these approaches for surface water quality management to groundwater quality management as well. This discussion will be confined to surface water quality–economic modeling, and the impact this modeling effort has had in the specific situations examined. Additional information on topics not discussed here are contained in the references listed at the end of the chapter.

9.4. River Models: St. John, Trent, and Rhine Rivers

9.4.1. Introduction

The St. John River in the USA and Canada flows some 700 km from the State of Maine into the Province of New Brunswick, and eventually into the Bay of Fundy. Quality problems stem from excessive organic wastes from potato and pulp-and-paper industries. The River Trent in England begins north of Stoke-on-Trent and flows west some 274 km to the Humber Estuary. It is heavily used for municipal and industrial wastewater disposal, but is also a potential source of public water supply, recreation, and fish habitat. The Rhine River in West Germany is one of the most heavily polluted large rivers in Europe, carrying wastes from France and West Germany through the Netherlands to the North Sea. The water is extensively used for drinking water, navigation, and recreation. The portion modeled includes 450 km from Mannheim and Ludwigshafen to the Dutch–German border. The models developed for these three rivers were designed to help predict water quality and costs in response to various water quality management alternatives.

The St. John River models were developed by H.G. Acres, Ltd. and Meta Systems, Inc. They involved both a prescriptive steady-state optimization model and a descriptive time-varying simulation model for both biochemical oxygen demand (BOD) and dissolved oxygen (DO) prediction and management. The Canadian model development was closely supervised by Environment Canada. The US model was developed for use by the Northern Maine Regional Planning Commission and the US Environmental Protection Agency (Biswas, 1981).

The development of the Trent model was undertaken by a group within the Water Resources Board. The model has been used by the Trent River Authority, the Water Research Centre, and the Water Resources Board. This model included 16 constituents and considered the trade-off between water treatment costs and wastewater treatment costs. It also included estimates of fishing and recreation benefits (Newsome *et al.*, 1972).

Two Rhine models were developed by a group of analysts from the Nuclear Research Center, Karlsruhe, and at the International Institute for Applied Systems Analysis in Laxenburg, Austria. These models included components for predicting BOD–DO, bacteria, and protozoa concentrations. Model extensions included artificial instream aeration, scheduling treatment plant construction over time, and multiple economic and environmental objectives (Rinaldi *et al.*, 1979).

9.4.2. Model structures

Each river model can be viewed as containing at least four main components, as illustrated in *Figure 9.1*. Each model differs, however, in its definition of some of those four components. The St. John, Trent, and one of the two Rhine models assume mostly first-order reactions

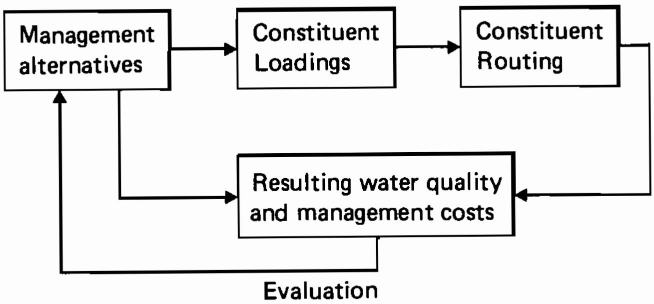

Figure 9.1. General schematic of river water quality management models.

for the prediction of the concentration C_j^k of a constituent k at various quality sites j resulting from the vector $\mathbf{W}_i$ of waste from discharges at each upstream site i:

$$C_j^k = \sum_i f_{ij}^k[\mathbf{W}_i\mathbf{X}_i),Q_j] \qquad (9.1)$$

where: $\mathbf{W}_i(\cdot)$ = a vector or waste discharges at site i, that are dependent on $\mathbf{X}_i$ = vector of waste treatment efficiencies for each constituent at site i, $f_{ij}^k(\cdot)$ = the transfer or predictive function for constituent k at site j, resulting from the waste discharges at site i, and Q_j = river flow at site j.

Higher order predictive relationships, contained in the Trent and in one of the Rhine models, involve more complex nonlinear predictive functions,

$$C_j^k = F_j^k[\mathbf{W}_i(\mathbf{X}_i):\forall i;Q_j] \qquad (9.2)$$

The management models for the St. John had the general form:

$$\text{Minimize } \sum_i \text{Cost}_i(\mathbf{X}_i) \qquad (9.3)$$

Subject to: $C_j^k \leq$ maximum allowable concentration at site j for each constituent k

The Canadian St. John model considered both carbonaceous and nitrogeneous BOD, and DO quality standards. The management vector $\bar{X}_i$ included wastewater treatment efficiency at major point sources of wastewater discharge.

The US St. John model defined biomass potential as a linear combination of carbonaceous BOD, total reactive nitrogen, and phosphorus. Quality standards were applied to this biomass potential. The model explicitly considered capital and operating costs as functions of waste removal efficiency and wastewater flow. It also considered budget limitations and a two-period staged implementation program.

The Trent model included a river model for quality prediction and both water and wastewater treatment costs for specified river water abstractions and wastewater discharges. An expanded river model called an allocation model examined additional alternatives for satisfying water quantity and quality abstraction demands. The objective of the management model was to minimize the total annual cash flow for wastewater treatment, $\text{Cost}_i^E(X_i, E_i)$, which is dependent on removal efficiency X and effluent flow E; plus the total annual cash flow for water treatment, $\text{Cost}_j^W(C_j, Q_j)$, which is dependent on the stream flow quantity Q_j and quality C_j; less the water quality-related benefits, $B_j(C_j)$; at all sites i and j.

$$\text{Minimize} \sum_i \text{Cost}_i^E(X_i, E_i) + \sum_j [Cost_j^W(C_j, Q_j) - B_j(C_j)] \qquad (9.4)$$

Sixteen different constituents were considered, including temperature.

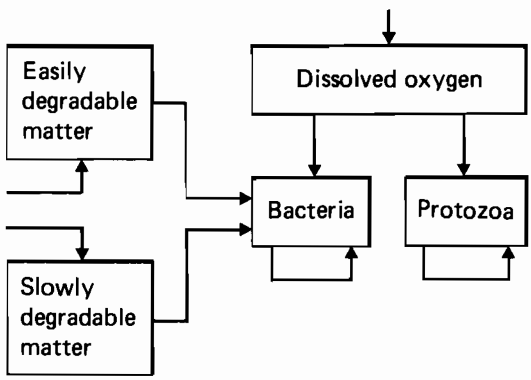

Figure 9.2. Schematic of Rhine river ecological model.

Both Rhine models (*Figure 9.2*) considered easily-degradable organic matter, slowly degradable organic matter, and dissolved oxygen. The ecological model also included bacteria and protozoa biomass as well as point and nonpoint wastewater loading.

These steady-state river water quality models were solved using linear, nonlinear and dynamic programming methods coupled with parameter estimation techniques. An unsteady time-varying water quality simulation model was developed and used to simulate the results derived from the St. John linear programming water quality management model. Discrete dynamic programming was used to solve the Trent model. Linear, nonlinear, and dynamic programming methods were used for the Rhine models.

9.4.3. Model evaluations

In all cases, the modeling efforts were considered as valuable learning experiences. The Canadian St. John model was transferred to, and later modified by, the Department of Environment, but in both cases little use is currently being made of either the Canadian or US St. John models. Cost-effective solutions have been of little interest, in spite of the fact that substantial cost savings could result if such policies could be implemented and administered. The model results helped highlight the need for improved financial, administrative, and energy cost data as well as the need for ways of estimating water quality benefits and their distribution.

The Trent model has had a little more success. Some of those who were involved in its construction are currently involved in its continued use. The exercise has increased the understanding of water quantity–quality–cost relationships in the River Trent, although the model's accuracy based on average flows is questioned. The model has been usefully applied to other rivers in England.

The Rhine modeling exercise was just that. No client seems to have been involved. As a research project, it was extremely thorough and innovative. Artificial instream aeration and time-sequencing of increased treatment capacity, all in a multiobjective framework, were considered. A major difficulty in the application of the ecological model will be obtaining the data necessary for parameter identification.

9.5. Lake Models: Lake Balaton and Neusiedler See

9.5.1. Introduction

Lake Balaton and Neusiedler See are two very shallow yet large lakes in Hungary and Austria, respectively, that are both subjected to heavy loadings of nutrients. These loadings come from increasing agricultural, tourist, and industrial activities in their respective watersheds. Substantial loading comes from the wind as well as from run-off. Increasing lake eutrophication has been the result. The policy issue in both cases is how to increase tourism and agricultural production and still preserve lake water quality which affects especially tourism.

9.5.2. Model structures

A series of models, ranging in detail and scope and purpose, have been developed for Lake Balaton by modelers from the National Water Authority and Academy of Sciences in Hungary, and from the International Institute for Applied Systems Analysis in Austria (Somlyody, 1982). The decomposition and aggregation modeling approach for Lake Balaton is shown in *Figure 9.3*.

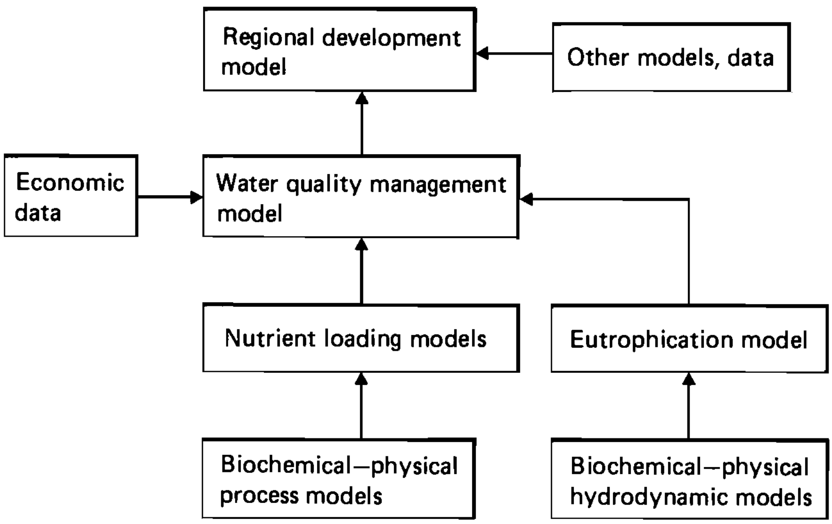

Figure 9.3. Schematic of Lake Balaton models.

This pyramid of models was designed to discover the relative importance of various processes in the entire eutrophication – water quality management problem without making prior assumptions regarding the need for model component complexity.

For water quality management, a linear stochastic optimization model was developed. The objective was to find control strategies that maximized a weighted combination of the mean and standard deviation of water quality improvement ΔC, given a prespecified level of funding.

For various lake sections j, weights w_j and reductions in random maximum chlorophyll a concentrations ΔC, the objective was to:

$$\text{Maximize} \sum_j w_j E[\Delta C_j] - \delta[\Delta C_j] \qquad (9.5)$$

$$\text{Subject to: } \Delta C_j = j[\mathbf{W},\mathbf{X},\mathbf{K}]$$

$$\text{Cost} = C_1(\mathbf{X}) + C_2(\mathbf{K}) \leq \text{maximum cost}$$

where: $f_j(\cdot)$ represents the decrease in chlorophyll a resulting from the impact of additional wastewater treatment or diversion **X**, or prereservoir storage **K**, on the total loading **W**. $C_1(\mathbf{X})$ and $C_2(\mathbf{K})$ are the present value of the costs associated with these control alternatives.

The Neusiedler See Interactive Water Quality Management–Regional Development model was developed at the International Institute for Applied Systems Analysis (Fedra, 1982). The model was designed for interactive sunylation of any particular set of control strategies one wanted to examine or evaluate. Computer graphics was used to aid in the human–model–computer interaction. The lake quality simulation model contained many of the equations of constituent interactions that were used in the Lake Balaton and other shallow lake eutrophication models. In comparison with the Lake Balaton model, the interactive Neusiedler See model included considerably more detail on the activities that produced nutrient loadings, and the impact that lake water quality had on those activities.

The model's general form is illustrated in *Figure 9.4*.

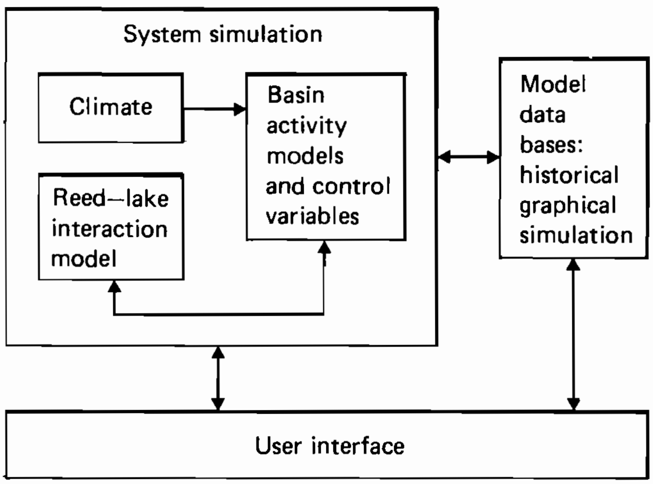

Figure 9.4. Schematic of Neusiedler See model.

9.5.3. Model evaluations

The Lake Balaton Models were developed over a period of several years by some 30 scientists from Hungary and 20 scientists from eight other nations. It was a multidisciplinary effort, and it has resulted in a series of models now being used in Hungary to examine alternative control options. In addition, several methodological achievements in model development, parameter estimation, model structure identification, sensitivity, and uncertainty analyses and in coupling hydrodynamic, biochemical, and economic modeling, have occurred. Current decisions being taken in Hungary are in accord with the modeling results, but it seems clear that these decisions have not been made without some resistance by those whose objectives differ. This modeling effort has been a major one and is currently having a significant impact.

The effectiveness of the Neusiedler See model in helping to identify regional development policies is not clear at this time. However, the effort clearly contributed to the art of interactive modeling and model use. Viewers and users of the modeling system always preferred graphical and symbolic displays to alphanumeric displays. They also seemed to prefer relational information to absolute data, yet qualitative descriptions of water quality such as excellent, good, bad, and dangerous proved to be controversial. Participants of planning or policymaking processes clearly preferred to be involved when and where any subjective judgment was required.

9.6. Estuary Models: Delaware Estuary

9.6.1. Introduction

Perhaps the first application of any water quality–economic modeling was that initiated by the US Public Health Service in the mid-1960s on the Delaware Estuary in the USA. Later transferred to other government agencies, it became known as the Delaware Estuary Comprehensive Study (DECS). Its purpose was to test the feasibility of providing a benefit–cost approach to regional water pollution control (Ackerman *et al.*, 1974).

Policy issues focused on the deteriorating water quality conditions in the heavily industrialized estuary extending some 85 miles (150 km) south of Philadelphia, Pennsylvania to the Delaware Bay. Of particular concern were constituents (e.g., sediment, toxics, pathogens) that would affect the esthetics of the water body and the use of the estuary for recreation. BOD and DO were the water quality constituents selected for modeling, since these could be modeled more successfully at that time than could sediment, toxics, or pathogens. Once this decision was made, the policy issue resolved around what the average DO concentration should be in the most critical (lowest DO) section of the estuary.

About 10 years after the initiation of the DECS modeling effort, analysts at Resources for the Future (RFF) undertook another comprehensive modeling effort that included the estuary. Their policy objective was to see if one could capture within a regional residuals environmental quality management modeling framework the major production, discharge, transport, and political and economic as well as environmental impacts of all major residuals in the region. The Lower Delaware Valley was selected for an application of the model (Spofford *et al.*, 1976).

Issues addressed by the RFF analysts included assessing the importance of linking together in a single model gaseous, liquid, and solid residuals and multiple environmental (air, land, and water) media; identifying possible modeling, data, and computational problems; defining regional cost-effectiveness, and cost–distribution (equity) tradeoffs; and evaluating alternative residuals management strategies and impacts.

9.6.2. Model structures

The DECS model involved dividing the estuary into 30 segments, and completing a mass balance of the BOD and DO in each section. BOD loadings into each section were a function of the treatment at each waste source site. Modeled for each section were the processes of BOD discharge from waste sources, BOD and DO dispersion (due to water inflow and outflow) and diffusion (due to concentration differences), BOD decay that decreased the DO, and reaeration that increased the DO. Two finite difference equations were written for each estuary section, one for estimating the rate of change of BOD and another for estimating the rate of change of DO (Thomann, 1972).

The steady-state versions of these predictive equations were incorporated into a linear programming model for identifying cost-effective solutions similar to the BOD–DO river models discussed in Section 9.4.2. The DECS team also estimated recreational benefits as a function of DO, and obtained solutions that maximized estimated net water quality benefits.

The RFF modeling team also attempted to measure environmental quality benefits in monetary terms. They derived economic loss or damage functions associated with violations of environmental quality standards. The RFF model considered modifications in the production of water quality constituents, as well as the treatment or removal of those constituents. The model included, in addition to BOD and DO, nitrogen, phosphorus, bacteria, algae, zooplankton, and fish. It contained a nonlinear aquatic ecosystem model linked to water quality standards and to economic production and residuals generation models (*Figure 9.5*).

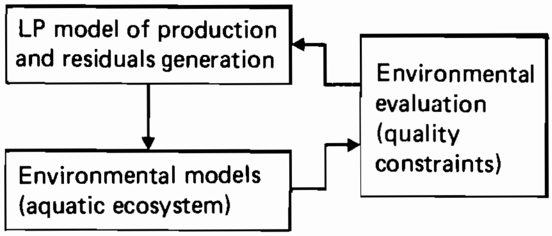

Figure 9.5. Schematic of RFF environmental model.

In mathematical terms, the RFF model can be summarized as:

$$\text{Minimize } [\mathbf{CX}_1 + P(\mathbf{X}_2, \mathbf{S})] \tag{9.6}$$

$$\text{Subject to: } \mathbf{AX} \geq \mathbf{B}$$

$$\mathbf{X} \geq \mathbf{0}$$

where: $\mathbf{X}_1$ is a vector of residuals production activities; $\mathbf{X}_2$ is a vector of residuals discharges; $\mathbf{X}$ is a vector of activity levels = $\{\mathbf{X}_1, \mathbf{X}_2\}$; $\mathbf{C}$ is a vector of cost coefficients for producing, handling, modifying, and disposing of residuals; $P(\cdot)$ is total penalty associated with quality standard violations; $\mathbf{S}$ is a vector of environmental quality standards; $\mathbf{AX} \geq \mathbf{B}$ is the set of equality and inequality constraints on minimum production levels; these constraints insure some level of economic activity since there is no benefit of production included in the model.

9.6.3. Model evaluations

The DECS modeling effort cost about $1.2 million and took over 4 years. What began as a test case for water quality management modeling became increasingly visible and political. Eventually the model results played a key role in focusing the debate over regional water quality management goals and costs. Cost-effective solutions were not considered very seriously because of administrative and equity problems. Zoned treatment (everyone in the same zone must implement the same wastewater treatment practices) failed because of the political influence some waste dischargers could exert to reduce their treatment requirements. Model precision was an issue debated in the courts. Finally, the whole modeling effort was accused of not focusing on the correct issues and water quality constituents. Nevertheless, the modeling effort did more to stimulate discussion and debate and further research in water quality modeling than any other similar such model at that time.

The RFF research effort was more ambitious in its scope, but was clearly more limited by its budget. While completed, its conclusions are tentative. In addressing the issues for which the model was designed to test, RFF has concluded that:

(1) The model was too complex, too expensive, too data-demanding, and yet not accurate enough to examine anything but a steady-state environment.
(2) Cost distribution is important and varies as alternative management strategies vary. Total cost, in this situation, was not sensitive to strategy changes.
(3) Aggregation is necessary, but a source of error.
(4) Incorporating nonlinear ecosystem models within management models substantially increases solution costs. Simpler models should be used for management analyses. Their solutions can then be checked using more detailed nonlinear simulation models.

9.7. Overall Model Evaluation and Effectiveness

A review of these and other modeling exercises for rivers, lakes, and estuaries suggests that the uncertainties and difficulties of accurately predicting both water quality and economic impacts of control measures stems largely from our lack of knowledge about the systems being modeled. Policymakers know this, and are also fully aware of the difficulties of implementing any water quality management strategy that is not considered equitable or politically supportable. Minimum-cost models, or ones that attempt to quantify and then maximize monetary benefits minus costs, have not been of interest to those in public agencies responsible for water quality management policies.

Yet models can help public policymakers. Models coupling water quality to economics, having appropriate degrees of complexity depending on the data base and issues being addressed, can be structured to assist in the exploration, analysis, synthesis, and evaluation of impacts of alternative strategies if they are designed to do this from the beginning. Less concern should be placed on how a model solution is going to be optimized than on how to make the model and its results, however obtained, more useful and meaningful to those who may benefit from them. Unless incentives change, this is not likely to be a major goal of university researchers involved in water quality–economic modeling.

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CHAPTER 10

Models for Outdoor Recreation

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10.1. Introduction

Outdoor recreation can be defined as the assembly of activities by human beings outside their residential areas when not at work. These activities are formal, i.e., within an organizational structure, or informal. Tourism is a general term for the whole of recreational activities and the adherent socioeconomic circumstances in space and time. In this chapter outdoor recreation and tourism are considered identical terms and will mostly be referred to simply as "recreation".

Recreation is economically important because it takes a large amount of private and public budgets. Particularly in the industrialized world, many people spend much of their leisure time away from their residential areas (holidays and weekend trips notably). Hotels, restaurants, and other recreation facilities generate incomes for millions of people. The public budget is also influenced by recreational activities; by revenues (taxes) or by expenditures (provision of recreation areas, but also maintenance of road networks). Even the profits of informal fishing or hunting by sportsmen have some economic importance.

For ages people have known that unlimited hunting or fishing may cause an ecological problem, depletion of stocks. Particularly since the 1930s (e.g., Bates, 1935) there has also been an increasing awareness that other recreational activities than killing animals can be detrimental to ecosystems and landscapes. High recreation densities may disturb breeding birds or may cause severe erosion. Creating new recreational facilities means use of space which was formerly occupied by plants and animals. In densely populated areas the remaining nature reserves become isolated islands with increased chance on local extinction of species.

The economic or ecological relevance of outdoor recreation issues cannot be analyzed properly without taking into account the geographical and sociopsychological aspects. The question *where* and *when* recreational activities take place require a spatial/temporal analysis of distributional features. The question *why* people spend their leisure time as they do is important because this may explain changes in participation or in spatial or temporal distribution. Most studies of recreation issues therefore include some of these aspects.

Recreation studies often use models for description, analysis, projection, or evaluation purposes. Statistical methods are also frequently used but these are not considered models here. Expertise in economic modeling of recreational benefits has been built up by Clawson (1959), followed by a vast number of authors. Ecological modeling of the impact of recreation started in the 1960s but is not yet done very frequently.

This chapter is mainly based on easily accessible publications on recreation in international journals such as *American Journal of Agricultural Economics, Ecological Modeling, Environment and Planning, Environmental Management, Journal of Environmental Economics and Management, Journal of Environmental Management, Land Economics, Regional Studies*. The chosen sample does certainly not represent all available kinds of models on the subject, nor all techniques used. However, most models that are of broader relevance than only specific (local) management purposes have been published this way.

The nest section analyzes outdoor recreation as a policy issue (cf. Chapter 1). Then possible use of models for such policy issues is indicated, including a short review of the results from the IvM-IIASA questionnaire (*see also* Chapter 4 and Appendix A). Next, some specific real world problems related to recreation are discussed with respect to using models to analyze or solve them. The chapter ends with a review of some integrated models and with an outlook as regards integrated recreation modeling.

10.2. Recreation Models for Policy Issues

10.2.1. Recreation policy issues

Within the broad field of outdoor recreation and tourism only those activities that relate to the more natural environments will be dealt with here. Thus the use of artefacts (recreation sites and parks, urban recreation areas) is excluded.

Economic–ecological policy issues regarding recreation focus on two specific aspects of the interface between economic life and the environment:

(1) Recreational exploitation of environmental resources. This exploitation includes both activities like hunting and fishing, and more general activities like hiking, camping, horse-riding, enjoying scenery, etc. (i.e., using the environment as a substrate for activities)
(2) Recreational disturbance and pollution of ecosystems. In this case something is added to the ecosystem: noise, waste, chemicals, etc.

Looking in greater detail at the classes of policy issues (*see* Chapter 1), recreation concerns several classes. This is indicated in *Figure 10.1*.

The policy issues in classes 1 to 6 may be studied in a monodisciplinary way. The issues in classes 7 to 9 are focal issues if the problem under consideration is to be viewed from different angles. Studies regarding these issues should have both a predictive and a prescriptive function (apart from analytical aspects such as dose–effect relationships or interaction levels between recreationists).

Most issues mentioned can be qualified as optimization problems. Dependent on the issues, single- or multi-objective optimization will be required. Next, if "sustainable use" is at issue (which may be the case in most classes) the time dimension becomes extremely important. This implies a dynamic approach.

10.2.2. Modeling for recreation issues

Why use models for recreation issues? At least two important reasons can be given. First, understanding of recreation as a real world process is almost impossible without coupling different aspects of it within one frame of reference. This frame may be verbal (say conceptual); much scientific literature on recreation and leisure only conceptually analyzes the field. Quantitative understanding, however, requires mathematical formulations. Much literature uses standard formulations as expressed in more or less well-known statistics and computer-based graphics, but quantitative *projections* are almost impossible without mathematical models as a base. Second, models may be helpful in formulating effective policies for the issues mentioned above.

As regards "understanding", particularly economists have developed families of recreation models (for measuring benefits, for congestion problems, etc.). It is remarkable that ecologists have almost not tried to do the same; they usually adhere to regression analysis, ANOVA, factor analysis, etc. (which, in this paper, are not considered models but statistical techniques). Geographers (and also economists) sometimes use gravity models to explain or predict spatial features of recreation.

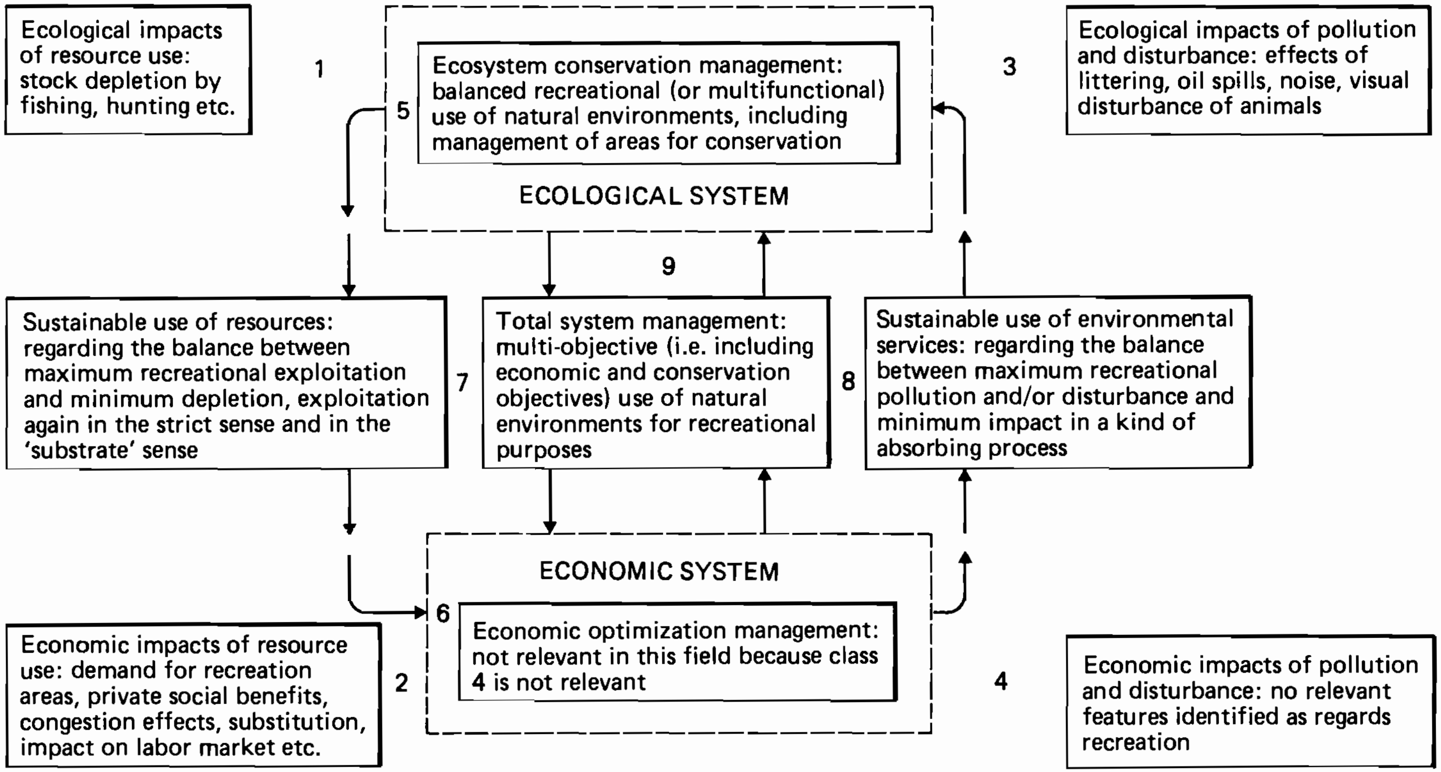

Figure 10.1 Recreation and policy issue classes.

Most policy issues relating to recreation include some time dimension. If statistical methods like moving averages do not satisfy, dynamic models are needed. This is not necessarily the case in models used for assessing or evaluating policy alternatives. Both static and dynamic models may be used then, depending on the objectives defined. Again, in economics families of decision models (e.g., optimization, multicriteria analysis, cost–benefit) have been widely used while in ecology almost nothing of this kind has been attempted.

Inevitably, decisions are being made on basis of sound economic modeling only, without an ecological counterpart. In many cases this may negatively influence the natural environment without that influence being analyzed or predicted. This gap of skill and knowledge enforces biased, imperfect decision making.

Thus the conclusion would be that economic and ecological modeling, whether or not as an integrated exercise, would probably contribute to a better case for the environment.

10.2.3. Recreation in the IvM-IIASA questionnaire

In the questionnaire, the issue has been mentioned 19 times against the average of 25 for all issues. There seems to be a more than average interest from modelers in densely populated countries like Japan and the Netherlands. In models announced from Canada and the FRG recreation was not mentioned at all.

The sample of 19 models markedly differs from the survey sample as a whole in several respects. First, the purpose of 67% (40% in the whole survey) is application to a specific (policy) case. Consequently, 52% (*vs.* 33%) has been applied in an actual policy context. Many of these models, however, have not been tested yet.

Recreation seems to be specifically linked with some other fields of application, particularly with "Land use" and "Water", but also with "Nature conservation", "Agriculture", and "Fisheries". Many of these models deal with three or more application fields and can therefore be characterized as complex decision-making models.

The majority of the submodels is dynamic; this holds for both economic and ecological submodels. The geographical scale is almost invariably the regional scale. The economic submodels use both optimization and simulation techniques, the ecological submodels are mainly simulation models.

None of the studies in the questionnaire sample mentions outdoor recreation or tourism as the main field of application. Possibly recreation is not considered an isolated problem but is analyzed in the scope of "multiple use of areas" (*see* above for combinations of fields of application). However, this does not agree with the rich history of *economic* modeling of recreation issues (which sometimes are labeled environmental studies).

10.3. Economic Recreation Models

Economic studies concerning outdoor recreation usually focus on three issues:

(1) Assessment or optimization of recreational benefits.
(2) Benefits (and costs) of recreational areas to be developed.
(3) Economic aspects of distributional features in recreation.

For assessment of recreational benefits, most authors adopt the Hotelling–Clawson–Knetsch (HCK) method, usually called "travel–cost–approach" (Clawson, 1959; Clawson and Knetsch, 1966). In this approach rate of use (demand) of a specific area for different distance zones is related to the costs of visiting the area. The marginal user pays the maximum price and from the demand curve consumers' surpluses can be derived as:

$$\sum_i n_i \int_{p_i}^{p_0} f(p)\,\mathrm{d}p \tag{10.1}$$

where: q = the demand (visit rate) = $f(p)$, p_i = the travel cost for zone i, p_0 = the travel cost for the marginal zone 0, n_i = the population in zone i.

Although this method has been widely adopted, there is a vast amount of literature criticizing it. Entrance fees are regarded equal to travel costs as to impact on behavior, which is a strong assumption (Common, 1973). Time spent in traveling is regarded as part of the travel costs (Burt and Brewer, 1971). Cesario and Knetsch (1976) suggest to use trade-off functions between time and money. This issue is also discussed by Freeman (1979) and Bishop and Heberlein (1979). Another problem is formed by the assumption that recreationists have only one purpose for their trip, which is not always true: people may also derive benefits from the travel as such, or may be heading for another site and just pass by (Cheshire and Stabler, 1976). Income differences may be another source of uncertainty (Seckler, 1966), and a Hicksian (compensating variation) demand curve should be used rather than a Marshallian one. Harrison and Stabler (1981) conclude that travel mode and distance traveled are strongly dependent on income. Congestion may lower the quality of a site, particularly in wilderness recreation areas (Smith, 1975); this issue has recently been discussed by Wetzel (1977, 1981) and others (e.g., McConnell, 1980; Smith, 1980).

Smith, Desvousges, and McGivney (1983) again studied the problem of the opportunity cost of travel time, extending research of Cesario (1976) and McConnell and Strand (1981). They used a semi-log specification:

$$\text{n}V_{jn} = a_{oj} + a_{1j}MC_{jn} + a_{2j}TC_{jn} + a_{3j}Y_n + a_{4j}SC_{jn} + \sum_s b_{sj}X_{sn} + \varepsilon_{jn} \tag{10.2}$$

where: V_{jn} = the number of trips to site j by individual n; MC_{jn} = vehicle-related travel cost of a trip (j,n); TC_{jn} = travel time costs of a trip (j,n); Y_n = family income of individual n; SC_{jn} = on-site time costs of a trip (j,n); X_{sn} = sth socioeconomic feature of individual n; ε_{jn} = stochastic error term for jth demand function and associated with nth observation; a_{oj}, etc., are coefficients.

This model was used to investigate the importance of TC_{jn} and the possible estimators for it, applying data of 20 aquatic recreational resources. Apparently, however, present data do not allow a definite conclusion, although it has become clear that there is a considerable difference between "local" sites and "national" sites as regards individual time constraints. Anyway the time factor has proven to be crucial in estimating benefits.

Optimization models for exploitational benefits are mostly developed for "sustainable resource use" problems and will be dealt with later.

Benefit studies of *planned* recreational resources also mostly use the HCK-approach (e.g., Burt and Brewer, 1971; Mansfield, 1971). However, as there is a change in spatial distribution of visitors as a result of adding one new facility, often gravity models are included (*see* below).

Comparable approaches can be used in estimating the opportunity costs of recreational sites where non-recreational activities are planned. Krutilla and Fisher (1975) use the following computational model:

$$PV = \sum_{t=1}^{T'} b_o(1 + \alpha)^t \; (1 + i)^{-t} \tag{10.3}$$

where PV = the present recreational value; b_0 = the recreational benefits (obtained by the methods described above) in the first year; $\bar{\alpha}$ = an average rate of appreciation, α_t; i = the discount rate; t = index of time (in years); T' = the terminal year (PV of an initial money unit growing at rate α_t, discounted by rate i, falls low).

A different approach was proposed by Bouma (1976), estimating opportunity costs by using the (travel and time) costs of surplus distances for visitors who travel to the most nearby recreational facility with the same qualities as the one forgone. Of course this approach requires both source studies and site studies while HCK and derivations are site-specific. The model has not been actually calibrated and tested.

Gravity models have been widely used for estimating numbers of visits in cases where several trip origins and destinations, each with specific properties, are at issue. In many cases the gravity model is being coupled to a demand estimation model. Usually the general equations are in a form like:

$$T_{ij} = P_i \frac{A_j \ F_{ij}}{\sum A_j \ F_{ij}} \tag{10.4}$$

where: i = origin; j = destination; T_{ij} = number of activity days produced at i and attracted to j; P_i = number of activity days produced at the ith origin; A_j = number of activity days attracted to the jth destination; F_{ij} = a calibration term for interchange ij. F_{ij} estimates the probability of traveling from i to j. Usually A_j contains an "attraction factor", representing site quality and area size. Examples include Mansfield (1971), Cesario and Knetsch (1976), uaxter and Ewing (1981) and Sutherland (1982). Baxter and Ewing (1981) also included barrier effects (e.g., from cities or from large surface water areas), and multi-stop trips.

From the above a wide interest of economists in recreational issues can be concluded. Remarkably enough, many models have not been calibrated at all. Apparently most authors have been interested mainly in the refinement of theories about valuating nonmarket (public) goods.

10.4. Ecological Recreation Models

As already stated above, models relating recreation to ecosystem properties are scarce. Much attention has been paid by ecologists to statistical determination of dose–effect (or activity–effect) relationships, mainly by regression analysis (e.g., Van der Zande *et al.*, 1984). Curve fitting (Hylgaard and Liddle, 1981) and factor analysis (Boomsma and Van der Ploeg, 1976) have been used as well.

Getz (1978) developed a system of nonlinear ordinary differential equations to simulate temporal patterns in stressed ecosystems, notably recreation in a coniferous forest. The basic formulation of the model FERM runs:

$$\dot{x}_i = a_i x \{1-[x_i / s_i({}^i e)]^{\beta_i}\} + \sum_{j=1}^{p+q+r-1} b_{ij}\, {}^i e_j \varphi_{ij}(x_i) \qquad (10.5)$$

where: $i = 1,\dots,n$; $j = 1,\dots,n$; $j = i$ (for jth system variable influencing ith population); x_i = a measure for the actual dynamics of the ith population; s_i, β_i = carrying capacity constants for i; $b_{ij}^i e_j \varphi_{ij}(x_i)$ = activity–impact or predator–prey function; and:

$${}^i e = \begin{bmatrix} x \\ u \\ y \\ \lambda \end{bmatrix} \begin{array}{l} = \text{environment of } i \text{, i.e., the remaining state variables;} \\ = \text{management functions } u_i(t), i = i,\dots,p; \\ = \text{stress (recreation) functions } y_i(t), i = 1,\dots,q; \\ = \text{driving (abiotic) functions } \lambda_i(t), i = 1,\dots,r; \end{array}$$

The model was detailed into four equations for biomass (kg) of timber, deer, fish, and forage (herbs, grasses). Parameters were taken from current literature and simulation runs were done for three different recreation intensities and for timber harvesting, simulating over 4 years with monthly intervals. Because of the data available no tests were done.

The model TERRA (Hunter, 1979) uses sets of difference equations in mechanistic and descriptive simulations with management alternatives as input. The overall structure of the model is shown in *Figure 10.2*. Relationships used are not based on regression but on known physical and biological relationships. There are five submodels: the abiotic, the forage (grasses, herbs), the timber, the consumer (deer, fish), and the recreation submodel. The latter is split up in three parts: a part determining the number of recreationists "at the gate" of the ecosystem (a watershed); a part allocating recreationists through the watershed by vegetation zone and user type, also calculating user days, and a part determining fish caught and deer killed by recreationists. A separate submodel enters management alternatives into the other ones. Various simulation runs were done and the results were qualitatively validated (as no sufficient data were available).

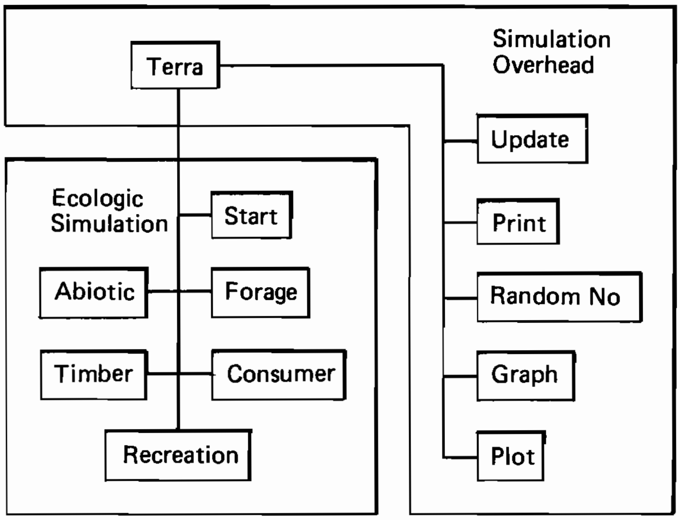

Figure 10.2. Modular design of the model TERRA (Hunter, 1979).

Potentially these models are applicable for the policy issue classes 1, 3, and 5. Obviously, however, the lack of substantial data is a prohibiting factor in refining and validating them. Although not extremely complex, the data requirements for both models seem large and therefore further applications to planning or management issues seem unlikely. Reduction of such models to a modest size, however, is only possible by using full-range information. There is a vicious circle here.

10.5. Economic–Ecological Recreation Models

10.5.1. Models for sustainable recreational resource exploitation

Policy issue class 7 contains problems of exploiting natural resources for recreational purposes like fishing or hunting. As regards commercial fishing, many bioeconomic models have been developed. This issue is reviewed by Clark (Chapter 5).

For recreational hunting on waterfowl, Brown and Hammack (1972) have done an extensive study concerning prairie wetlands in the USA and Canada. A summary and some comments are given in Krutilla and Fisher (1975). The model focuses on the optimum number of ponds used by ducks (notably mallards) for breeding. Farmers owning land with some of these ponds do not share in the benefits of hunters or photographers on the flyways to overwintering areas. The objective is maximization of the present value of net benefits. Benefits are represented by consumer (hunter) surplus, costs by the opportunity costs of the ponds (agricultural benefits forgone) (Hammack and Brown, 1974):

$$\text{Max} \int_0^{\infty} \left\{ N^* V(B,I,S,E) - C(P) \right\} e^{-\alpha t} \, dt \tag{10.6}$$

where N (number of hunters), V (the individual hunter valuation function), B (the bagged waterfowl), I, S, and E (socioeconomic characteristics of the hunters, i.e., income, number of seasons of hunting, and hunter costs per season) are the benefit variables, while C (pond cost function) and P (number of ponds) represent the costs. α represents the discount rate (8%).

The given objective function is subject to a population dynamics differential equation:

$$dW/dt = -W + s_2(I + s_1 W - c_3 NB) \tag{10.7}$$

where W = the number of mature birds; I = the number of immatures; s_1 and s_2 = survival fractions of adults; c_3 = an adjustment for kills not bagged.

As the ponds are assumed to be a reversible resource, there is no time function problem in this optimal control issue. Thus the system evolves toward a steady-state solution for P, B, and W, using a Hamiltonian and three necessary conditions. From an ecological point of view this is somewhat doubtful as the creation of new ponds cannot replace foregone ponds in an ecological sense. As to the issue under consideration, however, this doubt does not seem obstructive.

Maximum sustained yield (duck numbers) for a fixed number of ponds is calculated from the above population equation combined with a multiplicative production function. From existing data both the optimization model and the maximum sustained yield model are estimated.

Krutilla and Fisher (1975) emphasize the (empirical) difference between the economic solution (larger breeding stock at any suggested pond cost level, allowing a larger duck kill) and the biological "optimum" which is much lower. This difference is mainly caused by the number of ponds itself (goal variable!) which is assumed constant in the population equations.

This bioeconomic model is promising because of its simplicity, but it also has some drawbacks. First, ducks are considered a consumable stock rather than part of ecosystems. Second, maximum sustainable yield is considered "optimal" instead of "possible". Third, only hunters (consumptive users) are considered; nonconsumptive users (photographers, etc.) are disregarded. Fourth, creation of new ponds is part of the optimal economic solution, regardless spatial or ecological limitations. Thus the model is essentially an economic optimization model, with auxiliary ecological equations as partial boundary conditions.

Recent publications on recreational resource exploitation invariably emphasize economic modeling without paying much attention to problems of sustainable yield. Several issues are dealt with, such as the market approach to hunting leases (Livengood, 1983; Sandrey *et al.*, 1983), impact of fish-stock enhancement (postulated only) on demand curves (Anderson, 1983), measurement of time cost (McConnell and Strand, 1981) and more general valuations of resource service flows (Brookshire *et al.*, 1980). The latter authors use the concept of "willingness to pay" (WTP), but also the concept of "willingness to accept" (WTA) decrements in goods, services, and amenities. WTA is not always measurable (the authors used iterative bidding techniques for contingent valuation) but can be derived from WTP. Only part of these models has been calibrated or tested with empirical data. It is not quite clear whether the models have been actually used for policymaking.

The approach used by Miller (Miller and Hay, 1981; Miller, 1982), based on logit formulations, incorporates geographical and "substitute" ecological aspects, next to the predominating economic approach. Miller and Hay (1981) use acres of habitat (waterfowl, wetlands, and upland waterfowl) as explanatory variables in participation equations for duck hunting, with the aim of determining recreational benefits as opposed to possible agricultural land use of these areas. Miller (1982) investigates the relationships between game availability and hunter participation, using a gravity model related to the observed distribution of elk herds. An empirically tested example suggests that a decrease in elk populations induces a larger decrease in elk hunting days. The model, however, is not dynamic so that impacts of population changes over time are not incorporated.

Another interesting issue concerns the possible conflict between commercial and recreational fishing. McConnell and Sutinen (1979) present some theoretical modeling approaches for open access regimes versus optimal management. Price elasticities of effort in the commercial and the recreational sector are supposed to be the critical parameters. The paper uses a common differential equation for the natural growth rate of the fish stock.

Bishop and Samples (1980) go the reverse way, starting with the Clark–Munro bioeconomic model for commercial fishing (Clark and Munro, 1975). This model is revised including the noncommercial sector. Both a nonlinear model and a predator–prey model (with two predators: commercial and recreational fishers, and one prey: the fish population) are surveyed. The resulting model only accounts for optimization and not for changes over time. Moreover, no empirical calibration or validation is tried. However, this kind of model may be useful in determining optimal levels of both commercial and recreational fishing.

Finally, most papers referred to emphasize that fishing under open access may result in decrease of stocks. Although no empirical information for the ecological aspects is given, linkage of these models to operational ecological population models seems to be an easy next step.

10.5.2. Models concerning sustainable use of environmental services

Policy class 8 deals with the balance between recreational pollution or disturbance and the capacity of ecosystems to assimilate these inputs or influences. A conceptual model for this class would consist of three levels:

(1) A submodel generating actual total use, e.g., by combination of a demand function and a distance function (Cesario and Knetsch, 1976).
(2) A submodel generating distributional patterns of recreation within the area under consideration (Hunter, 1979).
(3) A submodel describing impacts of visitor activities on parts of ecosystems (Getz, 1978).

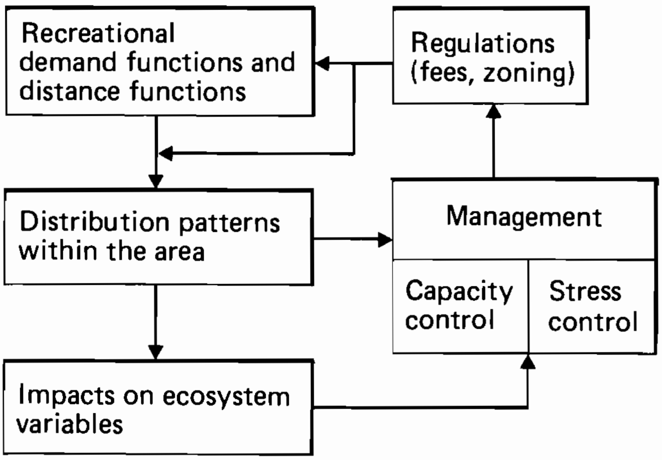

Figure 10.3. Conceptual sustainable service model.

Feedbacks from the second and the third submodel through (4) a management submodel should be added. *Figure 10.3* shows the submodels and their connections.

Studies incorporating all issues and their connections have not been found in current scientific journals. However, some attention has been paid to the different aspects of the model of *Figure 10.3*. Particularly for wilderness recreation in the USA models have been developed. Some examples are given below.

Romesburg (1974) presented theoretical models for hiking trails and for float trips. These are optimization models with options for linear, quadratic and goal program solutions. Markov models for boat trips were used by Gilbert *et al.* (1972) and by DeBettencourt *et al.* (1978).

Shechter and Lucas (1978) compare optimization and Markov-based techniques with the simulation model they developed. They conclude that simulation models for these issues generally give more opportunities to incorporate several aspects of the problem without being hampered by technical obstructions. The simulator developed (*see also* Shechter and Lucas, 1980a) includes a route network, user characteristics, route–user interaction, user–user interactions, and policy variables (with objectives as criteria). A general flow chart is shown in *Figure 10.4*. The model has been applied several times by USDA Forest Service and much attention has been paid to different possibilities for validation (Shechter and Lucas, 1978; 1980b). No economic features are included, although the authors refer to coupling the simulator with the models for wilderness travel developed by Cicchetti and Smith (1976) and Smith and Krutilla (1976).

Walter and Schofield (1977) present a model for wilderness recreation resource management which approaches the problem from the economic viewpoint. The model aims to maximize net management bureau income, tourist expenditure benefits and consumers' surpluses, under constraints for environmental costs, congestion costs, and minimum prices. Only for part of these variables, however, empirical estimates were made (not for the environmental costs). Thus the model is only partly calibrated.

Bertuglia *et al.* (1980) have developed a theoretical optimization model for managing recreation in natural environments. Basically this model used optimal control methods with ecological population dynamics explicitly formulated in the constraint. No application or empirical estimation is given. Bertuglia and Tadei (1983), however, report to have estimated some values. This paper deals with the locational behavior of visitors by means of a systems model. As both demand and ecological impacts are dealt with in a rather obscure way, the model does not help very much in elucidating the policy issue, though it is mathematically interesting.

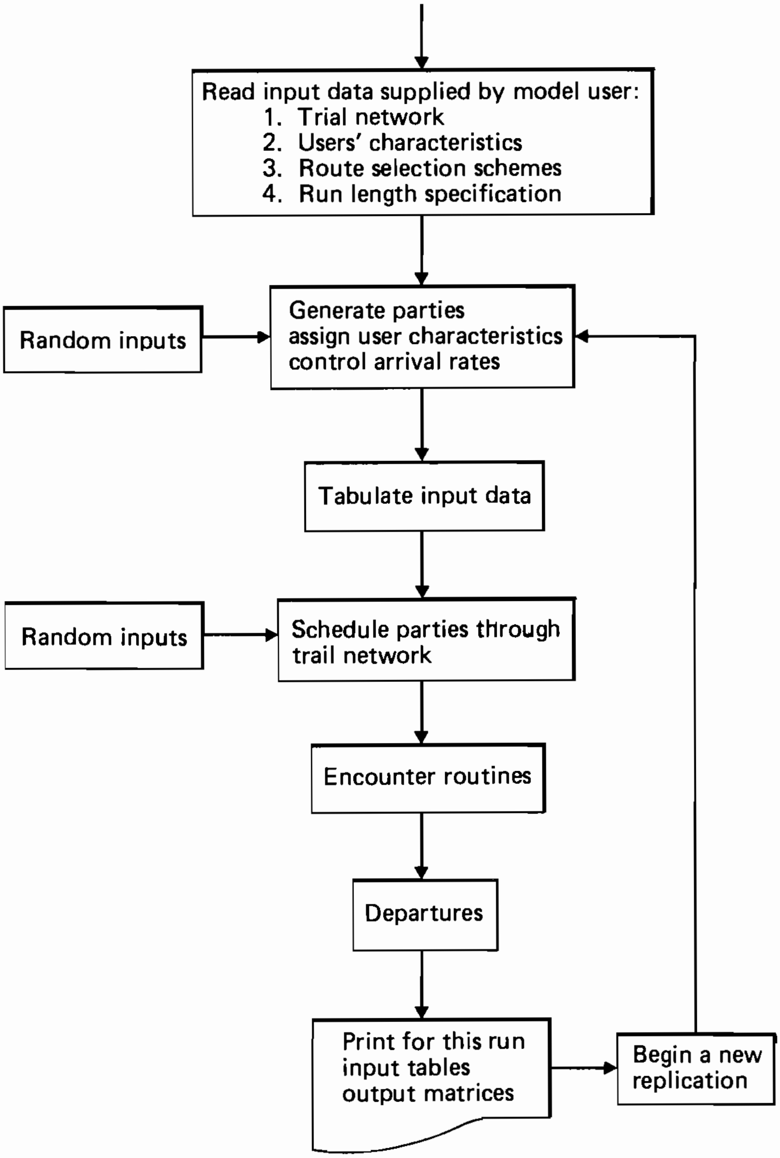

Figure 10.4. Basic flow chart of the wilderness use simulation model (after Shechter and Lucas, 1980a).

In conclusion, models dealing with problems which can be classified as sustainable service use ones, are not yet covering the total issue as designed in *Figure 10.3*. Rather, these models deal with the economic–distributional features, or only the distributional ones.

Ecological impacts are mostly introduced by "best possible judgment constraints".

10.5.3. Total system models

Most models pertaining to policy class 9 regard regions, countries, or the world as a whole. Therefore these models handle recreation only as one out of various land use possibilities, as the ecological impacts of recreation are usually small compared with other land use forms.

Elsewhere in this book examples are given of such multiuse models. The compartment model of Ikeda (*see* Chapter 12) explicitly mentions recreation in the coastal resources demand model, the pollutant emission model, and the marine ecological model. As the model is not really operational, no conclusions for the role of recreation in it can be derived yet.

In Chapter 13 Odum describes his method based on energy as a common denominator. An early example of applying this method to land use planning has been given by Boynton *et al.* (1977) for a region in Florida. Although recreation is explicitly involved in the model, impacts are considered marginal (in terms of energy flows) in comparison with other land uses (*see also* Chapter 11).

Although York *et al.* (1977a,b) present a flexible model for assessing benefits of multiple area use, ecological impacts are only included as decrements of semitangible benefits from environment-related activities. The empirical study (York *et al.*, 1977b) indeed refers mostly to hunting and fishing as recreational activities, as opposed to timber harvest, though explicit attention is paid to compatibility of activities, including nature observation. No ecological data, however, are required for the model.

Finally, total system modeling for recreational use of the natural environment in the Netherlands has been tried by Van der Ploeg *et al.* (1984). The model specifications used have partially been described in Braat and van Lierop (1984). The model includes demographic changes, relevant economic variables, total recreational demand functions, spatial recreation distribution equations, and ecological impact equations. The model system (divided into submodels) is roughly arranged as in *Figure 10.3*. Because of lack of appropriate data, only part of the ecological submodel has been estimated empirically. Results have been used in policymaking for National Parks.

10.6. Conclusions

Many studies incorporating models have been referred to; few studies really concentrate on an integrated approach of recreation policy issues. Obviously, economists and ecologists are living in different "real worlds". Also obviously, in both worlds recreation is an important phenomenon. It is as well in the world of geographers (and also of social scientists). But there does not seem to be an incentive for cooperation.

Nature conservation and environmental health have proven to conflict with recreational use. Thus, it might be sensible to develop management schemes in which:

(1) Recreational use (physical use as well as exploitation).
(2) Recreational benefits.
(3) Ecological impacts of recreational use are optimized.

The apparent combination would then be:

(1) Simulation models for recreational use and ecological impacts.
(2) Optimization models for recreational use and benefits.

The models presented in this chapter allow for almost any combination of different objectives and methods. It seems that no further sophistication of these instruments is urgently needed. Rather, amalgamating of expertise is required now.

An astonishing example of the "different worlds" of economists and ecologists is given in Jeffers (1972) where, at a Symposium on mathematical ecology, a conceptual optimization model was proposed by the late George Van Dyne for approaching future ecological impacts of a barrage scheme on Morecambe Bay, Lancashire. No reference is made to recreation. At the same time (spring, 1971) the paper of the late N.W. Mansfield (1971) was published, estimating the benefits of exactly the same area. No reference to the ecological qualities of the area is made in this paper. Moreover, no optimization proposal in the economic paper; no simulation proposal in the ecological discussion.

Finally, multiple land use models probably reflect reality best of all for small or densely populated countries. Recreation is then only one of the issues involved. However, in large countries (like the USA or Canada) land use is often strongly segregated. This difference in overall land use requires different assemblies of model types.

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CHAPTER 11

Analyzing Productivity of Multiple Resource Systems

F.L. Tóth

11.1. Introduction

The earlier chapters in Part II have reviewed models useful to those concerned with a specific resource, such as forests or fisheries. In this chapter I review three models of regional development, in which it is always necessary to consider interactions among a mixed set of resources. Regional planners should ensure that productivity of one resource is not pursued at the cost of significant deterioration in several others. That is the general goal of the three models I consider here. In addition to the biological and ecological problems found in management of natural resources, the models developed for use in regional planning should always analyze economic, political, and social factors involved in the interaction between man and nature. The three models covered in this report illustrate quite different approaches to that goal.

The first model (Boynton *et al.*, 1977) was developed to reveal the role of coastal resources in the ecological balance and economic vitality of a coastal region where the local economy is predominantly dependent on activities based on water resources. At the beginning of the study, new developments (causeway to offshore island, expansion of tourist trade, etc.) threatened these resources. Of special concern was fishery losses by pollution from sewage. Therefore, there was an urgent need to assess the impacts of different management decisions on the ecosystem and to use the results in policymaking.

The final outcome of the second study was a management tool developed to help policymakers plan land and water resource use in river catchments. The model was designed by Bennett *et al.*, (1977), specifically to formulate optimal land use policies and evaluate different dam building strategies. Catchments require more careful land use planning and regulations than most other agricultural areas, because, quite often, the water coming from the catchment has a value similar to that of the agricultural products that are grown in the catchment. Therefore, a model was developed to determine the socially optimal land use plan.

The third model was developed for a national survey (Láng, 1983) in Hungary where agriculture has a long-standing importance. The growth rate achieved in plant production in this country is the second highest in the world. The basic goals of the project were to determine "the limits to growth", the maximal amount of production that, given the natural environment, meteorological effects, soil properties, water supply, the genetic properties of plants, and the partial modification of environmental factors (land reclamation, irrigation), could be obtained by the turn of the century. The biomass program aimed to clarify internal relationships of the biomass production–transformation–utilization cycle and to the determine the future function and role of this system in connection with the rational use of renewable and non-renewable resources.

In the first study, a special systems simulation technique, energy flow modeling, was applied. The second and third projects used linear programming models but with different flavors. These two modeling approaches, simulation, and optimization, and different combinations of them, appear to be the most frequently and most successfully applied methods in investigating multiple or total resource management problems.

11.2. Regional Planning Model for Franklin County and Apalachicola Bay, Florida, USA

11.2.1. Introduction

Recent experience all over the world have shown that new construction often tends to take place on natural estuarine and coastal resources. In many cases the building activity may destroy some of the resources that were originally a vital attraction for the development. Some specific controversies that have arisen recently in coastal regions include such issues as channelization, nutrient loading, and the development of wetlands and coastal areas for industrial, commercial, and residential uses. This project aimed at determining the best mix of developed areas and self-renewing fisheries for a coastal county in

Florida. The work focused on evaluating the sensitivity of the local oyster fishery to additional developments of the tourist, retirement, shipping, and cattle industries.

The model was built by W. Boynton, D.E. Hawkins, and C. Gray at the University of Florida. Their work was supported by the Florida Sea Grant Program.

11.2.2. Model structure

The underlying philosophy of the method applied in this project is Lotka's maximum power principle. It states that two features characterize any system that tends to prevail over alternative systems: (1) it maximizes the use of all energy flows available to it; (2) it developes useful feedback roles for all participants. These feedbacks assure continued energy flows and capture any additional useful flows that become available. In a system such as the region under study containing both man and nature (towns, estuaries, fisheries, tourism, and forests), this means using a large variety of components wisely in combination that will obtain maximum power flows through the total system, survive periods of stress, and build means for a vital economy of both man and nature in the long run. The wedding of Lotka's principle to the type of system modeling used in this project is best summarized in Odum (1983b) and Odum and Odum (1981).

The authors developed a regional model of Apalachicola Bay, its principal economic factors, and the most important interactions among these factors. Their model is divided into three main sections: Apalachicola Bay, including the basic biotic and physical parameters, the oyster industry, and the developed portion of the county. River flow transports inorganic nutrients, organic matter, toxins, and the coliform bacteria into the bay in "baseline" concentrations independent of man's activities. Contributions to each of the above are also made from local sources that change as a function of development. Tidal exchange flushes the system with an average turnover time of several days. In the model, the bay is considered to be a homogeneous body of water with no local gradients. Phytoplankton production is influenced by sunlight and nutrient concentration. It was concluded that nitrogen was the factor most likely to limit photosynthesis. Other important model assumptions were derived in similar fashion, based on literature reviews, expert opinion, and field work conducted in the bay. (There are 54 exogenous constants in the model.)

The organic matter in the bay supports a population of oysters, other benthic organisms, and predators that feed in part on the oysters. Most oyster predators in this area are stressed or eliminated by salinity fluctuations. Toxins stress the oyster population directly, while coliforms are an index of sewage pollution. They operate a switch in the model regulating oyster harvest.

Oyster harvest is proportional to the standing crop of oysters and to the size of the oyster industry. The oysters are sold in an external market, and the income is used to buy needed goods and materials from both within the county and from outside sources. Prices regulate the ratio of dollars to goods in each exchange and have an important effect in determining the relative economic position of each sector in the model.

The developed sector in the model has income from the oyster industry in exchange of goods and services. The income of this sector is exchanged for goods, fuels, and services needed to maintain and add new structure (the physical infrastructure of the settled areas). The number of inhabitants is increased or decreased by migration, which is dependent on the regional image and the city structure. The losses are due to mortality and emigration from the county. The birth rate is approximately constant.

The flow diagram of the model is given in Figure 2 (p. 484) of Boynton *et al.* (1977). The corresponding system equations are listed below:

Q_1 = Developed land:

$$\dot{Q}_1 = k_1 Q_2 Q_3 - k_2 Q_1 \quad (11.1)$$

Q_2 = Available land:

$$\dot{Q}_2 = k_2 Q_1 - k_3 Q_2 Q_3 \quad (k_3 = k_1) \quad (11.2)$$

Q_3 = Local capital:

$$\dot{Q}_3 = k_4 Q_3 Q_4 I_0 + k_5 I_2 I_3 + k_6 Q_4 Q_5 I_{15} I_{16} + k_{55} Q_4 + k_7 Q_9 - k_8 Q_3 - k_9 k_{10} Q_6 - k_{11} Q_3 \quad (11.3)$$

Q_4 = City structure:

$$\dot{Q}_4 = k_{12} Q_2 Q_3 + k_{13} Q_3 + k_{14} Q_4 Q_9 - k_{15} Q_4^2 - k_{16} Q_4 - k_{57} Q_4 \quad (11.4)$$

Q_5 = Image:

$$\dot{Q}_5 = k_{17}Q_2 + k_{18}Q_4 - k_{19}Q_4^2 + k_{20}Q_{11} - k_{21}Q_5 \tag{11.5}$$

Q_6 = Residents:

$$\dot{Q}_6 = k_{22}I_2Q_4Q_5 - k_9Q_6 - k_{23}Q_6 + k_{56}Q_6 \tag{11.6}$$

Q_7 = Tourists:

$$\dot{Q}_7 = k_{24}I_{16}Q_5 - k_{25}Q_7 \tag{11.7}$$

Q_8 = Freshwater in bay:

$$\dot{Q}_8 = k_{32}I_6 - k_{33}Q_8 \tag{11.8}$$

Q_9 = Oyster industry capital:

$$\dot{Q}_9 = k_{27}Q_{10}Q_{11} - k_7Q_9 - k_{28}Q_9 \tag{11.9}$$

Q_{10} = Oyster industry structure:

$$\dot{Q}_{10} = k_{29}Q_9 + k_{14}Q_4Q_9 - k_{30}Q_{10}Q_{11} - k_{31}Q_{10} \tag{11.10}$$

Q_{11} = Oysters:

$$\dot{Q}_{11} = k_{34}Q_{12} - k_{35}Q_{11} - k_{36}Q_{11}Q_{16} - k_{26}Q_{10}Q_{11} - k_{37}Q_{11}Q_{13} \tag{11.11}$$

Q_{12} = Organic matter in bay: (11.12)

$$\dot{Q}_{12} = k_{38}I_7I_6 + k_{39}I_{10}Q_{14} - k_{40}Q_{12} - k_{42}Q_{12} - k_{34}Q_{12}$$

Q_{13} = Toxins in bay:

$$\dot{Q}_{13} = k_{43} I_6 I_9 + k_{44} Q_4 - k_{45} Q_{13} \tag{11.13}$$

Q_{14} = Nutrients in bay:

$$\dot{Q}_{14} = k_{47} I_6 I_8 + k_{48} Q_6 + Q_7 - k_{49} Q_{14} - k_{50} I_{10} Q_{14} \tag{11.14}$$

Q_{15} = Coliforms in the bay:

$$\dot{Q}_{15} = k_{51} I_6 I_5 + k_{52} Q_6 + Q_7 - k_{53} Q_{15} \tag{11.15}$$

Q_{16} = Oyster predators:

$$Q_{16} = k_{42} Q_{12} + k_{36} Q_{11} Q_{16} - k_{54} I_6 Q_{16} \tag{11.16}$$

11.2.3. Model evaluation

Since models of this sort have been used only a few years, there is no long history by which to judge the predictive value of large open system simulation models such as the one presented here. Instead, historical data were used to validate the simulation model. The modeling approach assumes that if all important time-varying energy sources are included in the model, then the model's general behavior will be characteristic of past and future trends in the real system. The authors presented one historical simulation using 1970 values for forcing functions and coefficients, but starting from conditions present in 1820. Although the simulated population growth of the region was similar to the real population growth, the development of land, and the growth of the oyster fishery were both considerably less than what actually has occurred. The reasons, however, can be found outside of the model.

The results of different simulation runs have already been used in planning decisions. This fact itself is a clear justification of this effort and of the methodology in general. This particular application shows that the energy flow modeling technique is one of the useful tools to describe economic decision alternatives affecting ecological processes in a given region, to explore possibilities of utilization of various resources available there, to examine feedbacks, trade-offs, and responses of the ecological system to economic policy alternatives, and, as a result, to develop improved policies for regional development.

There is one other advantage of this type of modeling. Once the well-defined and validated model structure is developed, it is easy and inexpensive to run a large number of different versions each representing a policy option. Evaluation of these simulation runs can lead to further refinement and improvements in the model and to further testing of development alternatives. This feature makes the cost/benefit ratio of this type of analysis very favorable.

11.3. Catchment Land Use Planning, Murray River, Western Australia

11.3.1. Introduction

Management of land and water resources in salt-affected catchments presents problems for both ecology and economics as well as for policymakers. Clearing of land for agriculture causes the water table to rise. Salt is leached from the soil, it passes into groundwater and into streams. While this mechanism is relatively well understood, it is, however, difficult to make adequate predictions of the effects of land use on water quality, because the catchment is a distributed, heterogeneous, and slowly varying hydrological system. The conditions that make hydrological predictions difficult also hinder economic evaluation of different resource management plans.

Timber, water, recreation, conservation, or mining? Which of these or several other uses should take priority? What forms of land uses are in the best short- and long-term interest of this state? The study addresses problems of multiple land use, water supply, recreation, agriculture, and forestry, and, in general, of regional planning in Western Australia.

The resources of the northern jarrah forest provide both wood and agricultural products, minerals (bauxite, gravel, and blue metal), and various recreational and conservational values. Its use as a catchment for household and industrial water supply and for irrigation in agriculture is also very important. The bulk of the land is still covered by forest. However, much of the region is privately owned, and it is being cleared for pasture or crops. Thus resource utilization involves different government departments, private companies, and individual landowners.

A number of experts from the Commonwealth Scientific and Industrial Research Organization (CSIRO) Division of Land Resource Management, the Department of Agriculture, and the Forest Department came together to propose a systematic and economic plan for the catchment. Members of other government departments and private companies assisted the group as advisers in more specialized fields. Public opinion on recreational use of the catchment was obtained through surveys and questionnaires administered by the Forest Department.

11.3.2. Model structure

A mathematical model was used to allocate land uses in the river catchment, under various scenarios of planning, including the option of not damming the river. The catchment was divided into 41 zones according to land form types, rainfall, and vegetation coverage. All probable land use activities were considered for each zone. A computer program was written that would account for all combinations of the activities in the zones. Eleven land uses were considered, including flora and fauna reserves, national parks, eucalyptus hardwood forestry, plantation forests, agriculture (as adapted to different zones), agroforestry, streamline plantations, water run-off enhancement by surface sealing land surfaces (building roads, for example), bauxite mining (followed by forestry, agriculture, or roaded subcatchments), and water storage. The task was to assess the economic value of each activity in each zone and to work out which of three water-using developments would be the most efficient land use:

(1) One dam with reforestation,
(2) Two dams and diversion of saline headwaters,
(3) No dams.

The flow diagram of the study is shown in *Figure 11.1*. The core of the procedure is an optimization package called TOPAZ-WA. Subroutine MURFIN-B supplies it with a matrix of the some of the discounted present values of water, land-use activities, salt, and recreation. Matrices of water and salt yields under the different land use activities are also entered for constraints on salinity. The problem stated in its simplest possible form is to allocate land uses to zones, so that something is optimized. The optimization depended on "merit values",

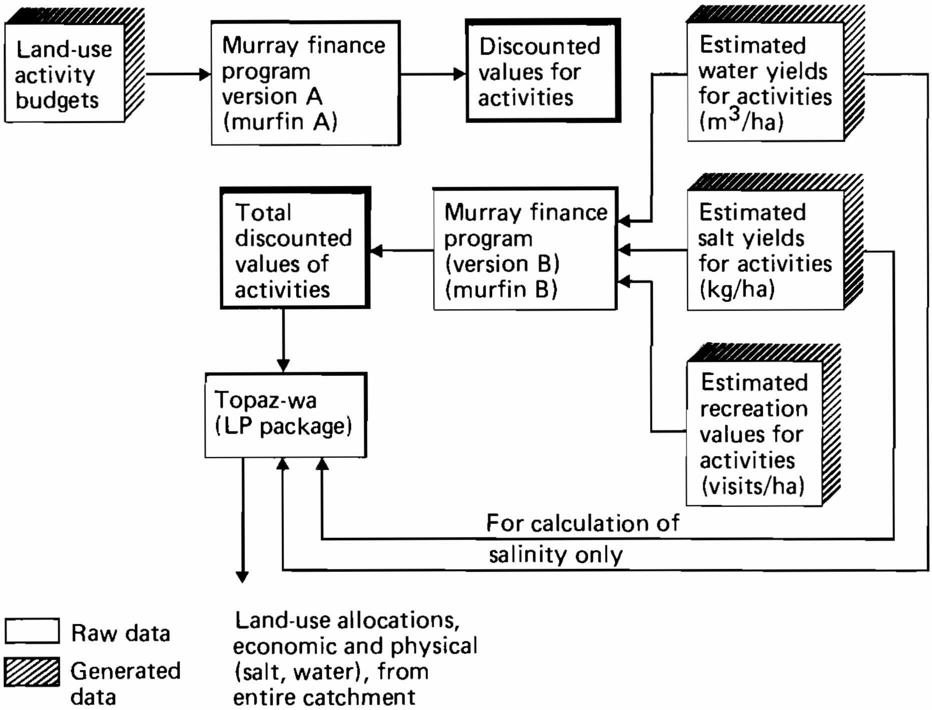

Figure 11.1. Relationships between computer programs and data used in the catchment planning study.

normally net present values of activities. These were maximized to give a measure of total merit $U(A)$ such that:

$$U(A) = \sum b_{ij} a_{ij} \tag{11.17}$$

where: a_{ij} = amount of activity i allocated to zone j (measured in hectares), b_{ij} = the discounted present value per unit of activity i in zone j. The allocation of activities was constrained by:

$$a_{ij} \geq 0 \quad \text{(nonnegativity)}$$

$$\Sigma_i a_{ij} = z_j \quad \text{(total coverage)}$$

in which z_j is the size of zone j.

For certain activities additional constraints were used. In some areas specific activities were limited by production capacity. For example, bauxite mining was limited by the projected capacity of the refinery:

$$\sum_j a_{ij} (<, =, \text{or} >) Y_i \tag{11.18}$$

where Y_i is constraint limit for activity i.

In some areas activities had to be set at fixed value:

$$A_{ij} = X_{ij} \tag{11.19}$$

where X_{ij} represents the amount of activity i to be located in zone j. This constraint was needed for two purposes. First, it specified the land to be flooded in either of the two dam proposals. This activity had a negative net present value due to the evaporation of water from the dam surface. Second, it was used to set aside land with unique flora and fauna, since attempts to value them proved too difficult. The level for a component of the merit function, in order to obtain salinity value for the river, must be less than, equal to, or greater than a present value:

$$\sum_{ij} b_{ij} a_{ij} (<, =, \text{or} >) W \tag{11.20}$$

11.3.3. Model evaluation

A large number of different assumptions about coefficient values, land use options, and water exploitation strategies were examined in the study. The most interesting of the solutions computed are those comparing three options:

(1) Existing land use without a dam.
(2) The one-dam version in which agroforestry is used as the method of achieving suitable stream salinities.
(3) The two-dam option with diversion of the saline headwaters.

In the absence of water resource development, present land use is close to the optimal allocation of agriculture and forestry.

The overall conclusion was that unless a very efficient method of reducing stream salinity were to be found for the catchment, other methods of satisfying Perth's water demand should be considered. The inclusion and optimal allocation of bauxite mining increases the overall net value, but it does not alter the relative merits of the water development projects, compared with no development.

The issue of model validity has several aspects. Those parts of the Murray model that estimate water yield and salinity from precipitation, land form, and land use are simple system models which can in principle be validated only for the existing situation against existing data. Here, of course, arise problems related to estimation of biophysical data and fluctuations occurring in natural phenomena. The authors admit that it would be advisable in the case of Murray River to improve the yield and salinity data before major decisions are based on the model results. The criterion of comprehensiveness for the land use planning objective set, was, however, satisfied in this model.

The preconditions to practical application of the model are that the purpose, objectives, and methodology should be clearly described; the assumptions and value hypotheses within the model made clear; and the effects of changing them clearly pointed out. This study appears to be very well documented. The authors prepared several reports on the project for various audiences and at various level of detail ranging from the general information for the public to one full report. Therefore, it is expected that the model structure developed for this project can be used to study similar problems in many other places.

It was a great asset to the project that the model TOPAZ described in the previous section was readily available on CSIRO's national computer network and only minor modifications were necessary to adapt it for catchment purposes. Collection of data required for this model, however, tends to make this type of project expensive. The possibility of errors in planning could be reduced but not eliminated by collecting more information in several areas. Of course this would raise the costs even higher. The same observations hold for the mathematical model. A dynamic and/or stochastic programming approach might provide better, more realistic, and more reliable results. However, these would be excessively expensive in comparison with the improvements they provided. In conclusion, scientists in this project seem to have attained a reasonable balance between the degree of elaboration and data collection and the costs of obtaining usable results.

11.4. Possibilities of Utilizing the Biomass in Hungary

11.4.1. Introduction

The Hungarian biomass project aimed at clarifying the internal relationships of the biomass production–transformation–utilization system and at determining the future function and role of this system in connection with the rational use of renewable and nonrenewable resources. In investigating the system, several types of questions were addressed:

(1) How should biomass production be organized in time and space as a function of changes in the external economic conditions of the country?
(2) What nonrenewable resources could be replaced by biomass if the biomass potential of Hungary were fully exploited?
(3) What economic consequences, leading to a simultaneous improvement in the environment, could be expected if a complete utilization chain were to be constructed that gave full utilization of by-products and waste materials?
(4) How sensitively will the system respond to changes in the external conditions, such as export–import openings, and how can the detrimental effects of drastic changes be eliminated or moderated?

These questions must be answered when elaborating long-term plans for utilizing the biomass. To address them, the study considered possibilities for the development of the following sectors:

(1) Plant production, including forestry.
(2) Animal husbandry.
(3) Food industry.
(4) Processing and utilization of industrial raw materials and of the by-products and waste materials of plant production and animal husbandry.

The project was initiated and coordinated by the Hungarian Academy of Sciences. Twenty working groups were set up, several hundred experts took part in the work. The modeling team was led by Zsolt Harnos.

11.4.2. Model structure

The following procedure was selected for the solution of the problem outlined above. The description of the problem was divided into two parts: a set of scenarios specifying economic conditions through the year 2000 and a production system. This is illustrated schematically in *Figure 11.2*.

The scenarios indicate alternative possible economic conditions and goals, while the production system describes the production, processing, and utilization of the biomass. This means that expert

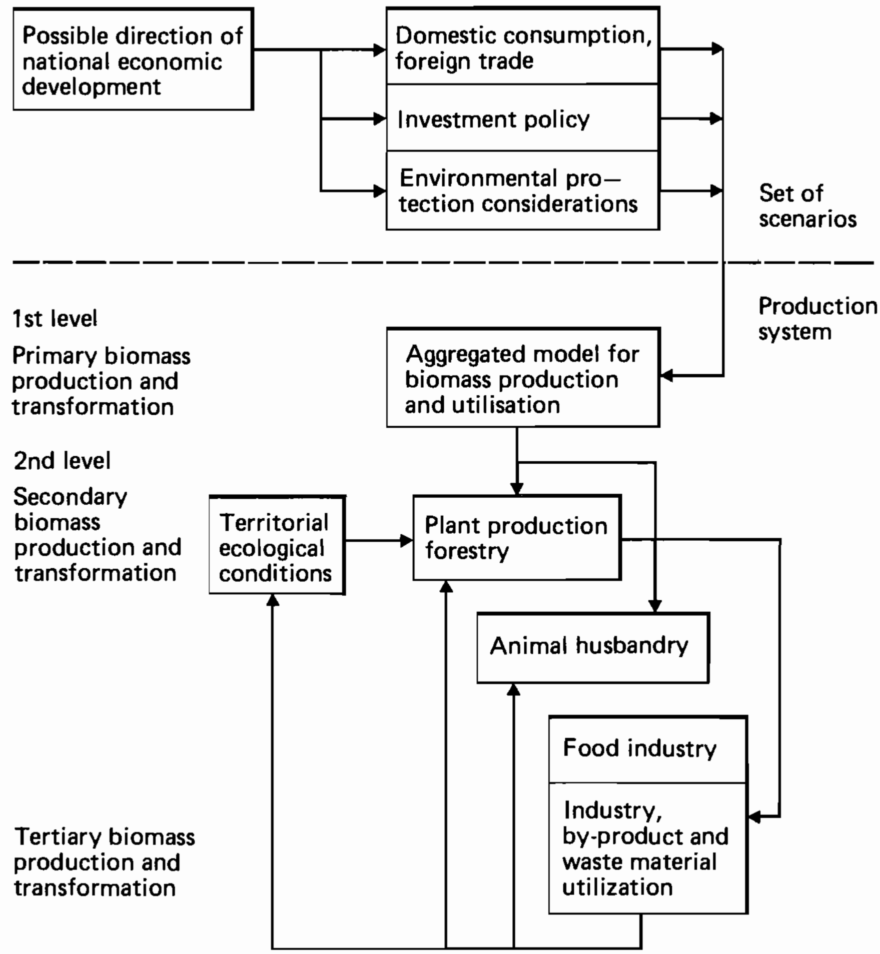

Figure 11.2. Causal structure of the model system of the biomass project.

forecasts and alternative plans for the long-term development of the national economy are used to set up a consistent system of conditions for the biomass production system. They give a broad outline of the rate of development for various sectors, the shifts in emphasis, the type and amount of resources that can be utilized, etc. The scenarios describing the alternative paths of development are defined, arranged into a consistent system and logically justified by a group of experts.

The production system is described by means of a family of models constructed in such a way that the operation of the various models of biomass production and utilization, including that of various sectors (plant production, forestry, food industry, etc.) can be analyzed independently of each other. It is also possible to link only certain sectors and thus analyze the effects exerted by their production structure on each other (e.g., plant production, animal husbandry, etc.)

The mathematical model of the system is formulated as a multi-objective control problem. The behavior of the system is basically determined by the level of plant production, that is the production of primary biomass. At any given time, the level and the product mix of plant production are controlled by:

(1) Conditions in the production site.
(2) The genetic capabilities of the species in production.
(3) The supply of nutrients.
(4) Constraints prescribed for the product mix.

Conditions in the production site are changing in time depending on the agrotechnology used and on land reclamation. The yields are basically determined by:

(1) The genetic potential.
(2) The type and state of the production site.
(3) The climatic conditions at the production site.

The supply of nutrients has a direct effect on the yields, since there must be an equilibrium between the nutrients extracted and the available stocks. The product mix is controlled by a number of factors, such as the demand for plant products and the limits for the sowing structure representing biological and production site conditions. Explicit production constraints were given for the primary products only, but the utilization of by-products was also included in the biological cycle.

Fodder needs of animal husbandry and demands of food industry affect the product mix of plant production indirectly. The sector of animal husbandry is determined by:

(1) The stock at a given time and the rate of reproduction.
(2) The mix and the quality of fodder available.
(3) The availability of shelter for animals.
(4) The demand for products of animal origin.

Sectors of plant production and animal husbandry are linked by constraints for product utilization. Processing capacity in industry and the food industry as well as domestic consumption control the biomass production cycle as a whole. Primary products are basically used for domestic consumption and exports. The possibilities of utilization of by-products are industrial raw material, energy generation, soil nutrient supply, and fodder. In addition to the demand side, product utilization is connected to plant production, animal husbandry and site conditions of the production as a feedback via the utilization of by-products and wastes.

The mathematical structure of models for plant production and animal husbandry are similar, therefore the latter one will be presented (*Figure 11.3*).

Breeding conditions are described by:

$$\begin{aligned} Bu &\le u_0 \\ x(t+1) &= Ax(t) + Cu(t) \\ x(t_0) &= x_0 \end{aligned} \qquad (11.21)$$

Dynamics in the animal stock are represented by:

$$z(t+1) = z(t) - E(t)\,z(t) \qquad (11.22)$$

where $E(t)$ reflects reproduction, deaths, and slaughtering:

$$H[z(t),\, y_P(t)] \le 0 \qquad (11.23)$$

represents the relationship between fodder need of the animal stock and the available feeds.

$$y_0(t) \le y(t) \le y_1(t) \qquad (11.24)$$

is the product output.

The precise dynamics of change in the animal stock are central in case of horned cattle and sheep, for the rate of development in these species is limited by their low reproduction factors. In case of monogastrics (pig, poultry), the rate of change in stocks has practically no biological limits. In addition to the rate of reproduction and slaughtering, the change in stock is affected by breeding conditions and the death rate. In feeding, the physiologically necessary fodder portion was determined for each species, and the composition of the stock (e.g., cow, beef cattle) and the breeding technology were also taken into account. Feed portions were expressed in digestible protein, starch value, fibre content, protein concentration, and lysine. The supply of feeding stuffs for the animal husbandry is provided by plant production, fodder imports, and the fodder coming from processing by-products and wastes of animal husbandry.

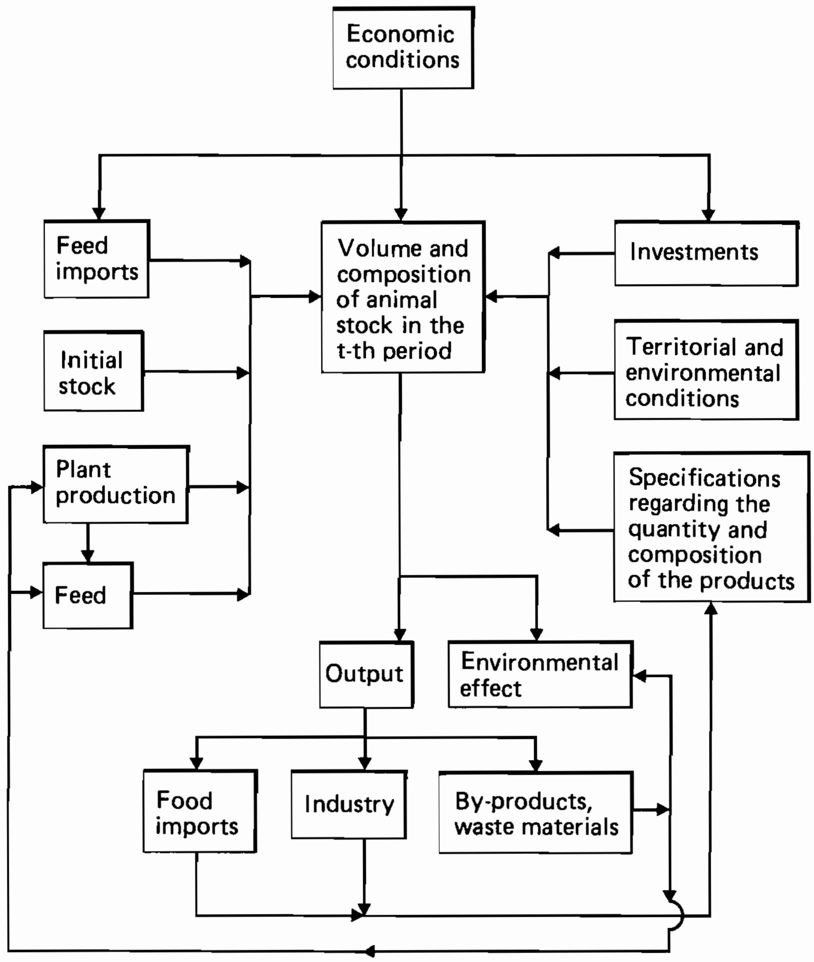

Figure 11.3. Flows in the model of animal husbandry.

Plant production and animal husbandry are linked by a system of conditions:

$$H(y_P, y_A, z_P, z_A) \leq c(t) \qquad (11.25)$$

ensuring the equilibrium between biomass production and utilization. (Here y_P, z_P and y_A, z_A represent the output and the product mix of plant production and animal husbandry, respectively.)

The optimality criteria are defined in such a way that the actual development pattern should be as close to a prescribed reference curve (development curve) as possible.

11.4.3. Model evaluation

The complex utilization of a country's biological resources (microorganisms, flora, fauna) must be an essential focus of the economic development strategy. Depending on the domestic and foreign demands and opportunities, the quality and composition of the biological production and the types of utilization should be reappraised and modified from time to time, in a manner aligned with the protection of the natural environment. In Hungary, both at the planning stage and in day-to-day practice, the rapid rate of development and the high cost of investments has led to a strong bias in favor of primary products. The possibility of utilizing biomass that is regarded as by-product or waste material has not been given sufficient attention, particularly as regards studying interactions between the utilization of primary products, by-products, and wastes. Consequently, as the study pointed out, the major change required is in present attitudes. Progress will be achieved by thinking, planning, and acting with a far greater awareness of factors governing full biological production and the natural circulation of biomass.

Certain links in the biomass production–transformation–utilization chain had already been studied in isolation. However, prior to this project, the complete biomass cycle had never been studied as a single system in Hungary. In the course of the survey, the results of all serious studies, concepts, forecasts, and surveys compiled over the past decade and dealing with any aspect of the biomass cycle were taken into consideration. Despite this earlier effort, compilation of the model system and data collection took many man-years of effort. In addition to synthesizing the results achieved in various fields into a single system, the aim of the present survey was to elaborate alternative methods of utilizing the biomass in the short and long term. In each case, these alternatives cover the full utilization of the biomass, i.e., the various alternatives are not development plans for the different "bioindustries", but are aimed at determining what biomass production and utilization structures are best suited for the simultaneous achievement of different goals (often of contradictory nature) and what type of development is to be attained without upsetting the biological and economic equilibrium.

In this study, scientists and professionals from the widest variety of domains made a concerted effort to assess the agroecological conditions of the country, to identify the natural conditions controlling the production of materials of biological origin and to estimate the prospective possibilities of producing and utilizing the biomass. This interdisciplinary effort directed attention to a number of interrelations and included them in a computer model which can be used in the future in compiling forecasts and plans at various governmental agencies.

11.5. Comparative Evaluation of the Models

These studies were undertaken in different countries, and they examined different policy issues. None the less, they seemed to escape the most common problems of model building. Namely, scientists in each study had close connection with clients and potential users, and the results had influence on decisions affecting development of the region. These two criteria appear to be obvious for any application-oriented research, but they are met in an astonishingly low proportion of projects.

Table 11.1. Comparison of the three ecological–economic models.

	Florida	*Western Australia*	*Hungary*
Clients involved	+	+	+
Results applied	+	+	+
Different scenarios	+	+	+
Social factors	+	–	–
Approach	Simulation	Optimization	Optimization
Method	Energy flow modeling	Linear programming	Linear control problem
Static–dynamic	Dynamic	Static	Dynamic
Data requirements	Medium	Large	Large

Comparison of models shows some other features in common (*Table 11.1*). Most prominent is the fact that in addition to investigating uses of different natural resources, they involved multipurpose utilization of each resource. Another common feature of these models is that different policy options were always evaluated on the basis of alternative development scenarios. This will help future applications of the models because some of these scenarios reflect events that are unexpected for the time being but can possibly occur in the future.

No one of these models is clearly the best. Each is better than the others along at least some criteria of evaluation. Two of them were used to calculate a physical or economic optimum, but they did not involve social and political factors in detail. On the other hand, the Florida model does not involve economic optimization, but it does simulate impacts of various decision alternatives on different social factors. In case of optimization models, however, the classical problem arises: different interest groups have different optima, and the final outcome usually depends on their relative power and not on any optimal solution of the model. There is no mechanism indicated in either case that would be able to force policymakers of different interest groups to follow the solution computed in the model.

It is only the simulation model that shows the internal dynamics of the system. In case of the Western Australian model, future values of different resources are discounted to their present value using the standard discounting equation:

$$P = \sum_{j=1}^{m} S_j (1+i)^{-t} \qquad (11.26)$$

where P = sum of present value of costs (or benefits); S_j = cost (or benefit) j incurred in year t; i = interest rate; m = number of individual costs (or benefits).

This approach will introduce serious uncertainties into the evaluation of model results (e.g., how to measure the future value of recreation, how to set the long-term interest rate, etc.). The Hungarian model is forced to follow so-called reference curves, but the time increment is 5 years. Therefore, only major structural changes can be followed in the results. The largest data base was required for the Hungarian model. It was a great asset that there had been long time series already compiled. Estimation of the 54 exogenous constants in the Florida model, however, seems to be very difficult. Some of them are dimensionless and represent, implicitly, the interrelationships of fairly obscure phenomena.

Despite their shortcomings, these models are good examples of ecological–economic models investigating multiple resource systems. Each of them could be improved, but it is impossible to make a precise judgment whether the costs involved in improvement would be reflected in more useful results. They are basically good models in their present form. They illustrate a mode of decision making that can help to raise the harmony between man and nature.

ECONOMIC-ECOLOGICAL MODELING
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CHAPTER 12

Economic–Ecological Models in Regional Total Systems

S. Ikeda

12.1. Introduction

The consideration of distributional aspects in regional environmental management has received widespread attention in the recent years. A rise in the level of environmental amenity in some region may cause disamenity or impaired environment in other regions, if there is a lack of total-system viewpoints in the policy evaluation. For example, interregional transfer of toxic wastes or pollutants from various socioeconomic activities, including those with pollution controlling facilities, has been one of the most conflict-ridden issues over regional and international boundaries (Thoss and Wiik, 1974; OECD, 1981). Within the region, antagonism between beneficiaries and losers who may have direct or indirect increase of benefits or damages, respectively, gives increasing importance to the distributional consideration in the planning models in view of diverse nature of ecological responses depending on the local specific circumstances.

Thus we cannot provide practicable policy alternatives to the decision makers without showing possible distributional outcomes of those alternatives on the basis of regional spatial conditions. Most decision makers in the regional levels are much more concerned with a spatial discrepancy of impacts in terms of equity and efficiency of both social welfare and resource allocation schemes (Russel and Spofford, 1977; Nijkamp, 1980). This is especially true where public goods are involved in the management models. This is one of the reasons why we need to emphasize the importance of building the "economic–ecological models (E–E models)" in the regional total systems framework.

This chapter deals with the "E–E models" in the regional total systems framework that are concerned with analysis of complexity in the interrelated processes among socioeconomic, biological, chemical, and physical components in the spatial setting. The scope of the regional framework includes such development and conservation problems associated with ecological components as:

(1) Land use management.
(2) Ecological resource utilization.
(3) Pollution control scheme.
(4) Conservation of open space, landscape, historical heritages wildlife, vegetation, and so on.

Kinds of models and modeling approaches examined in this chapter are primarily selected on the basis of the Japanese experiences in this field. This would be served as one of the typical examples undertaken in the countries under the rapid industrialization and urbanization for the recent quarter century. Because of urgent necessity of having policy evaluation tools for environmental pollution it is said that control in the days of environmental awareness of 1970s, some hundred of models have so far been produced in various levels in the field of environmental pollution problems in Japan. These models include at least either economic or ecological components. Typical use of these models was primarily to simulate or estimate such possible pollution phenomena as are caused by air, water, and industrial and urban wastes discharged from the industrial and public sectors in the regional scale.

There are, however, a few cases which dealt with comprehensively both economic and ecological elements in the sense of regional total systems view. Examples of these modeling efforts are:

(1) "Industrial–ecological models for evaluation of industrial policy" supported by the "Ministry of International Trade and Industry".
(2) "Environmental management models in the Special Research Programs on Environmental Sciences" supported by the "Ministry of Education".
(3) "Eutrophication management models of freshwater area in comprehensive studies on Lake Kasumigaura basin" supported by the "Environmental Agency".

In the following section, model structure, modeling approaches, and policy issues addressed by these models will be surveyed briefly, and then comparative evaluation of these models will be conducted together with discussion on the future improvement of the "E–E models" in the regional total system.

12.2. Conceptual Structure of E–E Models in Regional Total Systems Framework

In order to facilitate the comparative evaluation of the models in the regional total system, let us first set up the conceptual structure of the "E–E Models" in the regional total systems framework. *Figure 12.1* is an example of such conceptualization of the existing Japanese models with reference to worldwide works in this field.

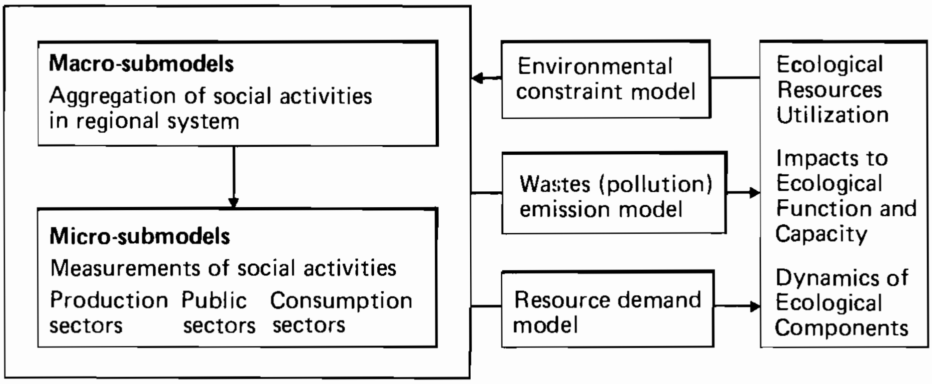


Figure 12.1. Conceptual structure of E–E model in regional total systems framework.

The socioeconomic model in the left-hand side consists of a "macrosubmodel" and a "microsubmodel" which are to be hierarchically connected. The macromodel describes the economic, social, and demographic indicators by aggregating various human activities resulting from the interacting social units (actors) residing in the concerned area. The whole area could be divided into several subregions from geographical, administrative, and other reasons in accordance with the availability of macro data on those indicators. The aggregated value to those indicators are transferred to the "microsubmodel" as the exogenous condition.

The microsubmodel gives the levels of socioeconomic activities associated with the following fundamental social units:

(1) Production sectors.
(2) Public sectors.
(3) Consumption sectors.

It is in this submodel that the land use pattern, levels of public investment for social infrastructure or pollution control, recreational demand by household, etc. are estimated either in "monetary terms" or "resource terms (material balance or energy balance)" (Kneese *et al.*, 1969). The demand and supply of private or public goods are subject to environmental/ecological constraints in the spatial setting of the concerned region.

The "ecological model" in the right-hand side of *Figure 12.1* simulates the biological, chemical, and physical interactions taking place in such spatial environments as air shed, river basin, lakes, reservoirs, coastal zones, or some combination of these environments, where the ecological goods are utilized by human activities or are impacted by receiving wastes and pollutants discharged from socioeconomic activities. It is this ecological model that needs a great deal of non-homogeneousness and nonlinearity in the dynamic behaviors of ecological components. For example, the recent rapid deterioration of water quality (eutrophication of water body) in semiclosed river basin, bays, or coastal zones surrounding urbanized areas requires further consideration of nutrient loadings (nitrogen, phosphorus, and other organic matters), besides the total quantity of nutrient loadings (Ikeda and Adachi, 1976), with respect to:

(1) Where nutrients come into and flow out.
(2) From what sources they originate.

This information is essential for the ecological models not only to predict algae concentration but also to estimate the control costs and damages to the fishery, recreation, irrigation, as well as city and industrial water use incurred by the frequent occurrence of phytoplankton blooms.

The middle section in *Figure 12.1* is composed of:

(1) Resource demand model.
(2) Wastes (pollution) emission model.
(3) Environmental constraints model.

These work as an interface between the economic and ecological models. The first one is to calculate the demand for ecological resources or commodities which are to be utilized by socioeconomic units when the production and consumption levels are given by the "microeconomic model". The second one is to estimate the amount of wastes and pollutants to be generated and to be discharged into the spatial environment after cutting down by means of adequate control scheme. The third one is to evaluate regional environmental capacity of each ecological commodity to maintain a sustainable level in regional spatial setting. The focal point in this model is how to set such threshold levels of sustainability or ecological stability in regional scale. Here it seems to be clear that we need enough scientific and technical information on the distribution of ecological components to determine these environmental constraints. *Table 12.1* illustrates some examples of the ecological components which are taken into consideration in this interface section for the case of coastal resources management model (Ikeda and Nakanishi, 1982).

Table 12.1. Illustrative examples of ecological components of E–E model in coastal management.

Man's resource activity	*Fishery (fish catches, farming)*	*Recreation (swimming, picnicking, fishing)*	*Transportation (harbors, ship service)*	*Industry (industrial bases)*
Ecological resource demand	Fish population Planktons Feeds	Fish populations Coast vegetation		
	Shoal water area Underwater grass	Natural coastline Beach/shore Water quality	Bay/inlet Open sea	Coastline Shallow Open sea
Wastes and pollution	Nutrients	Oil spill Wastes	Oil spill	Wastes Nutrients Oil spill
Environmental constraints	Fish stock Open sea	Coastline Beach Water quality	Bay/inlet	Coastline Shallow Open sea

The major policy issues to be addressed in the present model framework are:

(1) To predict primary and secondary effects and reactions which might be brought about from both the ecological resource utilization and the wastes or pollutants.
(2) To estimate the benefits and damages incurred by adopting alternative combinations of control instruments in terms of economic or ecological indicators.
(3) To bring the comprehensive or total system's views into the regional environmental management plans beyond both traditional economic and ecological evaluation with aggregated costs or efficiency.

12.3. Industrial–Ecological Model

12.3.1. Introduction

This research project was initiated in 1971 by the Ministry of International Trade and Industry (MITI) to respond to the national need of achieving harmonious industrial development with environmental preservation confronting severe pollution problems during the latter half of 1960s (MITI, 1972). It seems to be quite clear that the MITI needed to develop adequate industrial policies or guidelines of shifting gradually toward such an industrial structure as generates less wastes and pollutants to mitigate heavy burden to the environment. The prime purposes of this project, which has taken different shapes over the years, were to answer the following questions:

(1) What should be industrial activities in accordance with ecological principles in spatial environment?
(2) How should such industrial–ecological models be built?
(3) What are the industrial policies to meet various demands in pluralistic society on the basis of dynamic harmonization between human activities and nature?

The members of the project formed a multidisciplinary team (ecology, economics, earth-science, medicine, system engineering, urban planning, etc.) and reached some fundamental concept called "industrial ecology", which was stated as:

> Comprehensive analysis and evaluation of dynamical interrelation between human-activities of industrial development and the environment.

This was quite a new idea at that time, particularly in the authoritative body responsible for industrial development, although whether their idea and philosophy have finally been implemented or reflected in actual decision making is another matter to be examined. However, their initiated concept has been elaborated in various submodels and found reasonable places not only in the multiregional models of macro type, but also ecological models in regional total system (Industrial Policy Research Institute, 1976–1980).

12.3.2. Model structure and modeling approach

Figure 12.2 illustrates the overall structure of the "industrial–ecological model, which consists of a multilayered system of models.

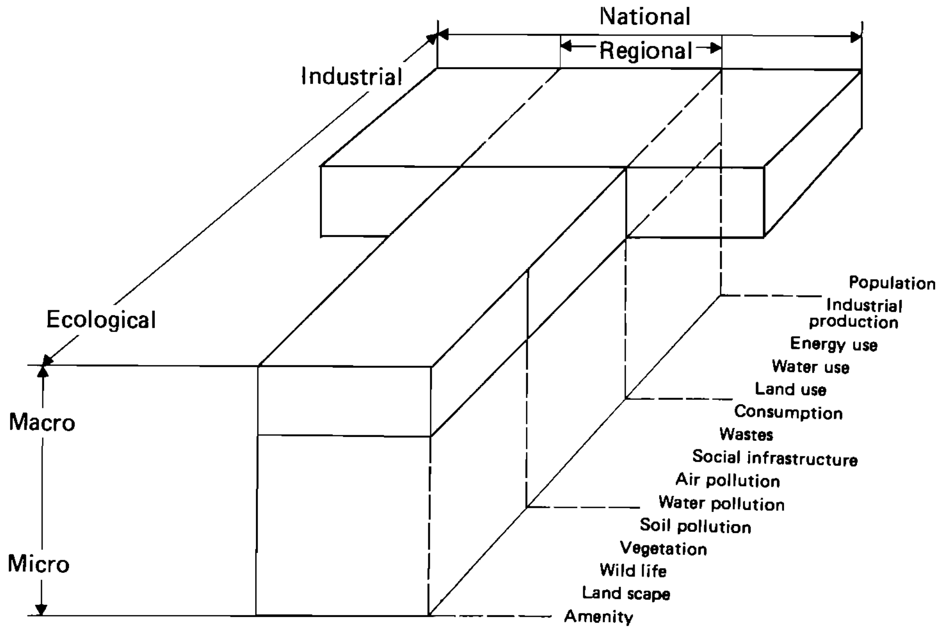


Figure 12.2. Multilayered structure of industrial–ecological model.

Only the macro model of the "Kanto Region" which includes the Tokyo metropolitan area will be introduced in this chapter. Other industrial–ecological models in specified regions and their submodels of ecological components are constituents of the "system of industrial–ecological models" as shown in *Figure 12.2* (MITI, 1972).

The macromodel in regional setting utilizes energy and material flow in the form of an "input–output table". The particular feature is to include explicitly natural and ecological resources, and pollutants in the "input–output table" as is now common in Leontief's model for pollution control (Leontief, 1970):

Production $X = AX + F$ (to be produced for intermediate/final use).
Resource $Z = BX + W$ (to be used for production and consumption).
Pollution $Y = CX + G$ (to be discharged into the environment).

Here, X, Z, and Y are vectors of industrial outputs, consumed resources and pollutants, respectively, and F, W, and G are vectors of final demands, amenity demand by households and wastes disposal by households, respectively. The matrices A, B and C represent consumption or generation coefficients by production sectors.

Then, amount of ecological resource Z and pollutants Y are taken into consideration in the ecological models at the lower layer of *Figure 12.2*. Finally, the pollution levels of air, water, soil, and so on are to be analyzed in reference with some "environmental quality standards" set as constraints for industrial development policies.

12.3.3. Model evaluation

It seems that the initial plan to develop the "Industrial–Ecological Model for Industrial Policy" is quite ambitious in terms of its scale and depth. Unfortunately we do not have enough measures to evaluate its performance and practicability when they are really applied to industrial policy formulation. Some simulation results, however, in "Kanto" and "Kansai Regions" (those are two major industrial centers in Japan) were partly taken into consideration in determining their industrial policy on the regional industrial structure.

Table 12.2. Comparative ranking of policy alternative (MITI, 1972).

Policy type	*Policy issues* (1)	(2)	(3)	(4)	(5)	*Economic growth*	*Environmental quality (GRD)*	*Required investment for pollution level*
Standard	+	+	+	+	+	4	2	2
No pollution control	+	+	+	–	+	3	5	5
Private sector	–	+	+	+	+	2	4	1
No migration	+	+	+	+	–	6	1	3
Industrial structure	+	++	+	+	+	4	2	4
Private sector and no pollution	–	+	+	–	+	1	6	5

(+), positiveness for each policy issue.
(–), negativeness for each policy issue.
(1)–(6), relative order among six policy alternatives – same number means equal rank.

Table 12.2 illustrates an example of the model simulation for the following industrial policy issues:

(1) Increase of the final demand ratio B/A between private investment (A) and public investment (B) for construction of social overhead capitals.
(2) Shift of industrial structure from the resource-oriented type to the knowledge-intensive type.
(3) Shift of consumers' behaviors from goods to services.
(4) Tightening pollution control schemes.
(5) Steady population migration from other regions.

These policy questions are summarized by the order of favorability to the six scenarios in terms of "economic growth", "environmental quality level", and "required investment for pollution control" in which "standard" type of industrial policy was the most visible course at the first half of the 1970s.

It is quite understandable that in order to get more quantitative and distributional responses of ecological components, this project has recently concentrated its effort to the elaboration of the detailed ecological submodels on the basis of "ecological data base" or "ecological maps" for exploring a dynamic harmonization between nature and highly industrialized society (Industrial Policy Research Institute, 1980).

12.4. Environmental Management Models in Regional Total System

12.4.1. Introduction

The "Special Research Program on Environmental Science" was created in 1977 by the Ministry of Education, Science, and Culture to support the basic and comprehensive studies in various fields of environmental science. The scale of the grant to aid the researches was, for example, about 950 million yen (about US$ 4 million) except salary for researchers in 1982 with over 1000 participants from governmental and private universities and research institutions. One of the specific purposes of this research program is to promote interdisciplinary research efforts and discussions on the relationship among the environment and human welfare, the dynamics of natural environment and the technology of environmental pollution control. It is, therefore, quite natural that there are a number of researches associated partially with the "E–E modeling" in the following fields involved in the Special Program (Takahashi, 1982):

(1) Ecosystem research.
(2) Human health effects.
(3) Technology for environmental management.
(4) Concept and methodology in environmental studies.
(5) Environmental monitoring.

However, there are not many studies on "E–E modeling" in the regional total systems framework. Since the major purpose of model building is directed to the scientific analysis and evaluation between socioeconomic activities and their impact on ecological environment, it requires a great deal of work for the scientists in academism to collect the necessary data and information in order to carry out empirical studies on these E-E models. Thus, some of the examples described in the next section are rather conceptual and have not yet been applied directly to policy formulation in the decision making bodies. The first and second examples are concerned with the models of the land-marine integrated development in the semiclosed sea area (Nishikawa *et al.*, 1980; Ikeda and Nakanishi, 1983). The second one is a successor of the former. The objectives of these models are:

(1) To understand the interactions between coastal land-use plans and marine-use plans that used to be administrated by different organizations scattered in various levels.
(2) To assess the dynamic changes of marine resources in terms of social costs (benefits and damages) for their utilization and preservation.
(3) To aid for decision makers in regional levels to grasp distributional figures of ecological commodities in space and sectors.

From the methodological viewpoint, these models consider explicitly dynamics of socioeconomical change, and bring nonlinearities of eutrophication dynamics into the coastal ecosystem (Ikeda and Yokoi, 1980). A variety of mathematical methods such as "linear programming (LP)", "systems dynamics" are developed in this ecological modeling area either in terms of "material balance" (Kneese, *et al.*, 1970) or "energy-balance" (Zucchetto and Jansson, 1979).

12.4.2. Model structure and modeling approach

In the coastal ecological model of *Figure 12.3*, the importance of nonlinear and dynamic characteristics of eutrophication process urges us to construct a "system dynamic (SD)" model rather than a static "input–output" model. *Figure 12.4* illustrates a conceptual structure of a marine ecological model for "Seto Inland Sea" which consists of ten components ($E_1 - E_{10}$) in three trophic levels measured by "biomass" of "organic carbon" that are originated from the primary production of phytoplanktons. Although the increase of nutrient loading into the coastal zone brings an increase of the primary production by phytoplanktons which uptake nutrients for their photosynthesis, the excess nutrients loading may destroy stable structure of marine ecological system with complexity of fish species. This semiclosed sea used to be one of the most affluent fishing banks in terms of productivity and variety of fish populations, but has been eutrophicated up to such a vulnerable level that it has a frequent occurrence of "red tides" (algae bloom) with mass death of fish under cultivation and a drastic decrease of fish population of higher economic value in upper trophic level.

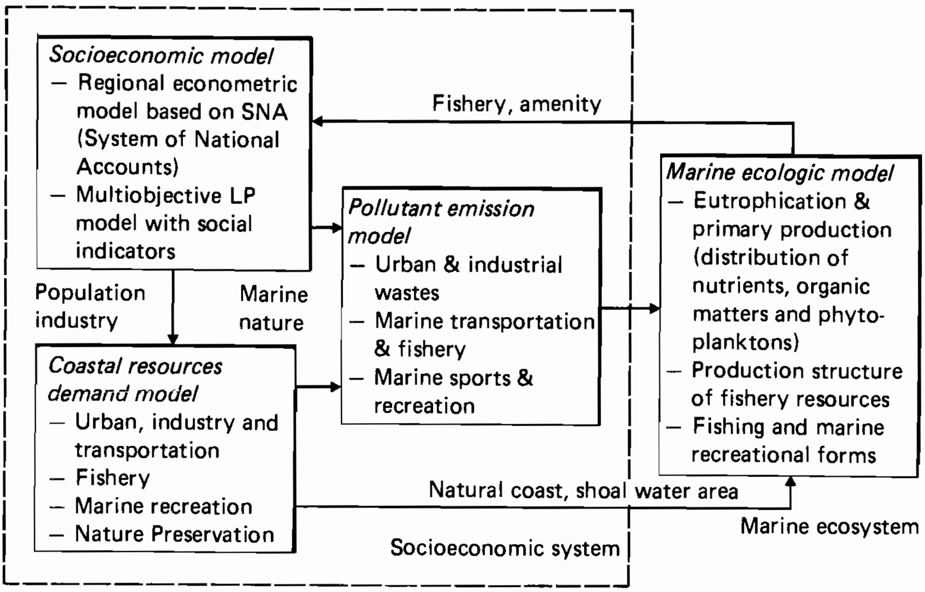
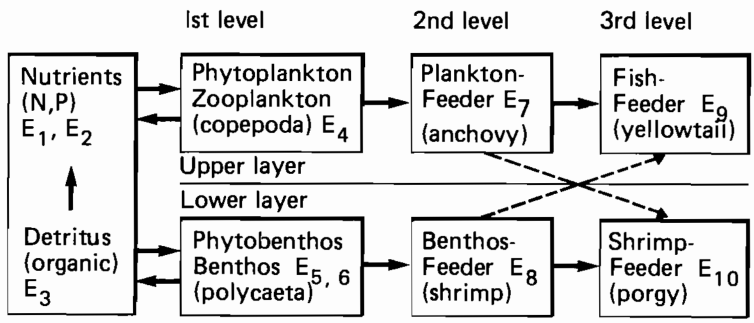


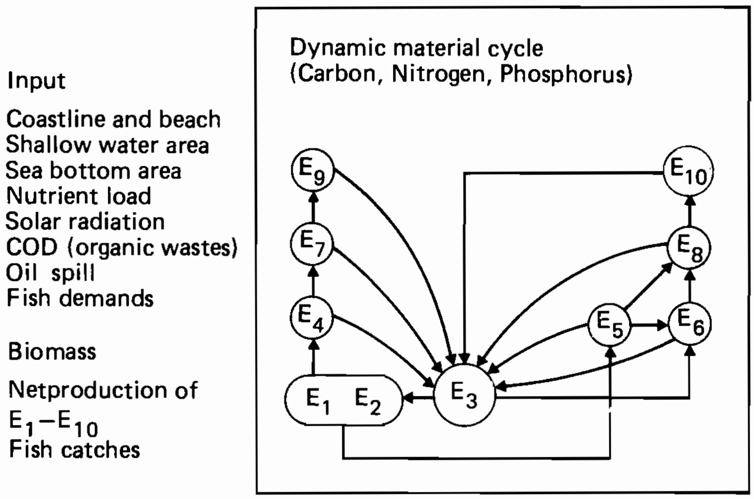
Figure 12.3. Overall structure of the coastal E–E model.

The interaction between the socioeconomic model and marine ecological model is analyzed by a relatively small number of elements in the interface of the "E–E model", i.e., "pollution emission model" and "coastal resources demand model" of *Figure 12.3*:

(1) Inflows of the nutrients (nitrogen and phosphorus) and pollutants such as COD and spilled oil.
(2) Physical destruction of the coastal zone by land reclamation and dredging for extension of the land area for industrial bases or urban uses, and construction of harbors, marine sport bases, etc.



(a) A simplified production structure in the Seto Inland Sea



(b) SD framework of marine ecological model

Figure 12.4. Structure of marine ecosystem in Seto Inland Sea.

(3) Harvest of marine products, both by fishery and by farming.
(4) Recreational use of coastal resource.

The overall resource allocation is carried out by the socioeconomic model. The socioeconomic model of coastal zone is an LP model which has a kind of multiobjective function associated with social welfare in regional setting. It has two types of constraints, that is, hard and soft constraints. The hard constraint is the one which is fixed rigidly by physical capacity of resource of exogenous conditions. The soft one, on the contrary, can be set in accordance with human or societal preference due to a variety of demands by sectors. These are, for examples, income, public services, demand for amenity, and recreation, and are calculated on the basis of SNA which is a System of National Accounts proposed by the United Nations in 1968.

The second model is an simplified version of the former model, and aims to work in a more operational way to reach practicable results in terms of cost efficiency of model building. *Figure 12.5* shows overall structure of the economic–ecological model for marine resource utilization and management of eutrophication (Ikeda and Nakanishi, 1983).

The major simplifications are as follows:

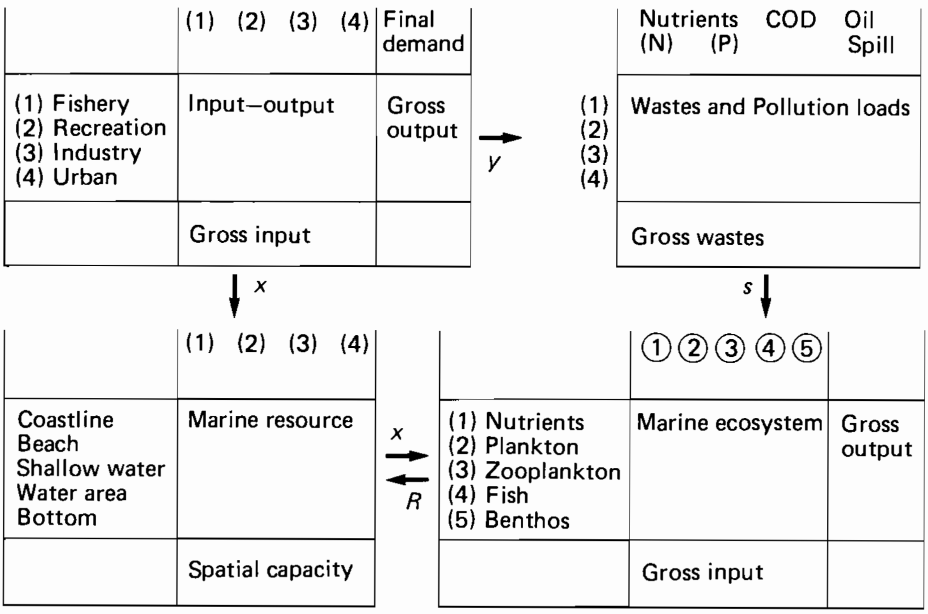


Figure 12.5. Structure of the simplified coastal E–E model.

(1) Instead of having utility and production equations of econometric model based on the SNA indicators, a kind of input–output table is set up to assess the human activities associated with marine resource utilization which includes specifically, fishery, recreation, and transportation as well as industrial and urban activities.

(2) The evaluation of such policy issues as defined in the case of former model will be carried out through the conversation type of communication between scenario writing and policy alternatives on the basis of "environmental economics" or "welfare economics" in order to mitigate the possible environmental externality, rather than by the use of LP optimization scheme of multiobjective type.

12.4.3. Model evaluation

From the theoretical viewpoint of environmental economics, our "E–E model" of coastal resource use can be formulated by the following equations of social utility maximization (Fischer, 1980):

Maximize $U(u_1, u_2, u_3, u_4, \mathbf{s})$:	societal utility	(12.1)
$u_1 = u_1(x_{11}, x_{12,\ldots,} \mathbf{s})$:	utility of fishery industry	
$u_2 = u_2(x_{21}, x_{22,\ldots,} \mathbf{s})$:	utility of recreation and amenity	(12.2)
$u_3 = u_3(x_{31}, x_{32,\ldots,} \mathbf{s})$:	utility of industrial production	
$u_4 = u_4(x_{41}, x_{42,\ldots,} \mathbf{s})$:	utility of urban services	

subject to:

$u_i = u_i(x_{1j}, x_{zj,\ldots,} \mathbf{s}) > u_i^*$:	utility of other sector ($i > 4$)	(12.3)
$f^k(y_{1k} y_{zk,\ldots,} \mathbf{s}) = 0$:	production function ($k = 1,2,\ldots$)	(12.4)
$\sum_j x_{ij} - \sum_k y_{ik} < R_i(s)$:	constraint for ith resource ($i = 1,2,\ldots$)	(12.5)

where $\mathbf{s}$ is a vector of environmental pollutants that bring environmental externality into each sector's utility, x_{ij} is a vector of the ith resource or good consumed in the jth sector, y_{jk} is a vector of the ith goods that is used or produced in the kth sector, and u_i^* is a minimum requirement for the other sectors.

However, since we have not established yet a workable and practicable utility measure to estimate values of environmental goods or services such as recreation and amenity, it is almost impossible to carry out such maximization as defined above. Even if we could succeed in identifying the utility functions (12.1–12.3), it would be misleading for us to determine a resource allocation scheme by some automatic optimization algorithms under such a condition that there is a great deal of discrepancy in utility value between industrial products and recreational services evaluated by the current monetary measure.

The E–E model of the simplified version of *Figure 12.5* is an attempt to construct an empirical and practicable model from the currently available data, rather than from direct application of the theoretical model. In addition, the maximization procedure of (1) or the multiobjective function with hard and soft constraints for the former LP model, would be replaced by conversation type of "trade-offs" between decision makers and computer simulations in the simplified model system. It seems, however, to be far from the goals of modeling objectives in terms of applicability and practicability in the use of the E–E models to the decision making process in the context of regional environmental management. This is partly due to the inconsistency of utility measures for economic and ecological goods or environmental services, respectively.

12.5. Eutrophication Models of Lake Kasumigaura in the Regional Total System

12.5.1. Introduction

These comprehensive studies on the eutrophication of freshwater area were started in 1977 as the special project in the National Institute for Environmental Studies (NIES) funded by Environmental Agency (EA) in Japan. The project focuses specific attention on the quantitative analysis of eutrophication effects on man's utilization of lake water resource (NIES, 1977). Since Lake Kasumigaura is one of the important water resources for the Kanto regions with over twenty million people, deterioration of lake water quality makes a significant impact not only on municipal water supply, but on other uses of the lake environment such as reactional activities, commercial fishing, and fish farming.

The study topics related with "integrative model for prevention of eutrophication" within this comprehensive project are:

(1) Production and consumption activities in regional society which include treatment at source of wastes that might be discharged into the water body.
(2) Man's utilization of lake water resources.
(3) Supply of amenity and recreational services.
(4) Flood control function associated with the lake physical capacity of water level and flood plains.

The policy objectives of these modeling efforts are primarily to establish some quantitative measures or indicators to be used in estimating benefits of pollution control or in estimating pollution-caused damage to production activities in the regional total system (Nakasugi *et al.*, 1979).

12.5.2. Model structure and modeling approach

Figure 12.6 illustrates the interrelationship among the submodels in their model framework, (Kitabatake, 1981). Except the evaluation of amenity and recreational activities, the models are largely formulated in the framework of environmental economics where a production and a damage function are estimated empirically by means of econometric method. For example, the welfare cost for fish farming industry (mostly carp) is derived from the information on water quality, dose–response relationship for cultured carp (damage due to water quality), and other economic data on fisherman's household.

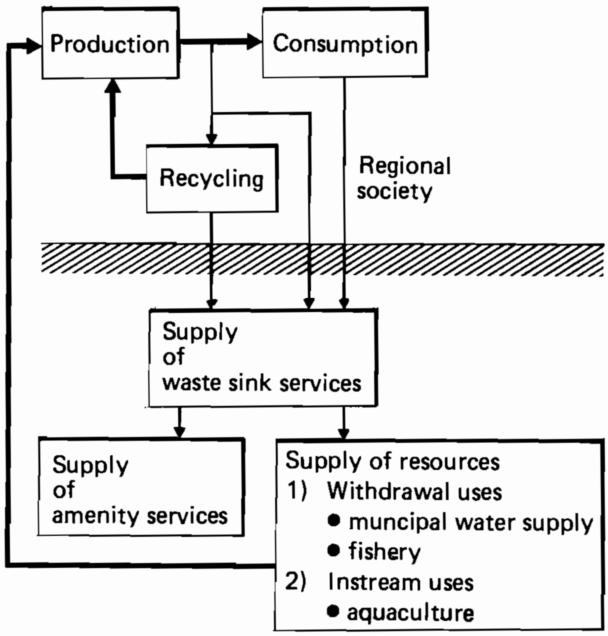


Figure 12.6. Structure of lake utilization in regional total system.

The unit damage function D per output of carp farming for operating equipped with feed supply boxes is regressed in the form of:

$$D = f(u_1, u_2, u_3, Q) \quad (12.6)$$

Example:

$$- 15.67 + 0.006(u_1 - \tilde{u}_1) + 1.249(q_1 - \tilde{q}_1 - 7.082(q_2 - \tilde{q}_2)$$
$$- 4.102\delta$$

where $\tilde{u}_1, \tilde{q}_1, \tilde{q}_2$ are sample means, $Q = (q_1, q_2, q_3)$ = temperature, transparency, chlorophyll a, u_1 = capital input, u_2 = labor cost, u_3 = feed cost, and δ = dummy of operation (use = 1, no use = 0). Then the welfare cost of eutrophication-caused production losses is calculated based on the damage function D, given a marginal production cost function for operators (Kitabatake, 1982). The same kind of analysis of eutrophication effect on municipal water supply is carried out by taking account of a purification cost in the short term and a change of the intake point along lake sites in the long term (NIES, 1981). As for a recreational use of lake water resource, psychological evaluation of lake sites is attempted to have a quantitative relationship between residents' responses and physical or biological features of landscape measured on site.

12.5.3. Model evaluation

The submodels of utilizing lake water resource are intended to quantify the economic values of the benefits or damages caused by the lake eutrophication in such sectors as fishery, fish farming, municipal water supply, and recreational use of lake sites. Besides studies on these submodels, there is a plan to develop a comprehensive economic modeling for controlling nutrient loading into the lake water. Indeed, it is quite difficult and time consuming to get practicable data for building these empirical models in regional spatial scale. But, it is also true that without these empirical models based on the concept of the economic and ecological modeling, any result of policy evaluation would become illusion. The next step the NIES project is going to tackle is how to combine a model of evaluating recreation and amenity values based on the sociopsychological analysis of recreational demand with the economic and ecological submodels in the framework of regional total system.

12.6. Concluding Remarks

Three important questions on E–E modeling are raised by Russell and O. Spofford (1977) with reference to their "integrated residual management model" at the regional level (Delaware Valley Region, USA):

(1) Whether to include intermedia trade-offs in regional analysis.
(2) Whether to include nonlinear ecosystem models in regional optimization models.

(3) Whether to provide constraints for restricting the distribution of costs and environmental quality in regional environmental management models.

They gave positive answers to these questions after examining their attainment of modeling work, and then stressed that "the distributional problem of costs and environmental quality was often the central issue in regional environmental management with efficiency consideration of secondary importance."

Incidentally, none of Japanese models has been directed toward a practicable use in regional authoritative decision making. They are rather assumed to be aids for policy evaluation of long-term planning either in central or prefectural government bodies to have a scientific validation or critical information of the proposed policies and projects. In this respect, we need further development of such a regional total system model that has more elaborate interface between economic and ecological models in order to facilitate not only distributional considerations, but also the intermedia trade-offs in various regional decision making bodies.

For example, the fish farming in the Seto Inland Sea area occupies a fairly large portion of water area in good quality and in calm condition, but it eutrophicates the seawater by itself due to excessive feeding or wastes from the dense fish population. The farming of seaweed which shares over one third of fishery production in this particular area needs a relatively eutrophicated sea water, but that sea area is also competitive for recreational or industrial users. The increasing demand for recreation such as swimming, boating, sport fishing, and so on is raising a question as to whether or not we should promote such artificial farming of fish of high market value. Instead we may be able to rely on natural farming in the Seto Inland Sea as a whole by implementing an adequate eutrophication management plan for preserving a sustainable fish stock adapted for regional-specific conditions.

In order to explore such a policy issue, there is still a large gap between economic and ecological models. The gap, for instance, lies between ecospatial information on fishery stocks, their dynamics and sustainability, and economic information on fishery industry, that is, supply and demand for a variety of fishes by consumption sectors. Nevertheless, given the recent progress in ecological modeling of ecosystem and in analysing eutrophication process taking place in the coastal and bay area, it would become more and more practicable and feasible to take distributional consideration into the economic evaluation of ecological resource management.

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CHAPTER 13

Models for National, International, and Global Systems Policy

H.T. Odum

13.1. Introduction

Because humans are a small part of the global system of nations and environment of our planet, they have been slow to gain simple overviews to accompany their emerging ability to control the biosphere. Now, however, global views are finally emerging. As the traditional micro-focus of ecology and economics turns to larger scale holism, the unity of the economy with the environmental processes becomes apparent. Models to represent both humanity and nature overview the unified realm.

Systems of environment and humanity are constrained by total available energy, hierarchical relationships, material balances, and recycling, limits to energy transformation efficiency and larger-scale control mechanisms. To gain overviews of these systems, we construct models by which we organize our experience, develop principles, make predictions, and recommend actions. In recent years, mathematical models by which our overview concepts can be numerically evaluated and simulated supplement the conceptual models that we all hold in some verbal language.

In this chapter, models are presented that portray the dynamics of economic, ecological, and integrated economic–ecological systems. Similarities and differences between economic and ecological systems are indicated in diagrams, differential equations, and graphs resulting from numerical simulations of the models. The focus is on growth and development patterns, and on pulses and steady states, all in relation to the constraints set by available resources.

13.1.1. Macroscopic minimodels

Overview models of large scope but simple structure may be called "macroscopic minimodels". This type of model has long been used in macroeconomics, large-scale energetics, and systems ecology. In earlier days of the computer era, the emphasis in mathematical modeling was on combining large numbers of relationships, using the power of the computer to include many variables. Such models were very complex, and not easily visualized by the human mind, and thus not often trusted. Quantitative calibration was difficult and errors easily overlooked. A macroscopic minimodel, on the contrary, with only a few variables and parameters, has a structure that is readily understood by reading the diagram. It is designed to correspond more with the mental concepts of the real world system structure and behavior.

Macroscopic minimodels are like controlled experiments. They consider the interplay of a few variable factors at a time, while other factors that might vary in reality are temporarily held constant. Thus, the simple overview models are like the thought processes of considering alternatives, looking at one or two variables at a time. Computer simulation of macroscopic minimodels involves programs with only a few statements, which can be run on microcomputers anywhere. Various alternative scenarios may be considered through user-friendly access programs by changing values of inputs, coefficients, or structural relationships and subsequently running the model again.

Macroscopic minimodels predict developments only to the extent that the variables chosen as predominant are the correct ones for the circumstance. There is apparently a correspondence between the detail in a model and the details of the ups and downs in the time graphs generated by the model. Models with only one state variable tend to produce steady state straight lines in simulation, unless sinusoidal inputs are applied. Two state variable models can produce pulsing behavior, as observed in many real world situations. Macroscopic minimodels generate main trends, without showing the relative short-term variations. Consequently, they are of greater value for longer-range considerations.

Adding more detail to a macroscopic minimodel is one way to generate a model of moderate complexity and more detailed behavior without losing the overview structure. The alternate approach of stringing together many relationships without macroscopic structure rarely develops the holistic features that constrain large real world systems. A particular minimodel may apply both to the small scale realm of molecules and to the large scale realm of the economy. In scaling the same model up or down, the principle to remember is that both time and space go up or down together according to some function, dependent on the particular system. The same simulation and graph may apply for large and small by reassigning the magnitudes so as to keep the time in proportion to the size of the storages.

13.1.2. Energy systems diagrams

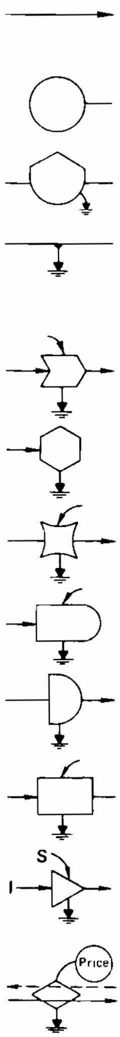
Energy language diagrams are used to portray the models in this chapter. This language has proven to be a rather simple but flexible way to visualize the structure and dynamics of the systems discussed. The diagrams in this chapter all contain only a few symbols, and verbal language is added to facilitate comprehension. This type of diagram is used because they are pictorial mathematical equations which include energy constraints. Despite this feature, their meaning and patterns of performance may be read without technical knowledge. Differential equations are given on the diagrams to help those unfamiliar with the energy language but at ease with mathematical formulations.

Also given in each figure are typical simulation graphs. Using the energy diagrams, the mathematical equations and the verbal explanations, it is usually easy to visualize the way inflows, storages, feedback actions, and outflows generate the simulation graphs. Thus, by inspection of what is in the model the reader may understand how it works, and predict the effects of varying sources or system properties. Because full explanations of the energy language and its energy, economic, and kinetic meanings are generally available (Odum, 1971b; Odum and Odum, 1976, 1982; Odum 1983b), details will not be repeated here. The symbols used are those in *Figure 13.1*. The system defined is indicated by a box frame with sources outside and components inside. Hierarchy and energy quality are represented by position in the diagram from left converging to the right.

13.1.3. Resources as driving sources in simulation models

The role of resources is an important aspect of the comparisons between models in the sections that follow. In diagramming the models, resources flowing into the system from outside are represented with the circle symbol. Examples are rainfall and foreign imports. Resources which are drawn from storages within the system boundary are represented within the system frame with the tank symbol. Examples are groundwater, coal, and information storages.

A critical aspect is whether the outside inflowing sources are flow limited or are so abundant outside that their available concentration is independent of the demand put on them. *Figure 13.2* shows the difference between a constant availability source [*Figure 13.2*(*a*)] and a constant flow source [*Figure 13.2*(*b*)] in diagrams and in mathematical representation.



Energy circuit. A pathway whose flow is proportional to the quantity in the storage or source upstream.

Source. Outside source of energy delivering forces according to a program controlled from outside; a forcing function.

Tank. A compartment of energy storage within the system storing a quantity as the balance of inflows and outflows; a state variable.

Heat sink. Dispersion of potential energy into heat that accompanies all real transformation processes and storages; loss of potential energy from further use by the system.

Interaction. Interactive intersection of two pathways coupled to produce an outflow in proportion to a function of both; control action of one flow on another; limiting factor action; work gate.

Consumer. Unit that transforms energy quality, stores it, and feeds it back autocatalytically to improve inflow.

Switching action. A symbol that indicates one or more switching actions.

Producer. Unit that collects and transforms low-quality energy under control interactions of high-quality flows.

Self-limiting energy receiver. A unit that has a self-limiting output when input drives are high because there is a limiting constant quantity of material reacting on a circular pathway within.

Box. Miscellaneous symbol to use for whatever unit or function is labeled.

Constant-gain amplifer. A unit that delivers an output in proportion to the input *I* but changed by a constant factor as long as the energy source *S* is sufficient.

Transaction. A unit that indicates a sale of goods or services (solid line) in exchange for payment of money (dashed). Price is shown as an external source.

Figure 13.1. Energy language symbols.

The constant availability provides no limits to growth, whereas the flow-limited source can support growth only up to the point where the demand pumping equals the limiting rate of supply from outside.

Examples of growth on unlimited (constant) availability of outside resources are given in *Figure 13.3*. Growth can be exponential. Examples of growth on outside flow-limited sources are given in *Figure 13.4*. Growth initially accelerates but levels as it reaches its source limited carrying capacity.

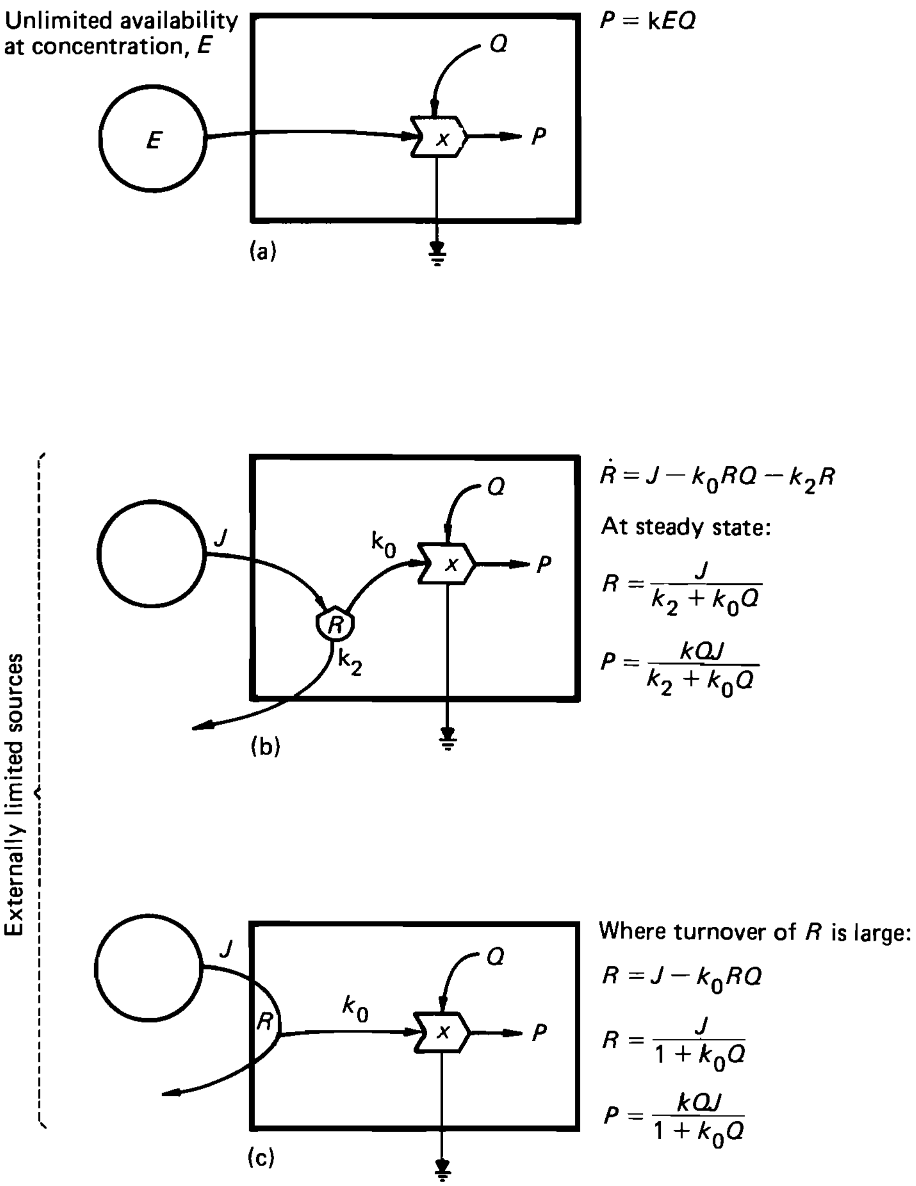


Figure 13.2. Comparison of sources (resource externalities) under demand by internal property *Q*. (*a*) Mathematical constant *E* means unlimited availability; (*b*) use limited to remainder available locally *R*; (*c*) use limited to externally limited flow *J*.

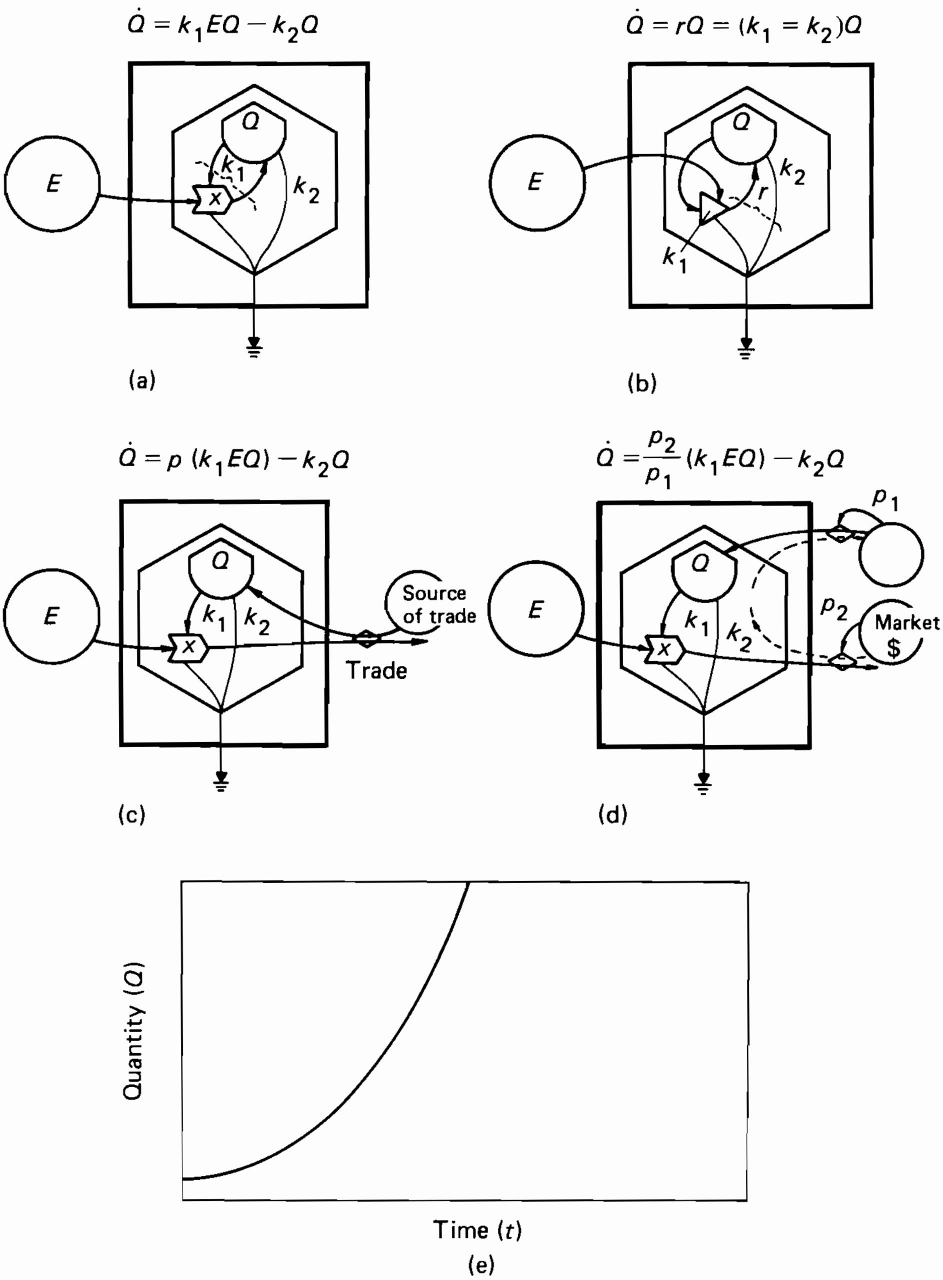
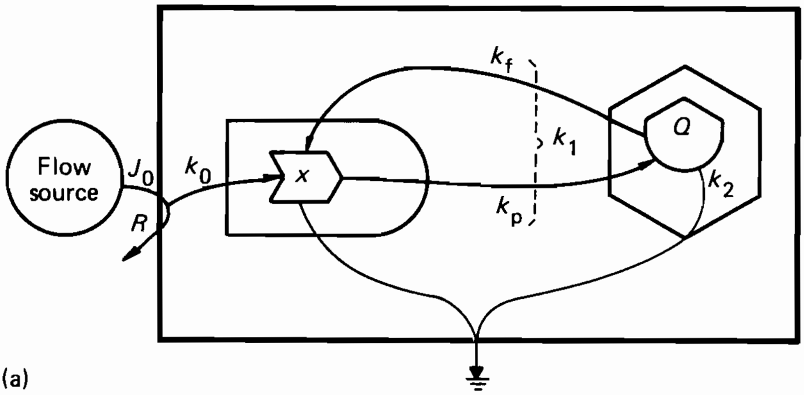


Figure 13.3. Exponential growth in ecologic and economic systems. (*a*) Constant resource availability (*E*); (*b*) independent of resource availability; (*c*) growth by barter; (*d*) growth by economic trade.



(a)

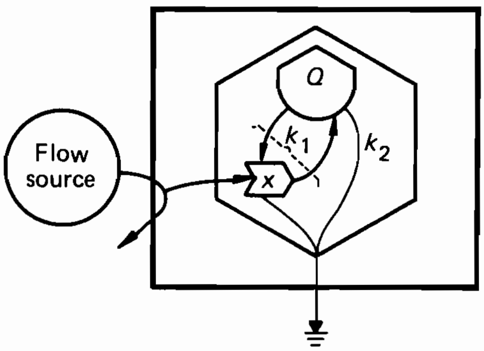
R = Remainder unused

$$R = J_0 - k_0 R Q$$

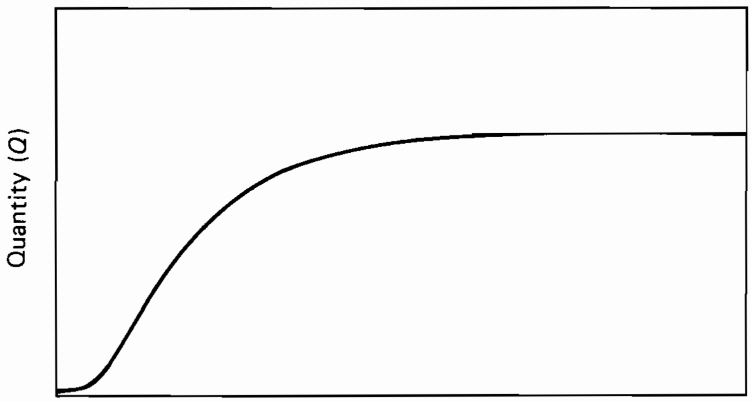
$$R = \frac{J_0}{(1 + k_0 Q)}$$

$$\dot{Q} = k_1 R Q - k_2 Q$$

Where $k_1 = (k_p - k_f)$

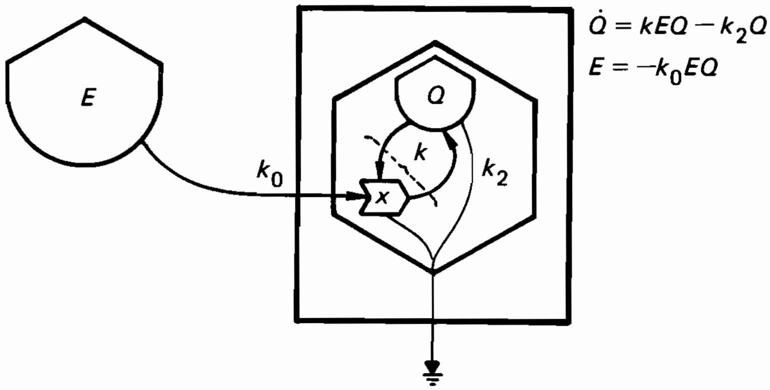


(b)

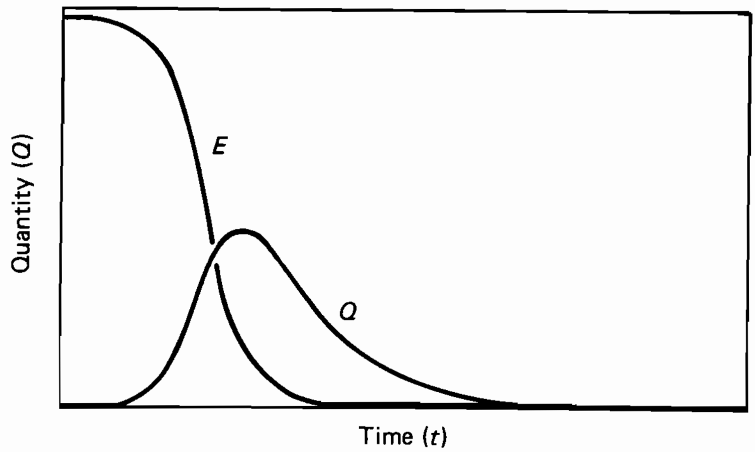


(c) Time (t)

Figure 13.4. Autocatalytic growth on steady renewable sources. (*a*) Visualized as production and consumers; (*b*) visualized as a resource consumer; (*c*) simulation.



(a)



(b)

Figure 13.5. Growth of nonrenewable sources. (*a*) Diagram; (*b*) simulation.

Where the resource is from a storage (*Figure 13.5*) that is not being replenished as fast as it is used, the resource is said to be nonrenewable within the time frame of concern. Such storage resources support only a burst of growth followed by decline as rate of supply falls below what is necessary to support enough maintenance to keep up with deterioration. This resource relationship is shown in diagram and mathematical form in *Figure 13.5*.

Growth of a "combination of renewable and nonrenewable resources" (*see Figure 13.6*) produces a peak of stored assets followed by a decline to a lower steady state. This model is useful in showing the reality of limitations to growth. However, a final steady state that is level, rather than pulsing, may not be realistic as already discussed.

In sections to follow, the energy systems language is used to represent systems models similarly for comparison. In the next section we will look at the controlling role of types of resources in controlling system development. Then concepts of *emergy* are introduced as a means to put different kinds of resources on a similar basis quantitatively.

13.1.4. Emergy and Transformity

[The names Emergy and Transformity were supplied by David Scienceman (1984), who recognized that the previously used ambiguous descriptive phrases, Embodied Energy and Energy Transformation Ratio (Odum, 1976, 1983b), were inadequate for fundamental thermodynamic properties.]

Previous use of energy concepts in models to predict macroeconomic value was reviewed recently (Odum, 1983b). Some change of terminology may now be warranted. In evaluating the resources for economic development *emergy* (embodied energy) may measure the ultimate potential for economic buying power of economic utility. Emergy is defined as the potential energy of one type of energy that must be used to generate another energy flow or storage. The unit of measure is *emjoules*. Representing all the types of energy which are used in equivalents of one type allows them to be added as a measure of total work. For example, in *Figure 13.7*, 100 joules per year of energy of type A is required in output of 10 joules of type B. There are 100 emjoules of type A in the output flow. There are 100 emjoules of type A emergy in the output.

The *transformity* is defined as the emergy of one type required to generate a joule of another type. In *Figure 13.7*, 10 emjoules of type A are required to generate one joule of the output type B. The type A transformity of type B is 10. A feedback is not counted unless it represents contribution of a source different from the one driving the transformation process. In *Figure 13.7* the feedback flow of 1 joule is not counted in calculating the transformity to type B in terms of type A because it is derived from the output.

If the actual energy of a flow or storage is known, its emergy (embodied energy) may be calculated by multiplying the quantity of energy by its transformity (energy transformation ratio). For example, the solar emergy in the photosynthetic production of a tree is the joules of wood produced multiplied by the solar transformity of wood. Transformities are obtained by evaluation of observed emergy inputs required for outputs with care to convert all the inputs into emergy units of one type. Since the resources for any system are of more than one type, the emergy supporting the system depends on more than one inflow and transformity. Emergy of one type is the common denominator for comparing economic contributions of resources of varying type such as solar energy, fossil fuels, rain, tides, human labor, etc.

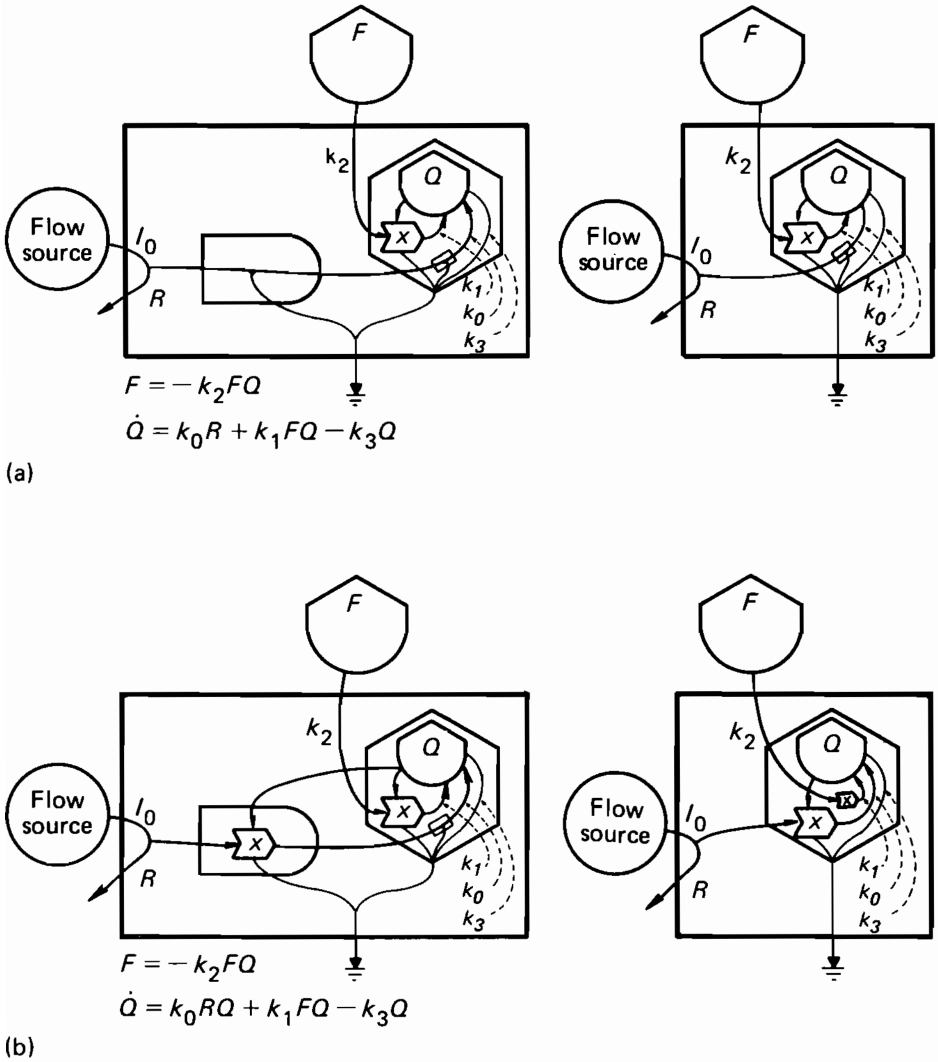
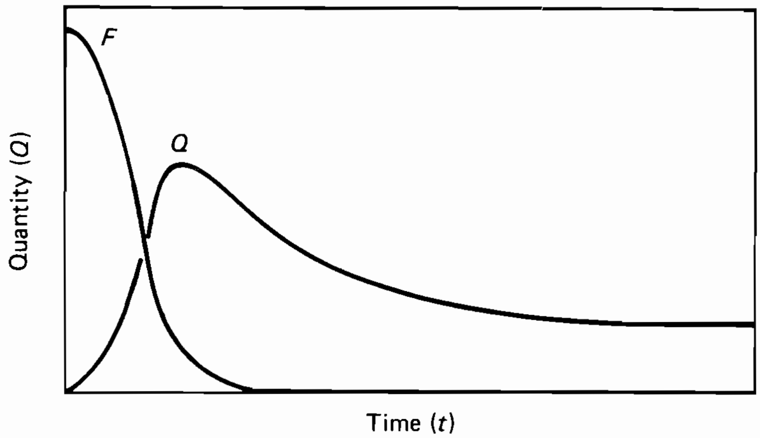


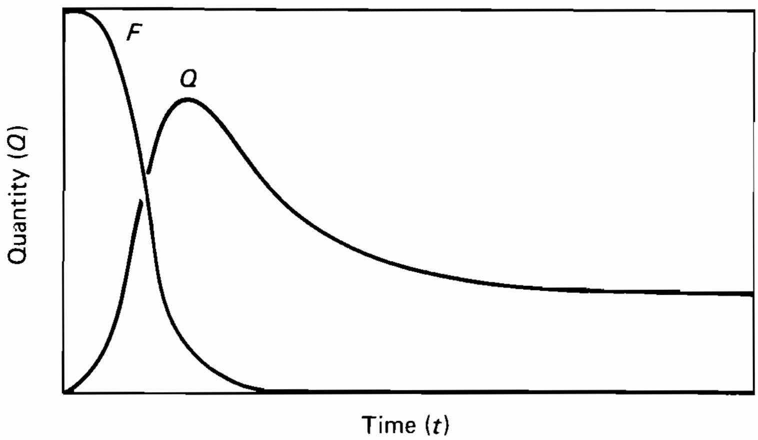
Figure 13.6. Growth of renewable (flow-limited) and nonrenewable sources. Diagrams on the left are the same kinetically as those on the right but have low-quality primary production separated from high-quality consumption. (*a*) Production is linear: (*b*) production is autocatalytic; (*c*) simulation of (*a*); (*d*) simulation of (*b*).

An economic system with circulating money has many driving resources that can be expressed in common units using emergy. The ratios of emergy required to circulating dollars is a measure of the buying power and inflation of that currency. In later sections the emergy per dollar is used to relate units of ecology and those of economics.

Next we review various kinds of models of ecology and economics.

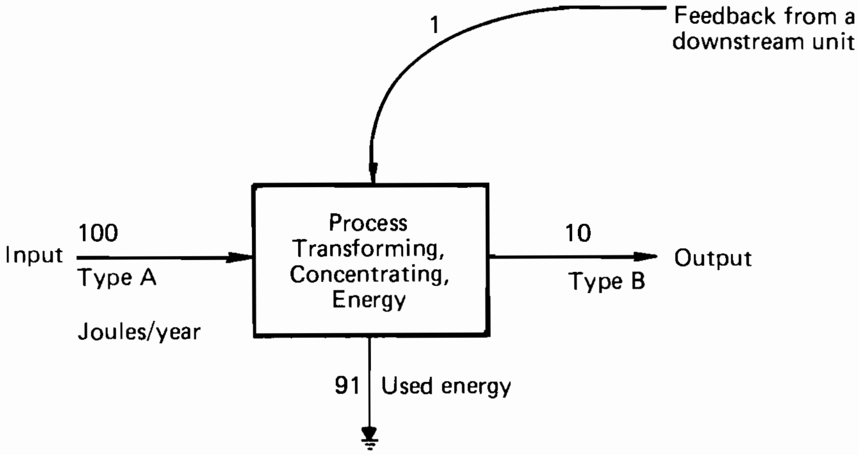


(c)



(d)

Figure 13.6. Continued.



Energy transformation ratio $= \frac{100}{10} = 10$

Embodied energy in output = 100 J/Y of Type A

Figure 13.7. Example of energy flows through a transformation process with calculation of transformity in emjoules/joule and emergy in type A emjoule/year.

13.2. A Review of Overview Models

Overview macroscopic minimodels were looked for in the literature of environmental science, economics, and ecology. Overview models found were generally complex. Fewer minimodels were applied to systems of environment and economy.

Because *econometric models* are based on empirical regressions, they were not included in this chapter; they treat as constant, energy-based relationships that change as resource availability changes.

13.2.1. Energy models

Energy has rarely been used as an overview common denominator, even though since the 1972 oil embargo, extensive energy modeling has been conducted. Most of this has been microscale models of supply, demand, particularly energy industries, conservation alternatives, etc. Most energy modeling concentrates on fuels and electricity and not so much on general energetics of environment and society. A review of five summary volumes on energy modeling (Searl, 1973; Energy policy, 1974; Roberts, 1976; Kavanaugh, 1980; Roberts and Waterman, 1981) has not provided a model in which energy is used as universal common denominator for overview models. Ayres (1978) and Edgerton (1982) have original and synthetic formulations of energy–economic principles, but without overview models.

Watt *et al.* (1977) adapted a famous minimodel used for animal populations to economic overview. Unit growth is proportional to the difference between the unit's energy use and a threshold. The total system's growth and energy consumption is the sum of the uses by the unit. This model generates a diminishing returns to energy use and negative effects of too much energy (*see* Odum, 1983a, p. 567, for an energy diagram of this model).

In another way of modeling the effects of a pattern of resource use over time, Fowler (1977) used the Gaussian Normal Curve to describe the probability of incorporating new fuel resources. The model was fitted to one arm of the normal curve representing fuel use in the past. The future use was extrapolated along the symmetrical opposite arm of the bell-shaped curve. This is tantamount to equal distribution of probability of finding oil relative to the stage of development and use. However, resource distributions are found to be skewed, and curves of demand and technological use are not symmetrical. Use is a function of systems dynamics as well as availability. Shapes to be expected may be more like those in *Figure 13.6*.

13.2.2. Complex overview models

The Forrester and Meadows models of the limits to growth (Forrester, 1971; Meadows *et al.*, 1972) are the best known of the early overview models of an economy and its resources. These are of medium complexity. In general, limits of growth were not found for developing nations until well into the next century although food shortages were found sooner. By comparison, minimodels with a more simple direct link of resources availability to production and consumption suggests limits developing now and stopping growth by the turn of the century.

The Forrester–Meadows models use many empirically determined coefficients as to the effect of one variable such as labor on others such as energy use, etc. Since these are determined now during a high-energy period, they may be quite inappropriate for simulating the future where these coefficients may turn out to be declining, resource-dependent variables instead of constants.

Others such as Boyd (1972) and Berry *et al.* (1976) found the Forrester–Meadows simulations too limiting. Boyd ran the model with increased technology representing the innovations of information and human ingenuity with the result that growth could continue without load on environment or quality of life. This simulation assumed unlimited technology without resource cost, which is fallacious, since most technology is the hidden feedback of equipment and services of high embodied energy. To develop, retrieve, and maintain information requires much embodied energy, although the actual energy content in information is small.

Barney (1980) after extensive consideration of the data on remaining world resources reviewed four of the complex models offered for national and international overview, starting with Forrester–Meadows models. Large models of several departments of the US Government, each dealing with a different sector, were linked, but each sector was mainly driven by nonmodeled assumptions about the whole and about connecting sectors.

A "Latin American World Model" allocated resources to capital investment and life expectancy rather than for consumer use. In this model and a "United Nations World Model", population growth was taken as unavoidable and the model used to show what consumption restrictions and developments would be required to prevent starvation. The possibility that sharp changes in social behavior would follow the reality of declining resources as given in many models in this paper was not allowed. The role of current human social patterns was taken as the cause rather than the continuously adapting response to the resource changes.

In such complex models, the degree to which the real resource limitations are allowed to operate or are assumed away by the functions used is not as clear as with minimodels where the relationships are readily seen by diagram inspection. Barney's document did discuss the issues in aggregate verbally, concluding that models considered may have been unreal in predicting too much economic growth. The power of the overview verbal concepts were used without the help that minimodels could have given. Meadows *et al.* (1982) summarize a decade of complex overview models with verbal minimodels.

Many complex models of energy, environment, and economics have been developed with some of the same concepts, interactions, external driving actions, etc., already given here. In these complex models many sectors of the economy have been submodeled, or input–output relationships used with disaggregation into numerous sectors. Yet when these efforts are presented and reviewed by others, the details are rarely shown. Instead, discussion concerns the main driving sources, main storages, main production and consumption functions, main mechanisms for control, etc. – in other words, the discussion and level of thinking is really at the minimodel level as readily diagrammed on a single page in the figures of this review.

For example, Mukherjee (1981) reviews overview models of economics by Hudson and Jorgensen (1974), Stanford Research Institute

(1975), Hoffman and Jorgensen (1977), and Parikh (1979). Population demography, empirical econometric extrapolations of demand for energy and resources, or demands based on human welfare criteria are given. Balance of payments and prices of foreign imports of energy and resources are varied, and effects of changes in the structure of the economy are considered such as changing input–output coefficients relating sectors. Generally, resource concentrations are not included as driving functions except through foreign price where resources are derived from external trade.

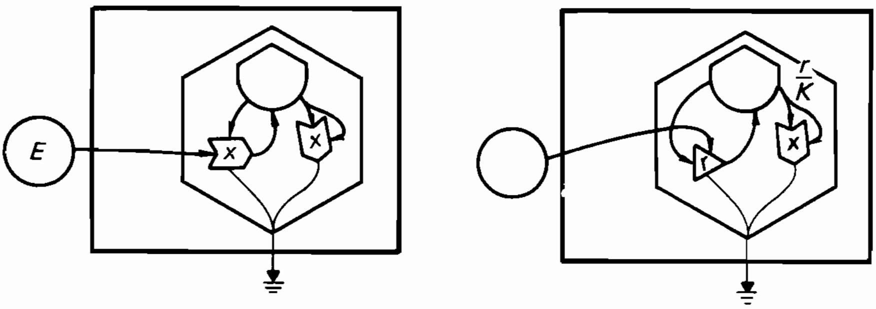
Edelman (1981) made the production functions and resultant Gross National Product dependent on environmental and energy resource base by including these in the Cobb–Douglass production function. Because these were made constants (constant availability) instead of based on a source limited function [*Figures 13.2*(*b*) and *13.4*], resources were actually made infinite. To make the complex models more realistic will require introducing the main limiting properties readily understood from minimodel simulations.

If the complex models referred to above are translated into minimodel form aggregating the separate sectors for perspective, they are found to be ramps or exponential growths reflecting the faith that human ingenuity and technology can always be substituted for resource even when there is no alternative resource to be substituted other than information.

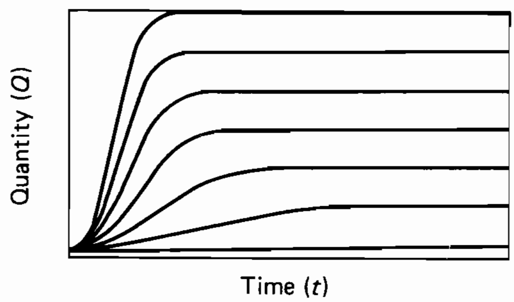
13.2.3. Logistic models

The logistic model was applied early to many growth patterns in biology and populations. See, for example, Richards' (1928) application to yeast growth where self-inhibition through accumulation of by-products of fermentation made quadratic drain appropriate. It was natural that it be suggested for nations and overall economies. Pearl and Reed (1920) applied the model to the USA, and Pearl and Gould (1936) to the world. Taagepera (1968) applied the logistic to the growth of civilizations fitting curves with growth data. The logistic curve expressed in differential equation form can be derived from many different kinds of premises. For example, 11 are given in a recent book (Odum, 1983a). Some premises have energy source unlimited but have the fraction of storage that is dispersed increase with the size of the storage (*Figure 13.8*).

In *Figure 13.8*(*a*) the standard logistic is shown in a form that recognizes the energy source (E) that many authors hide by making it constant, or embedded in constant r [*Figure 13.6*(*b*)]. *Figure 13.8* includes the equations in various forms. To test the fit of data, equation in the logarithmic form in *Figure 13.8*(*d*) is used. The effect on the asymptote of increasing energy availability (E) in *Figure 13.8*(*a*) is shown in simulation runs in *Figure 13.8*(*c*).



(a) (b)



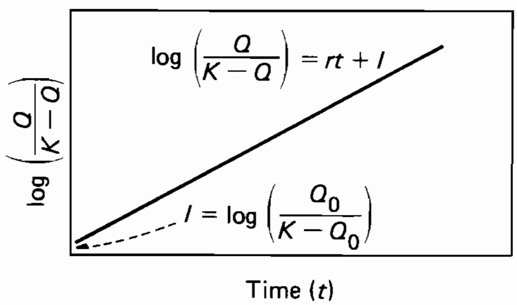
$$\dot{Q} = rQ\left(\frac{K-Q}{K}\right)$$

$$\int \frac{dQ}{Q(K-Q)} = \int \frac{r}{K} dt$$

$$-\frac{1}{K}\log\left(\frac{K-Q}{Q}\right) = \frac{r}{K}t + C$$

$$C = -\frac{1}{K}\log\left(\frac{K-Q_0}{Q_0}\right)$$

(c)



$$\log\left(\frac{Q}{K-Q}\right) = rt + C$$

$$\text{where } C = \log\left(\frac{Q_0}{K-Q_0}\right)$$

$$\log\left(\frac{K-Q_0}{Q_0}\right)\left(\left|\frac{Q}{K-Q}\right)\right) = rt$$

$$\left(\frac{K-Q_0}{Q_0}\right)\left(\frac{Q}{K-Q}\right) = e^{rt}$$

$$Q = \frac{Q_0 K e^{rt}}{K - Q_0 + Q_0 e^{rt}}$$

(d)

Figure 13.8. Standard logistic model. (*a*) With energy source (E); (*b*) with energy source assumed to be unlimited; (*c*) simulation with increasing energy availability (E); (*d*) graph to test logistic fit.

Marchetti (1975) makes extensive use of the logistic model for the growth of subsystems of energy use. The growth of whole economies is seen as the sum of a series of logistic energy systems contributing to the total economy but each starting at a later time. As shown in *Figure 13.9* the logistic is manipulated into a form with the expression as a linear function of time. So long as the logistic curve fits the data, the plotted points form a straight line, after which some other energy

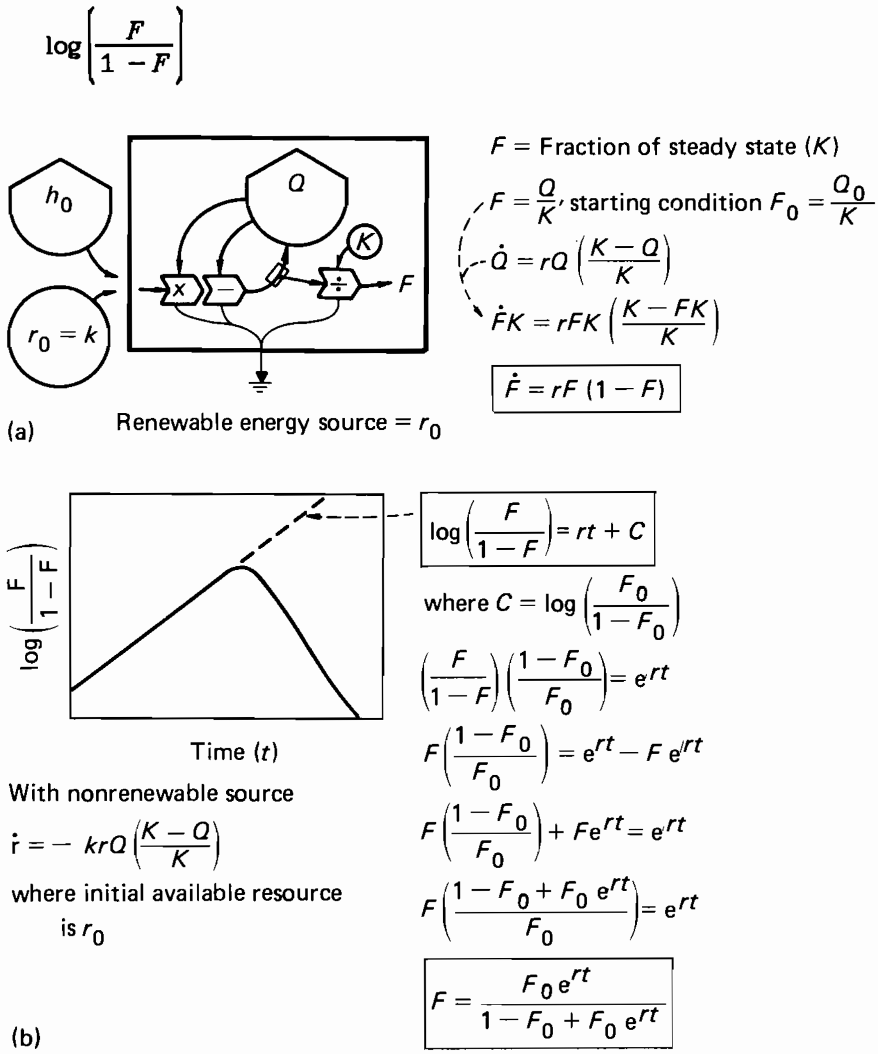
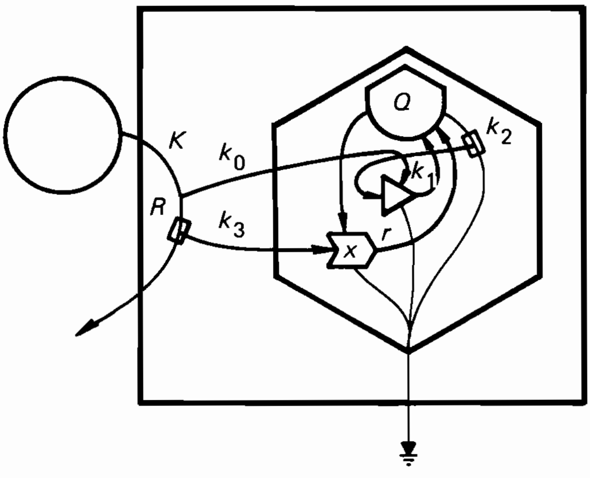


Figure 13.9. Marchetti fractional logistic for growth of consumption systems. (*a*) Energy diagram with renewable and nonrenewable sources which are assumed at constant pressure until the point where curve breaks away from logistic; (*b*) fit of resource consumption or consumer sector for early growth only.

use subsystem is found to be straight. It is not very clear by what reasoning the growth of the economy or subsystems of the economy are logistic except that increasing assets have increasing losses per unit. It is not likely that curve fitting can distinguish between the logistic and the source limited model in *Figure 13.4*. Finite resource and sector developments only follow the logistic for a limited time before breaking away.

Kriegl *et al.* (1983) develop a "hyperlogistic" overview model of nations in which hierarchical level of sectors and chain length are recognized by the power (mathematical sense) given to interacting members of the chain as they contribute to positive production functions and to limiting negative terms. Linear depreciation (i.e., mortality) terms are included. They call these rate equations a "differential tower", since the contribution to the whole is a cascade of terms that contribute exponentially to growth. Resources are unlimited since external effects on the production terms are constant. Limitations develop because of crowding effects as in the simple logistic. A simulation of future world energy use was made after calibration to past world energy use developed a leveling of growth late in the next century. Because an unrealistic, constant energy source was used, the model may develop a leveling to growth later than the real world. Although not enough detail was given to diagram the system, the presence of positive and negative terms involving multiple products of the various sectors suggests a network like that in *Figure 13.8(b)*.



$$R = K - k_0 Q$$
$$\dot{Q} = k_1 Q - k_2 Q + rRQ$$
$$k_1 = k_2$$
$$\dot{Q} = rRQ$$
$$\dot{Q} = rQ(K - k_0 Q)$$

Figure 13.10. A flow-limited logistic-consumer model with priority use of energy for maintenance.

Renewable resource logistic – it is possible to reconcile the logistic with a flow-limited renewable source with flow K as in *Figure*

13.10 which is yet another premise for deriving a logistic behavior. The diagram has first priority energy used to exactly replace linear depreciation. The remainder *R* is available to drive net growth as second priority. *R* declines linearly as *Q* increases. The priority feature is conceptually similar to one used by Wiegert (1974) for ecological systems.

13.2.4. Other overview minimodels

The logistic model has one state variable, represented in diagrams as one storage tank. Many other models with one or two tanks of total assets and population have been offered to gain overview perspectives. Several are given in *Figure 13.11*.

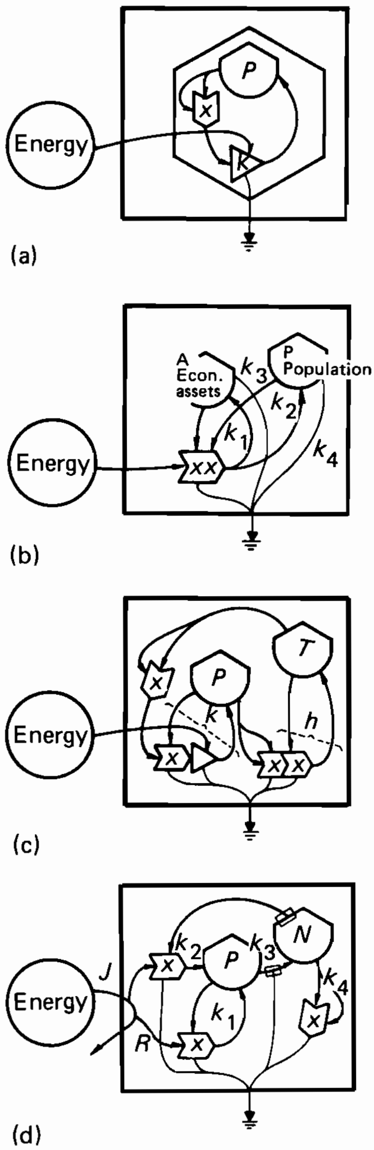
The most common perspective on exponential growth is as constant per capita growth rate of people with an implicit assumption that the assets of the economic or ecologic realm are supplied as needed in proportion. In the real world, growth of population (*P*) is dependent on assets just as the asset growth is amplified by population growth as given in *Figure 13.11*(*b*).

In frenzied growth the fastest of the growing nations actually grew faster than the exponential rate. Von Foerster *et al.* (1960) found the growth of the USA to be quadratic and thus according to the model in *Figure 13.11*(*a*). Quadratic autocatalytic growth suggests positive interactions of the stored units facilitating the growth and incorporation of required resources. Many aspects of modern economies have interactive features such as various interactions of business, stock markets, loan institutions, etc.

Taagepera (1976), after discussing Von Foerster's analysis of quadratic growth, develops a minimodel with a technology variable separate from population as growth indicator. Technology was not evaluated but postulated to be quadratic because of the self-stimulating action of inventors, etc. [*see Figure 13.11*(c)].

The author's overview minimodel in *Figure 13.11*(*d*) includes information and diversity (*N*) as generated from main assets (*P*) and feeding back to further augment the productivity.

Models that have several parallel autocatalytic production pathways each with a different mathematics have the self-organizing ability to use the combination that generates the most rapid growth. Such models may represent the process by which complex systems maximize power. A model given earlier (Odum, 1972) with linear, autocatalytic, and quadratic autocatalytic pathways uses the three sequentially during growth. At high levels of energy flow the pathways with higher mathematical power (higher exponent) prevails. The model in *Figure 13.12* is a variation that has renewable and non-renewable sources and



P = population

$$\dot{P} = k P^2$$

$$\dot{A} = k_1 EAP - k_3 A$$

$$\dot{P} = k_2 EAP - k_4 P$$

T = technology

$$\dot{P} = k T^n P$$

$$\dot{T} = h P^2 T$$

R = residual Energy Availability

N = number of cases

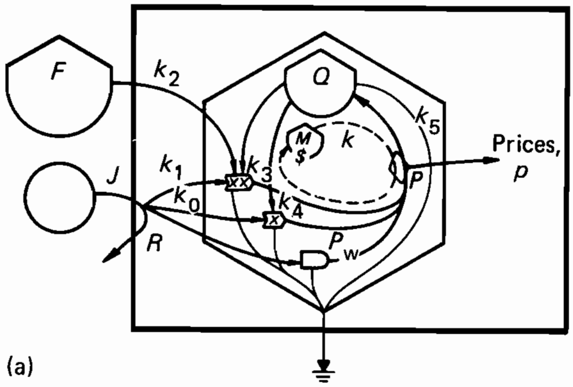
$$R = \frac{J}{1 + k_0 N + k_1 P}$$

$$\dot{P} = k_1 RP + k_2 RN$$

$$\dot{N} = k_3 P - k_4 N^2$$

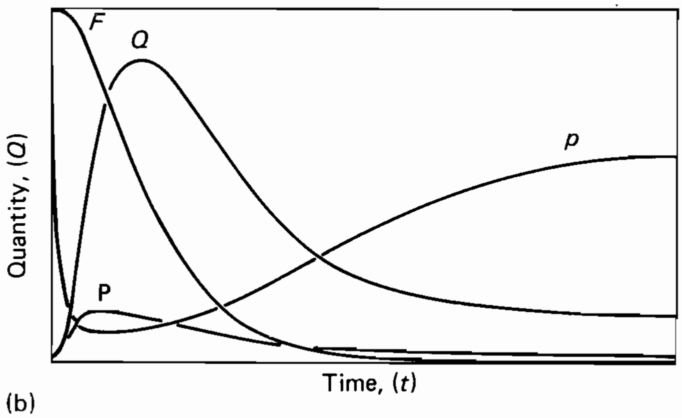
Figure 13.11. Quadratic autocatalytic growth. (*a*) Model from Von Foerster *et al.* (1960); (*b*) population-amplified economic growth tantamount to quadratic growth because A and P are similar; (*c*) revised model Taagepera (1976); (*d*) energy limit and quadratic cost of diversity (Odum and Petersen, 1972).

has the same shape growth curves as *Figure 13.6* because of similar kind of resource availabilities.



$$F = F - k_2 R F Q$$
$$P = k_3 R F Q + k_4 R Q + P_w$$
$$\dot{Q} = P - k_5 Q$$
$$p = \frac{KM}{P} \qquad KM = 100$$
$$R = \frac{J}{(1 + k_1 F Q + k_0 Q)}$$

(a)



(b)

Figure 13.12. **Growth with linear, autocatalytic, and interactive production from renewable and nonrenewable sources. (*a*) Diagram; (*b*) simulation. *See* Appendix C.**

Whereas some of the previously considered models dealt with economic assets, evaluated with dollar equivalents, money was not explicitly represented as a separately circulating quantity. In *Figure 13.12* money is circulating as a closed loop investment, counter current to the production. In that model the ratio of money flow to real asset production (*PR* = price) is calculated, but the money has no action on the growth model. A closer analysis of the model shows that with a constant amount of circulating money, the currency reaches its greatest value (highest energy to dollar ratio) when the combination of nonrenewable resources and stored assets generate maximum production rate (*P*). This occurs well before assets (*Q*) crest.

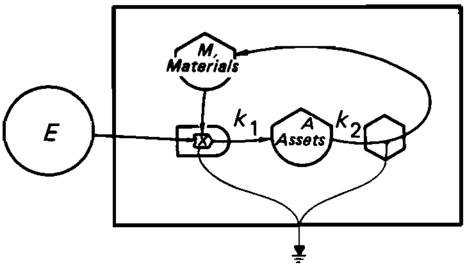
Subsequently, as nonrenewable resources fall and assets fall somewhat later, inflation results. In other simulation runs money is added with time as observed. If money is adjusted to maintain constant energy/$ ratio, inflation is eliminated. If this ratio is held constant beyond the growth period, money supply decreases with decreasing energy supply. If borrowing psychology prevails beyond the period of growth into the time of decreasing money supply, interest rates increase, helping to transform the borrowing psychology.

13.2.5. Production–consumption models

In both ecology and economics there are many models that combine production and consumption and circulation of materials or money or both between them. As a summary of many in the literature of both fields the simplest of this class of models are given in *Figures 13.13–13.15*. One or two energy sources drives producers shown on the left coupled to consumers on the right. Commodities are generated by producers flowing to the right to consumers, and services are generated and fed back to the left by consumers.

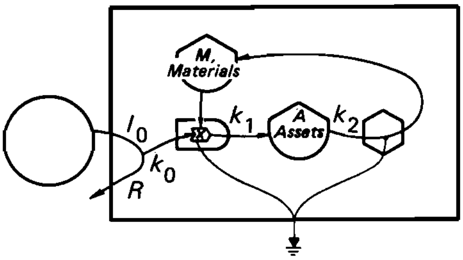
In *Figure 13.15* the circulation of money is added to the models in *Figure 13.14*, so they become typical of the models of money given in elementary economics texts. Odum and Bayley (1974) simulated a model of production and consumption in *Figure 13.15*(*a*) with money explicitly separated, but coupled by making price inverse to supply. Since the energy source (E) was one of unlimited availability, exponential growth resulted. In *Figures 13.15*(*b*) and *13.15*(*c*) there are resource limitations in the rate of supply of renewable resource and in *Figure 13.15*(*d*) in the storage of nonrenewable resource available. The resulting curves have some properties of the single storage models, growth leveling as the rate of use approaches the rate of supply of the source.

A more complex producer–consumer model includes land rotation. In one sense the human pattern may be overviewed as one giant land rotation (*see Figure 13.16*). The natural productive processes generate storages such as soils, forests, fisheries, and minerals. Then human cultures develop which use these stores as resource, developing the land for agriculture, and human settlements. In the process the land is cultivated for these new uses until its stores have been depleted. Then with less resources the surge of human development decreases, and the land is reclaimed by the natural environmental systems which start rebuilding the storages again. Because there is a limited quantity of land area that is recycled in the model, the example system is ultimately stable in its alternation between a time of productive rebuilding to a period of sharply developing human assets.



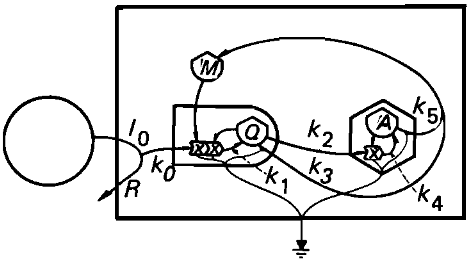
$$M + kA = K$$
$$\dot{A} = k_1 EM - k_2 A$$

(a)



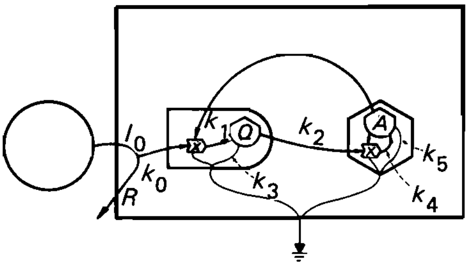
$$M + kA = K$$
$$\dot{A} = k_1 RM - k_2 A$$
$$R = \frac{I_0}{1 + k_0 M}$$

(b)



$$M + k_6 Q + k_7 A = K$$
$$\dot{A} = k_4 AQ - k_5 A$$
$$\dot{Q} = k_1 RMQ - k_2 AQ - k_3 Q$$
$$R = \frac{I_0}{1 + k_0 M}$$

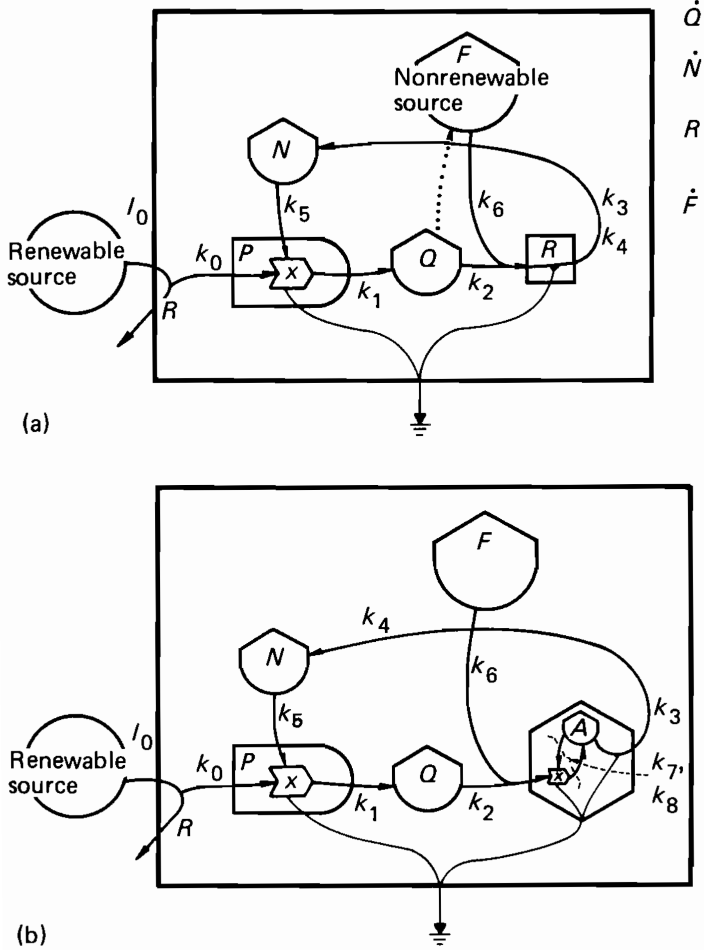
(c)



$$\dot{A} = k_4 AQ - k_5 A$$
$$\dot{Q} = k_1 RA - k_2 QA - k_3 Q$$
$$R = \frac{I_0}{1 + k_0 M}$$

(d)

Figure 13.13. Production–consumption–recycle models. (*a*) Limited by material recycle – Michaelis–Menten system; (*b*) same with renewable, flow-limited source; (*c*) with autocatalytic producer and consumer; (*d*) service feedbacks from consumer without explicit materials cycle.



(a)

$$\dot{Q} = k_1 NR - k_2 Q$$
$$\dot{N} = k_3 Q + k_4 F - k_5 RN$$
$$R = \frac{I_0}{1 + k_0 N}$$
$$\dot{F} = -k_6 F$$

(b)

$$\dot{Q} = k_1 RN - k_2 AQ$$
$$\dot{N} = k_4 A - k_5 RN$$
$$R = \frac{I_0}{1 + k_0 N}$$
$$\dot{F} = k_6 FA$$
$$\dot{A} = k_7 AQ + k_8 AF - k_3 A$$

Figure 13.14. Production–consumption–recycle models with nonrenewable source. (*a*) With fuel driven linear consumption; (*b*) with autocatalytic growth of consumer assets.

13.2.6. Global geochemical models

The simple production–consumption models in *Figure 13.13*(*a*)–(*c*) include the main metabolism of the biosphere in the long run, considering the biosphere as if it were a closed aquarium (Odum and Lugo 1970) and may be appropriate as the most simple carbon dioxide model. In *Figure 13.13*(*a*) because of the limited quantity of recycling materials (e.g., carbon) the production–consumption model is an internally stabilized Michaelis–Menten model. In *Figure 13.13*(*b*) and (*c*), the model is further limited externally by its outside renewable resource flow. In

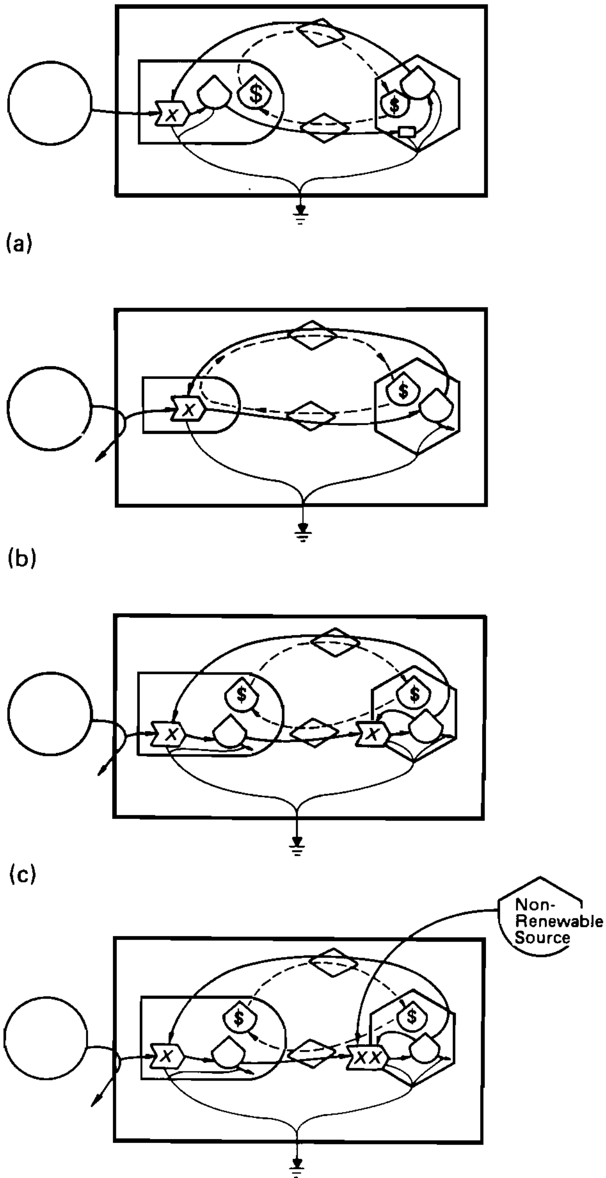
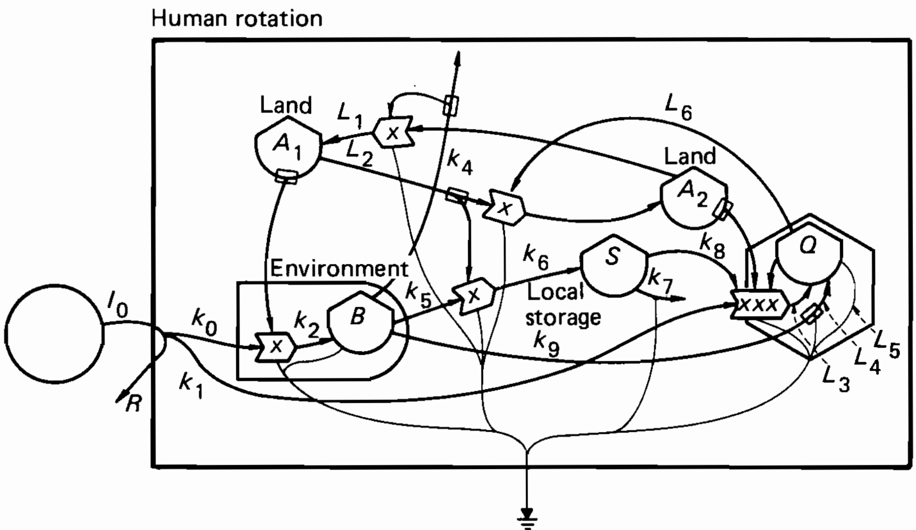


Figure 13.15. Production–consumption models with explicit money circulation. (*a*) Unlimited source, constant money supply (Odum and Bailey, 1976); (*b*) simple version with one source and storage; (*c*) with producer assets and consumer assets separated; (*d*) same with nonrenewable source added.



$$R = \frac{I_0}{(1 + k_0RA + k_1SQA_2)}$$

$$\dot{B} = k_2RA - k_3B - k_4B - k_5BA_1 - k_9B$$

$$\dot{S} = k_6BA_1 - k_7S - k_8SRA_2Q$$

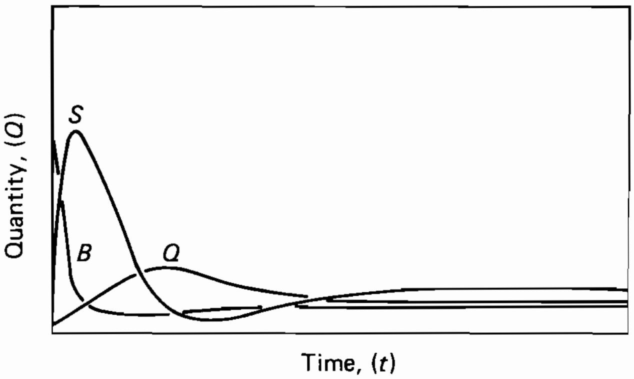
$$\dot{A} = L_1A_2B - L_2QA_1$$

$$A_2 = A - A_1$$

$$\dot{Q} = L_2QA_2SR + L_4S$$

$$= L_5Q - L_5AQ + L_3QA_2SR$$

(a)



(b)

Figure 13.16. Production–consumption model of humans and land rotation and restoration. (*a*) Diagram; (*b*) simulation.

Figure 13.13(d), the feedback of consumer service becomes a direct production factor (instead of acting through a storage). This model will not grow unless an additional inflow to *A* is supplied [not shown in *Figure 13.13*(d)].

Adding a nonrenewable source to the consumer side of the system makes the model appropriate for the biosphere during the current century [*Figure 13.14*(*b*)]. *Figure 13.14*(*a*) has a linear consumption function whereas *Figure 13.14*(*b*) has an autocatalytic consumer unit which can develop a surge of consumption. The land rotation model in *Figure 13.16* may represent the essence of humanity and nature in an agrarian economy.

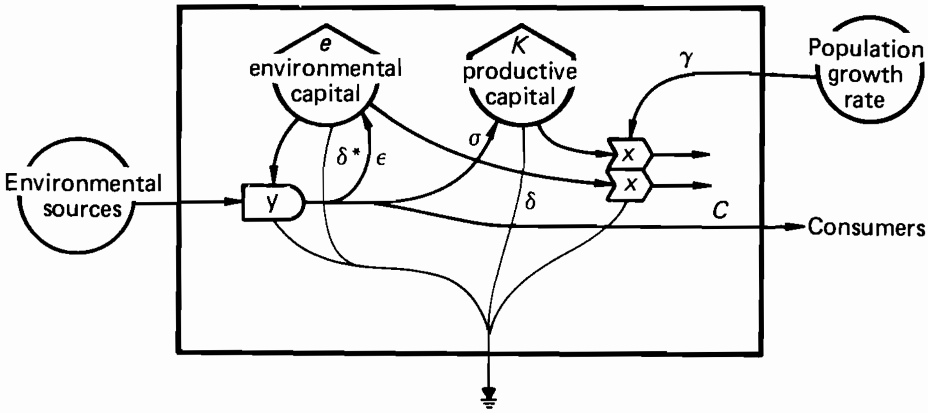
The many *geochemical models* of cycles of carbon, phosphorus, sulfur, nitrogen, etc., are mainly of larger time and space scale than the economic–environment systems under review here. The same is true of climatic–oceanic models that deal with ice ages, etc., such as that by Sergin (1980). A review of a few of these was given in Odum (1983b, Chapter 26). It was found that when a single material cycle such as carbon is isolated it is almost by definition not an adequate simulation model because the main factors that drive the cycle are coupled inputs from other parts of the global system, decoupled when one material is isolated.

13.2.7. Models of economic use interface with environment

Nijkamp (1977) provided equations for an interface model with environmental capital and productive capital. This is diagrammed in energy language in *Figure 13.17*.

The interface between environmental systems without money and economic use which involves human service and the circulation of money in its most simple form is given in *Figure 13.18*. The environmental externality generates flows and storages on the left which are incorporated in the economic loop shown on the right. Examples are forest plantations, fisheries, and agriculture.

Money is paid only for human service involved and not for the contribution of the externality. The contribution to the economy of the external resource may be tallied by the computer calculation of embodied energy contributed as done in Section 13.3. The circulation of money is explicitly given separately. For a microeconomic situation the ratio of money to energy flow are externally determined prices. In the example given in *Figure 13.18* money gained from sales all goes for inflow of goods and services that ultimately are derived from a different source of energy from the right side of the diagram. The price of the purchased goods and services relative to the price of the products sold is a measure of the general energy level supporting the main



$$\dot{K} = \sigma y - (\gamma + \delta)K$$
$$C = (1 - \sigma - \epsilon)y$$
$$\dot{e} = \epsilon y - (\gamma + \delta^*)e - h(k,C)$$

Figure 13.17. Energy diagram of environmental capital model given by Nijkamp (1977).

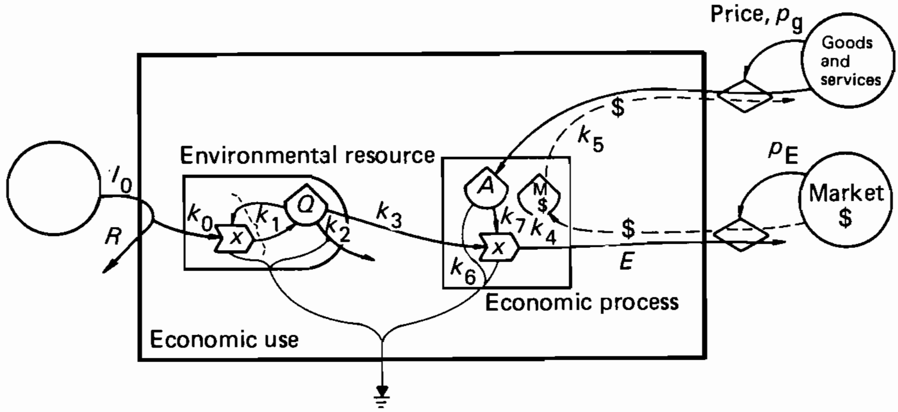
economy. When the total support energy decreases, each contributing externality becomes relatively more important to the economy.

Labor-mediated resource interface – Howe (1979) presents a simple model for resource effects on gross national product (GNP) facilitated by the assignment of labor. As diagrammed in *Figure 13.19*, the GNP is a product (through Cobb–Douglas production function) of the resource commodities (*RO*) mediated by labor (*Lt*) used in resource processing, and the labor remaining for more direct work in the economy (*LO*). The model was used to determine graphs of optimal assignment of scarce labor for maximizing GNP.

In his first "frontier" case, the resource processed per unit labor used (*g*) is constant – a constant availability case which generates unlimited growth if the supply of labor grows.

In a second case (*g*) is increased with time representing a rising efficiency of resource conversion attributed to new technology. This may not be realistic since a good part of new technology requires additional resource use for its support. To add an increase in (*g*) due to technology, the dashed feedback line "Technology" would be added from GNP to augment (*g*) in *Figure 13.19(a)*.

In a third case designated "Ricardian" there is a decline in resource contribution due to declining stocks of resources; *Figure 13.19(b)* shows an example of nonrenewable resource-based economy, this resource interface operating alone would generate a cresting and declining GNP.



$$R = I_0 - k_0 RQ$$

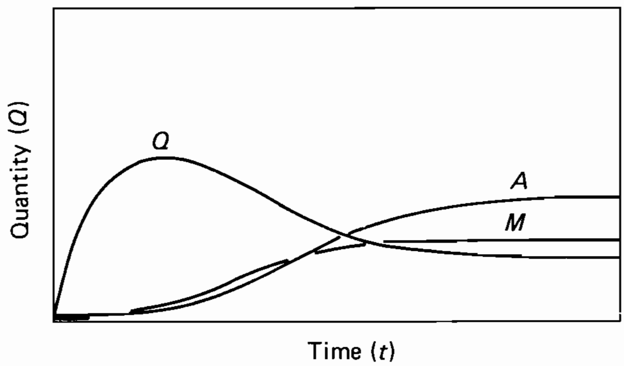
$$R = \frac{I_0}{1 + k_0 Q}$$

$$\dot{Q} = k_1 RQ - k_2 Q - k_3 AQ$$

$$\dot{M} = p_E\, k_4\, AQ - k_5 M$$

$$\dot{A} = \frac{k_5 M}{P_g} \quad k_6 A - k_7 AQ$$

(a)



(b)

Figure 13.18. A basic model of environmental interface. (*a*) Energy diagram; (*b*) simulation starting with small storages.

13.2.8. Macroeconomic models

Many of the main concepts of *macroeconomics* have been traditionally expressed in simple overview models of the circulation of the economy. (Summaries given by Beach, 1962; Mueller, 1966; Allen, 1968; Hamberg, 1971.) The flows of money circulating with closed circuits through production processes and consumers are expressed in simple equations

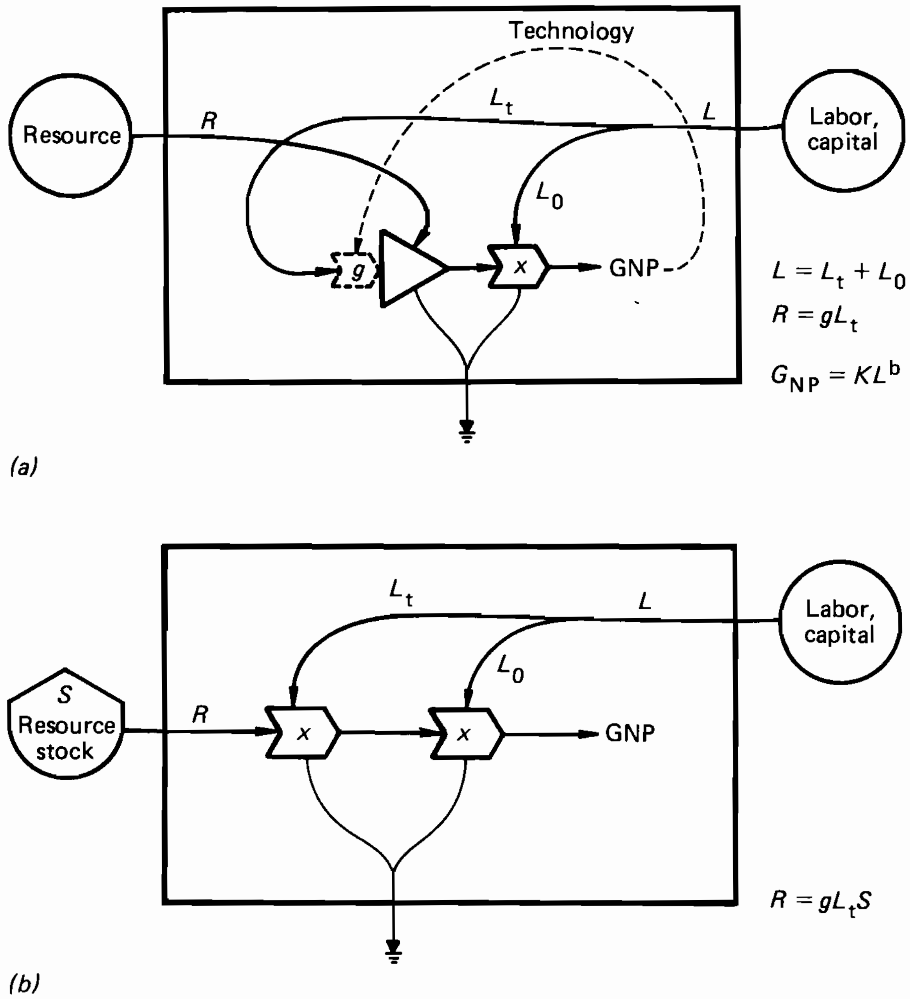
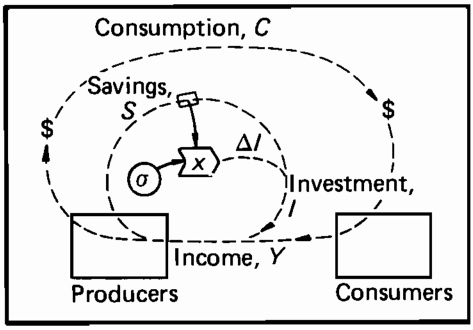


Figure 13.19. Labor-mediated resource base model (Howe, 1979). (*a*) Resource with constant (frontier) resource availability; technology effect with time increasing resource processing efficiency *g*; (*b*) Ricardian economy with declining stock (*S*) causing increasing processing costs (lower GNP).

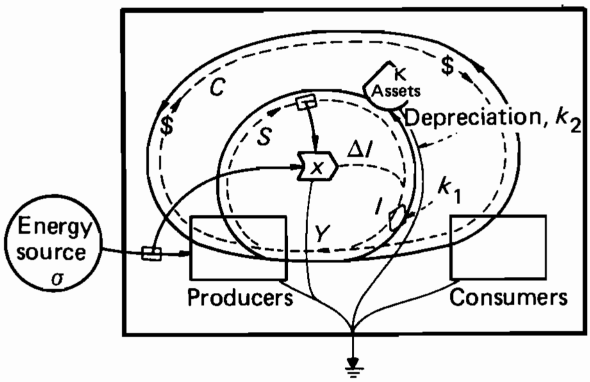
relating flow in one segment of the circulation in terms of contributing flows other segments of the circulation.

Domar (1937) and Harrod (1939) developed models of the growth of economic circulation which also generate exponential expansion. The model is given in *Figure 13.20* in equation and diagram form. Increments of new investment ΔI were generated with corresponding equal percent increases in the connecting flows of savings and consumption.



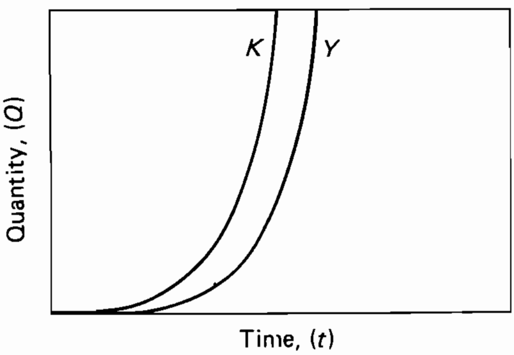
$$S = aY$$
$$Y = S + C$$
$$S = I = aY$$
$$\dot{I} = \frac{\Delta I}{\Delta t} = a\sigma I$$
$$\frac{\dot{I}}{I} = \frac{\dot{Y}}{Y}$$

(a)



$$\dot{K} = k_1 I - k_2 K$$

(b)



(c)

Figure 13.20. Domar–Harrod model of exponential growth of the economy with constant percent increase in investment. (*a*) Equations and diagram; (*b*) with assets and energy relationships added; production function is additive; (*c*) simulation.

No increase in capital storage was included in the model. The dashed lines represent the main money flows of the economy as viewed macroeconomically by Domar and Harrod.

The percent increase with time was recognized as a function of some accelerator action: (σ) in *Figure 13.20*(a). The relation of money in dashed lines as a counter current to the energy-carrying commodities and services to which they correspond is shown in *Figure 13.20*(b). The amplification which is possible is dependent on the available energy source. Macroeconomic variables, income (Y), savings(S), investment (I), and consumption (C) are shown.

Often macroeconomic models can be espressed as steady-state equilibrium models with the equations indicating the equalities among constant flows. Changes with time were dealt with by rewriting a new equation for the next year with some incremental increase in all the magnitudes over the previous year. If the circulation has branches, the web of closed-loop circulation of the economy can be represented as an input–output table (input–output models).

Such static equations are readily converted into dynamic equations by including storage quantities [tank symbol K in *Figure 13.20*(b)] where appropriate and describing the system with differential and difference equations.

In *Figure 13.20*(b) assets are added in proportion to money invested minus depreciation. Dynamic equations can then be computer simulated to represent the patterns with time [*Figure 13.20*(c)]. However, when dynamic models are generated for the converging and diverging of flows through production and consumption processes, questions arise as to the correct production function. In *Figure 13.20* production is represented as simple addition. But at most intersections the action of one flow controls and amplifies the other, and the relationships may be product type functions as in *Figure 13.21*.

The classical macroeconomic models did not have externally limiting inflows [*Figure 13.20*(a)]. Impetus and control of increments of growth were derived from properties within the circulation as in *Figure 13.20*(a). Whenever an increment of growth is in proportion to existing dimensions, the growth is exponential. Simulation of models that are all exponential is not very interesting and the emphasis naturally concerns the amplifier gain coefficients and what macroeconomic properties of the economic system were determining them.

Such models without external flow limits were appropriate for the period of rapid economic growth in which total external resource availability was not limiting. When the resources are large enough relative to use to cause exponential growth is when they are having their greatest stimulus to the economy. Their energy availability is constant even though demand for use is increasing. In other words, the classical macroeconomic models had constant parameters which were really

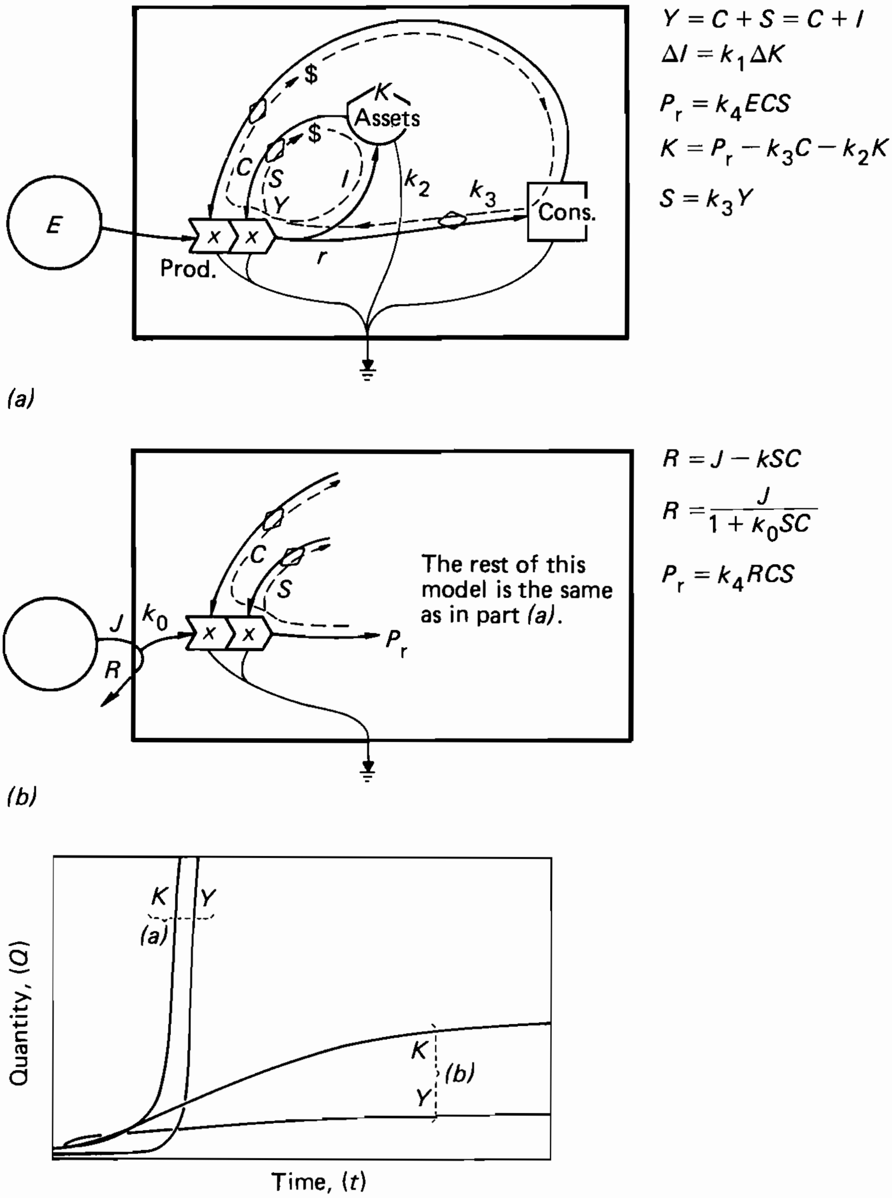


Figure 13.21. Keynesian macroeconomics model with energy sources, coupled money flows, capital assets, and multiplier production functions. (*a*) With unlimited resource availability (E = constant); (*b*) with source-limited resource availability; (*c*) simulations of models in (*a*) and (*b*).

unrecognized energy sources. Including a constant is the mathematical equivalent of adding unlimited resources [*Figure 13.2(a)*]. For example, in diagramming the Harrod–Domar model in *Figure 13.20* the implied constant source of energy availability can be identified as σ.

Adding external resource limitations [*Figure 13.2(b)* or *13.2(c)* changes macroeconomic models from exponential growth to "S" shaped growth forms. Modification of a traditional Keynesian overview macroeconomic minimodel to include an externally limited source is included in *Figure 13.21(b)*. Compare the simulations with constant source E and flow limited availability R in *Figure 13.21(c)*.

13.2.9. Input–output models

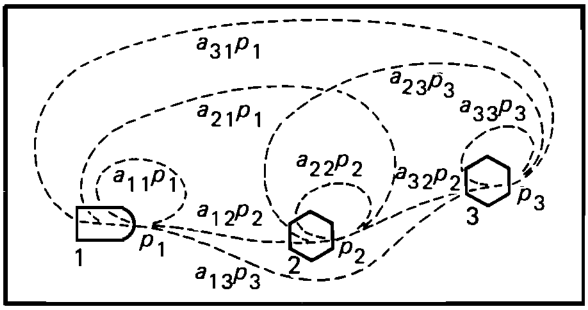
Input–output models describe the flows between the sectors of a system in terms of constants that relate the fraction of the productive output of a sector due to an input. Most input–output models are complex (with many sectors) and thus not pertinent to this review. However, very simple input–output models are sometimes used for overviews. For example, a three-sector model was given for New Zealand by Wallace *et al.* (1980). *Figure 13.22(a)* shows a three-sector input–output model in equation form, in matrix form, and with a pathway diagram for the flows. As applied to an ecosystem (Hannon, 1976), the flows may represent materials.

In an economic system where traditionally used, the lines represent money flow in one direction while also simultaneously representing the flows of goods and services over the same pathways in the opposite direction. In *Figure 13.22* the as are the input–output coefficients and the Ps are the productive output of each sector (sum of money inflow to that sector).

Like the static macroeconomic models there are no storages and the relationships of flows (input–output coefficient, as) are constant. Growth without change is sometimes simulated by increasing the flows while holding the proportions among flows constant. This may not be realistic simulation, since most of the intersections (1, 2, and 3 in *Figure 13.22*) have non-linear production or consumption functions and do not keep constant relationships with growth.

As illustrated in *Figure 13.22(a)*, the additive flows of commodities labor, services, capital, etc. converging are described by Leontief input–output coefficients which are a kind of linear production function (Diwert, 1971; Hanoch, 1971).

In *Figure 13.22(b)* storages and production functions are included to transform the model in *Figure 13.22* to a dynamic one. If no special external resource limit function is supplied this model implies constant resource availability (S = constant) and exponential growth. If S is externally limited, a steady state can result.



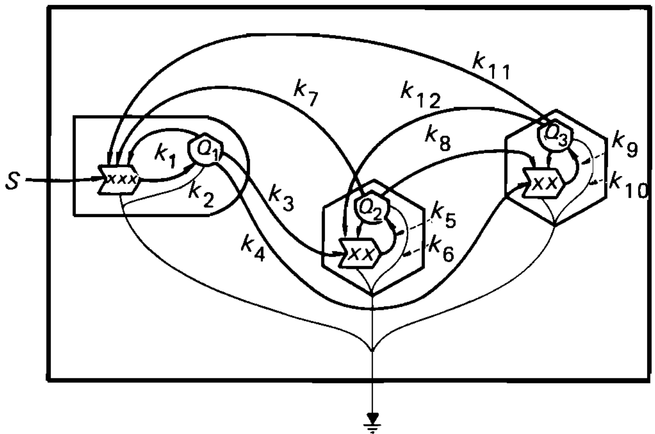
$$p_1 = a_{11}p_1 + a_{21}p_1 + a_{31}p_1$$

$$p_2 = a_{12}p_2 + a_{22}p_2 + a_{32}p_2$$

$$p_3 = a_{13}p_3 + a_{23}p_3 + a_{33}p_3$$

	1	2	3
1	$a_{11}p_1$	$a_{21}p_1$	$a_{31}p_1$
2	$a_{12}p_2$	$a_{22}p_2$	$a_{32}p_2$
3	$a_{13}p_3$	$a_{23}p_3$	$a_{33}p_3$

(a)



$$\dot{Q}_1 = k_1 S\, Q_1 Q_2 Q_3 - k_2 Q_1 - k_3 Q_1 Q_2 Q_3 - k_4 Q_1 Q_2 Q_3$$

$$\dot{Q}_2 = k_5 Q_1 Q_2 Q_3 - k_6 Q_2 - k_7 S\, Q_1 Q_2 Q_3 - k_8 Q_1 Q_2 Q_3$$

$$\dot{Q}_3 = k_9 Q_1 Q_2 Q_3 - k_{10} Q_3 - k_{11} S\, Q_1 Q_2 Q_3 - k_{12} Q_1 Q_2 Q_3$$

(b)

Figure 13.22. Input–output model. (*a*) Static, without storages; economic structure indicated by constant coefficients (*a*s). Money flows in direction of arrows representing commodities and services flowing on the same pathways in opposite direction; (*b*) same model made dynamic by adding storages, production and consumption functions, and depreciation.

The dynamic version in *Figure 13.22(b)* mathematically approaches the static version in *Figure 13.22(a)* at steady state. Thus, input–output models may be used to study steady state flow relationships but not the nature of growth, since sector relationships are usually changed by the nonlinear interplay of commodities, labor, capital, controls, etc.

As in the example in *Figure 13.22(a)*, input–output models can be derived from a dynamic model as one technique for study of the flows. The reverse is not true. The dynamic model for study of change with time is not determinable from the input–output model. There is more than one possible dynamic model for the same input–output data.

13.2.10 Pulsing models

Models with two or more storages are capable of oscillations, alternating production with consumption. In this respect these models may be more realistic than the single storage models discussed first. In economics the alternation of a period of storing and a pulse of rapid consumption is modeled as the stock inventory cycle in the short run (3 years) and a longer Kondratief cycle (60 years). These represent a cycle of net construction and net using up of capital infrastructure (bridges, roads, railroads, etc.). Similar alternation of production and pulsed consumption appears to be very general in ecosystems.

Figures 13.23 and *13.24* show the production–consumption pulsing in two ways. *Figure 13.23* has a logic threshold which turns on a surge of consumption which recycles materials to generate a long period of production. In *Figure 13.24* multiple pathway inputs to storage are included in the consumer storage which gives the producer–consumer model the special property of self pulsing (Alexander, 1978), alternating periods of net production with periods of frenzied consumption. When this model is calibrated with the production of resources by world geological processes and use of resources by twentieth century civilization to develop current world assets, the model generates a very sharp blip as seen over geological time (*Figure 13.24(c)*, resembling King Hubbert's blip (Hubbert, 1973). A fossil fuel source is nonrenewable when considered in a short time frame but is really a renewable one on a pulsed production and later consumption.

When the model in *Figure 13.24* is calibrated against soil and wood biomass storages, it generates pulses of shorter term, possibly appropriate for earlier human civilizations such as Mayan culture which was mainly supported by photosynthetic products. If these models are thought to represent the biosphere, the period following the consumption surge is not a steady state but a period of gradual net production rebuilding products capable of supporting later frenzied surges of consumption.

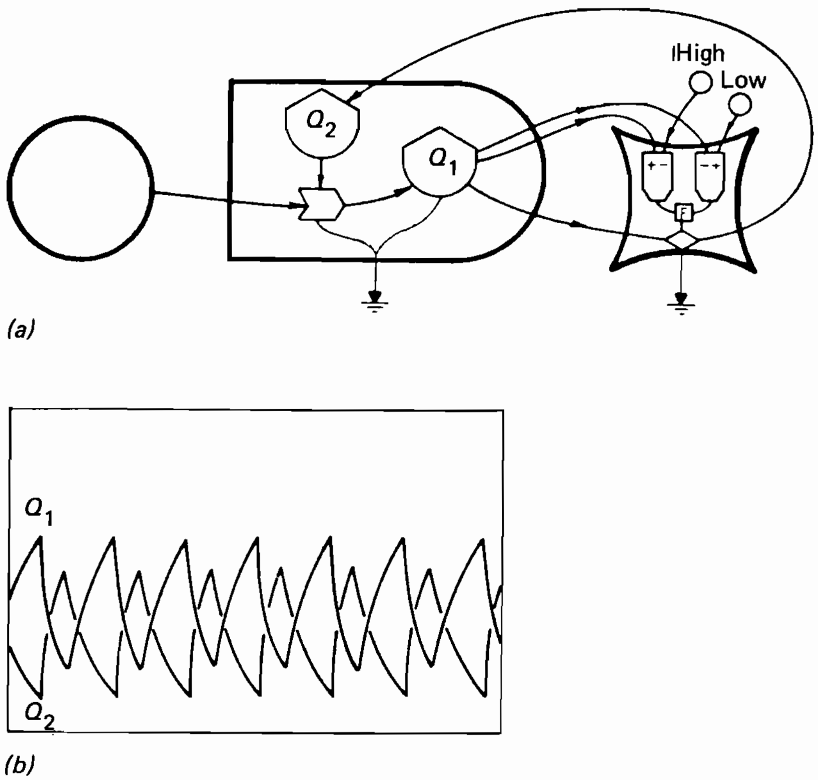


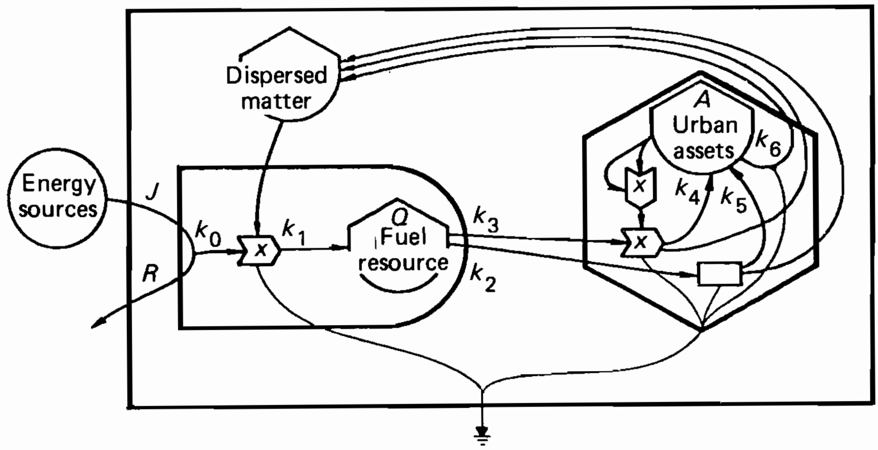
Figure 13.23. Self-pulsing models of production and consumption. (*a*) Threshold pulsing of rapid consumer by logic; (*b*) simulation of (*a*). *See* program in Appendix A (Odum, 1983*a*,b; p. 76: Figs 6–8).

13.3. New Modeling Approaches

The energy systems diagramming of overview models suggests new avenues of the realms of ecology and economics.

13.3.1. Simulating emergy and transformity

Since simulation models generate products by representing the interaction of contributing factors, commodities, capital, services, etc., these same models may generate emergy and transformity values as an additional tally of the iterations. Two examples of this are given in *Figure 13.25*. Q is the energy storage. The extra tank (E) is the emergy storage. Where (E / Q) is calculated, the result is the transformity. In *Figure 13.25(a)* this is multiplied by the energy flow to obtain the emergy flow. In the second example [*Figure 13.25*(*b*)], the process has



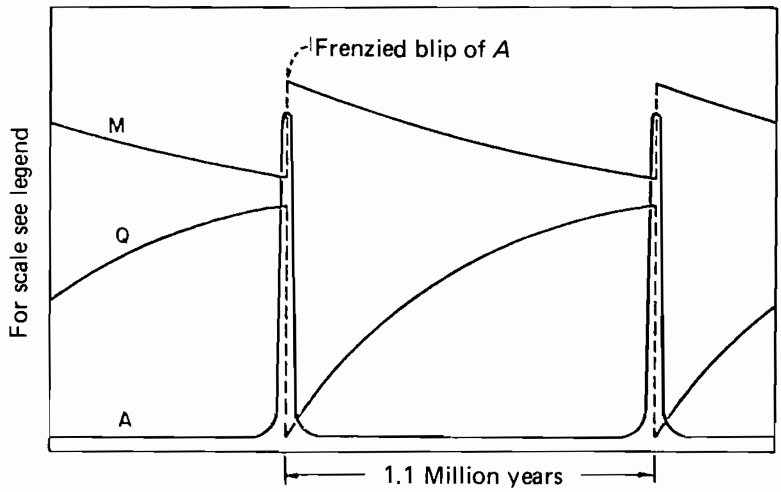
$$R = \frac{J}{1 + k_0 M}$$

$$\dot{Q} = k_1 RM - k_2 Q - k_3 QA^2$$

$$M = K - k_7 Q$$

$$\dot{A} = k_4 QA^2 + k_5 Q - k_6 A$$

(a)



(b)

Figure 13.24. Production–consumption minimodel with quadratic pulsing mechanisms. (*a*) Simulation model; (*b*) simulation results.

inflows of other input sources (Y and Z). Here the transformity is supplied from outside as a property of the source.

As shown by the simulations in *Figure 13.25*, transformities increase during growth of assets (Q). Larger storages require more emergy for their maintenance and are of higher quality and have larger feedback amplifier abilities.

13.3.2. Macroscopic minimodels of nations

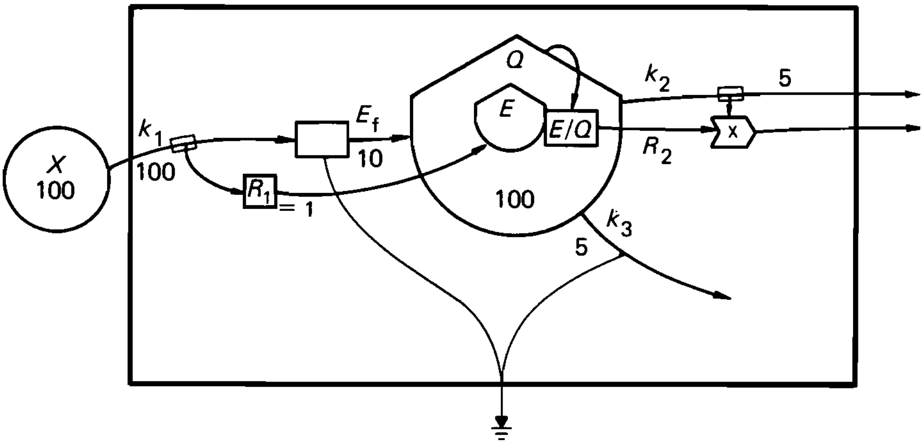
Macroscopic minimodels of nations need to represent a nation's assets as based on its renewable sources, its nonrenewable sources, its import and export exchange with areas supplying raw products more rural in the hierarchy and import–export exchange with developed centers. A generic model that has these influences is given as *Figure 13.26*. Approximate numbers for Brazil were used for the preliminary simulation in *Figure 13.26*. Another example was given for Switzerland (Pillet and Odum, 1984) and New Zealand (Odum, 1984).

13.3.3. Simulating foreign trade using emergy

Variation in exchange prices of foreign trade is a main external source factor controlling the assets that a nation may develop and maintain. This causal property is included in many concepts, models, and recommendations for national policy. Sometimes minimodels are developed with this as the main variable. For example, in *Figure 13.27* is diagrammed a model for national assets given by Samouilidis (1981) after Sweeney (1978, 1979). Increase in energy price decreases the outside energy inflow but also diminishes the energy use per unit production according to price elasticity. Whereas this model was generated for fuels, it applies to all commodities in trade. It is a static equilibrium model since it lacks any storages and assumes constant instantaneous values for constant external conditions.

In foreign trade there are imports and exports of commodities and for each money is paid so there is a circulation of money. A model of trade between two nations is given in *Figure 13.28*. Sales may be based on world prices and the flow of money in one direction roughly balances the flow of money in the reverse direction. When there are imbalances in these money payments, currencies accumulate locally, affecting their value in converting of one currency to another so that circulation of money is facilitated. The prices involved represent the momentary scarcity value to buyers and sellers.

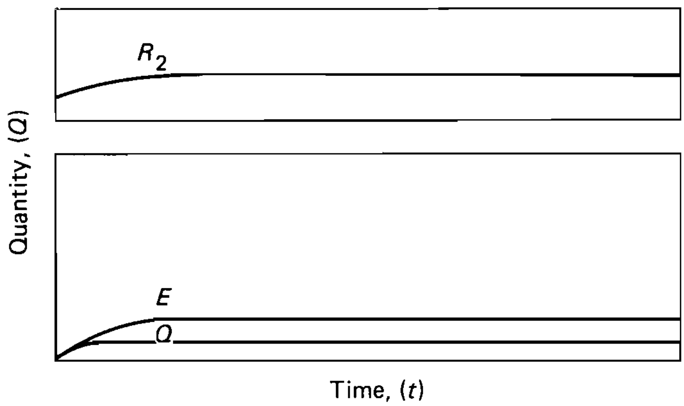
However, sale prices are not an indication of the contribution of the commodities to the economies because they only include payments



$$R_2 = E/Q$$

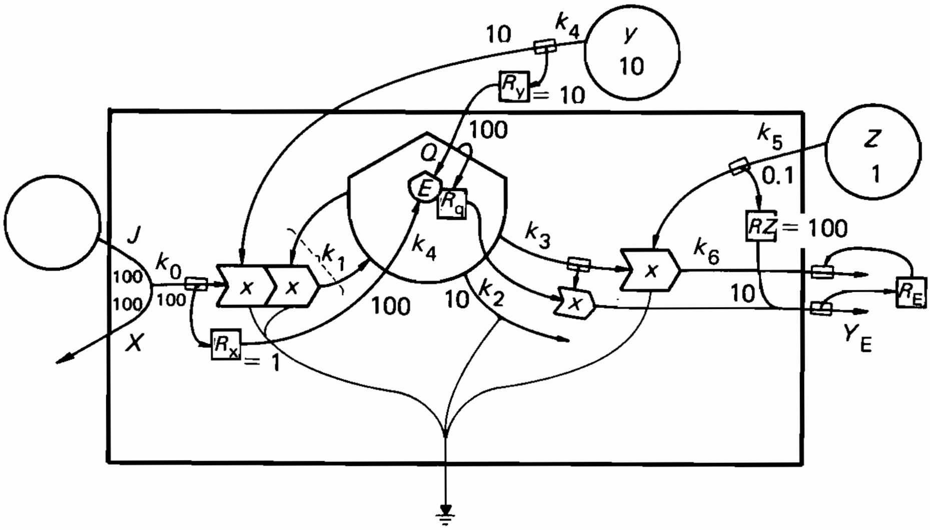
$$\dot{E} = R_1 k_1 X - R_2 k_2 Q$$

$$\dot{Q} = E_f k_1 X = k_2 Q - k_3 Q$$



(a)

Figure 13.25. Models that simulate both energy and emergy. (*a*) Simple storage.



$$X = \frac{J}{(1 + k_0 y\,Q)}$$

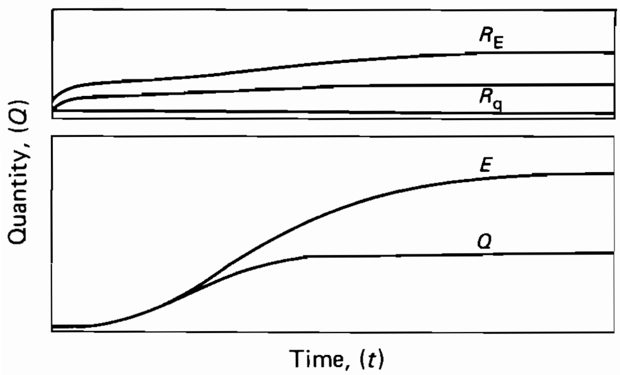
$$R_q = \frac{E}{Q}$$

$$\dot{Q} = k_1 Q\,y - k_3 Z\,Q - k_2 Q$$

$$Y_E = R_z k_5 ZQ + R_q\,k_3 ZQ$$

$$R_E = \frac{Y_E}{k_6 ZQ}$$

$$\dot{E} = R_x k_0 XQy + R_y k_4 XQy - R_q k_3 ZQ$$



(b)

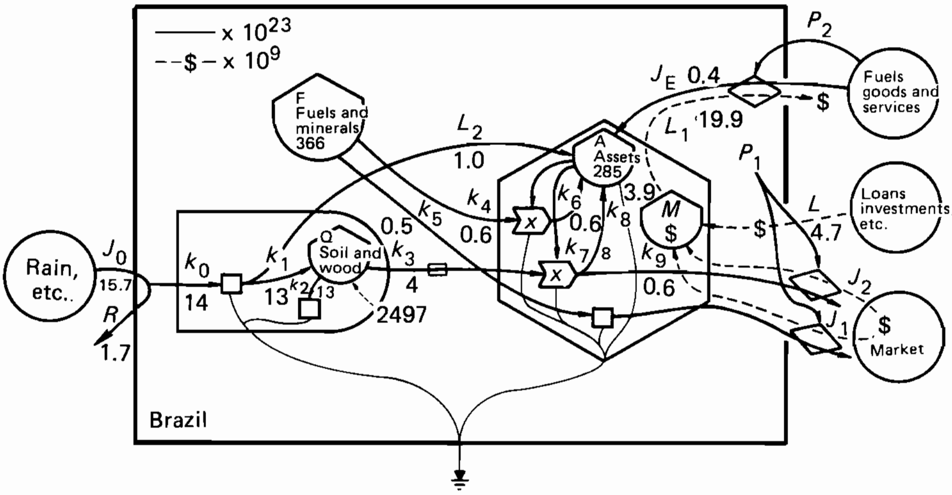
Figure 13.25. Continued. **(b) Autocatalytic unit with three sources of different quality.**

for human service. The money paid for the human service involved in economic processing of resources has no simple relationship to the embodied energy in the resource being used. In the trade between two countries there may be large differences between the embodied energy exported and that imported even though there is a balance of payments in dollars. This means that the economy with the most favorable balance of embodied energy is being stimulated more than the other. See examples for 11 nations given recently (Odum and Odum, 1983).

In the hierarchical spatial organization of the earth some areas are rural, converging raw products to towns, cities, and urban centers. The highly developed centers return finished products to the rural areas diverging their goods and services. Nations occupy various positions in the world hierarchy. Many nations are rural and process raw products from environmental resources, receiving back the finished products. Others such as Japan, Germany, and Holland are highly urban, receiving raw products and returning finished products. Still other nations are large enough and develop enough to contain rural and urban components of the hierarchy. When there is trade between a rural nation supplying environmental products with large embodied energies and an urban nation, there is a large imbalance in the relative contributions of the trade to the economies. In order for the trade to be truly mutual, the exchange would have to be calculated on an embodied energy basis rather than the money basis that only measures the human contributions. In the model in *Figure 13.28* trade is between a rural nation on the left and an urban one on the right. There is included a computer tally of the balance of embodied energy.

Energy mutualism – it may be postulated that equitable distribution of embodied energy use is a design principle for the international fabric of nations (Odum, 1984). If the embodied energy is the predictive measure of economic vitality, then this equity provides a mechanism for economic equality. Economic equity may favor international stability and peace because equity makes each area equally capable of economic and military influence with capability for defense but without enough superiority of power to control areas beyond. Accompanying the economic equity may be equity of the populations and their perceptions of economic justice.

The ultimate reason for an equitable energy distribution may be the maximum power principle. The total power of the global system may be maximized by feedback reward loop reinforcements that accelerate rate of resource use including production and consumption. (*See* uppermost feedback pathway in *Figure 13.28*). Nations with imbalance of embodied energy in foreign trade are not feeding back to sources pumping actions in proportion to energy received. Energies are going into pathways which do not maximize world power. Evolution of patterns with more energy mutualism may increase the combined power of



$$R = J_0 - k_0 R$$

$$R = \frac{J_0}{(1 + k_0)}$$

$$\dot{Q} = k_1 R - k_2 Q - k_3 QA$$

$$\dot{M} = J_1 + J_2 + L - J_E$$

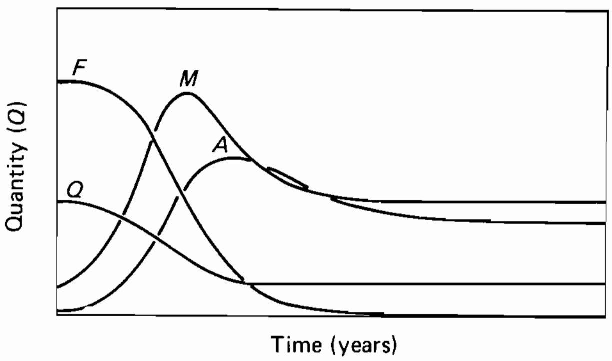
$$\dot{A} = k_6 FA + k_7 QA - k_8 A + \frac{J_E}{P_2} + L_2 R$$

$$\dot{F} = -k_4 FA - k_5 F$$

$$J_E = L_1 M$$

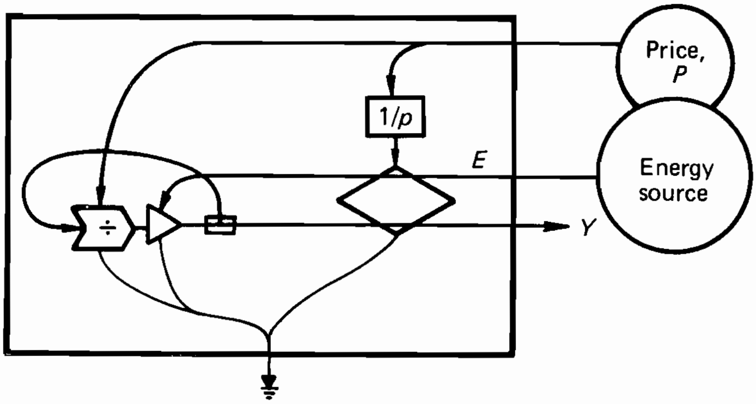
$$J_1 = p_1 k_5 F \qquad J_2 = p_1 k_9 QA$$

(a)



(b)

Figure 13.26. **Generic minimodel of a nation and its dependence on types of energy source and position in trade. (*a*) Diagram; (*b*) simulation using data for Brazil.**



$$\frac{dY}{dp} = -E$$

$$E = AY^n p^{-E}$$

Figure 13.27. Minimodel of effect of price of energy (p) on net production (Y) after Samoulidis (1981) and Sweeney (1978).

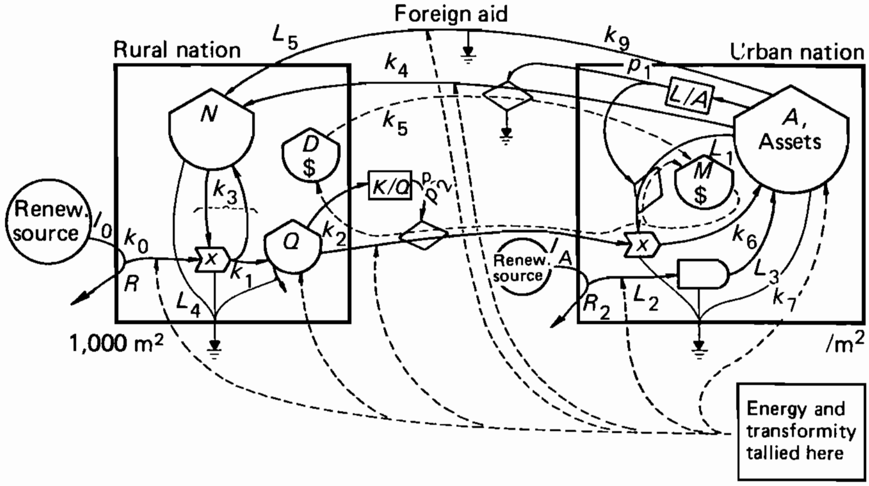
nations favoring the selection of the economic and political patterns capable of developing a better balance of energy.

Since embodied energy imbalances in trade are readily calculated, it maybe suggested that additional feedbacks from central place nations to the rural nations should be supplied linking the nations as better partners. This may take the form of culture, information, technology, military protection, etc. This principle is already recognized in the patterns of foreign aid and international subsidies to undeveloped areas, but the means for calculating what amount is appropriate is now provided with the emergy balance method. In the model in *Figure 13.28* one simulation run was made with the embodied energy imbalance in the economic trade corrected with the additional feedback from A to N shown. Total power was increased.

13.3.4. Driving regional models with macroscopic minimodels

Often modeling is done with main attention to parts and their mechanisms. In any system the largest effects come from the long period and large amplifier effects of the next larger surrounding system. Thus, any model of a nation or regional system needs a global minimodel to drive those effects of the whole on the smaller area. For example, rise and fall of world growth may cause a similar pattern on its regions by causing increase than decrease of availability of commodities for trade.

In *Figure 13.29* a world minimodel drives a minimodel of Florida. The environmental tank (S) of the world model is calibrated with



$$R = \frac{I_0}{1 + k_0 N} \qquad R_2 = \frac{I_A}{1 + L_2}$$

$$P_1 = \frac{L}{A} \qquad p_2 = \frac{K}{Q}$$

$$M = M_T - D$$

$$E_x = \text{energy exchange ratio} = \frac{T_Q k_2 A Q}{\dfrac{T_A k_5 D}{P_1} + T_A k_9 A}$$

$$F = \text{fraction feedback wthout double counting} = \frac{k_9 A + \dfrac{k_5 D}{P_1}}{L_1 A Q + k_9 A + \dfrac{k_5 D}{P_1}}$$

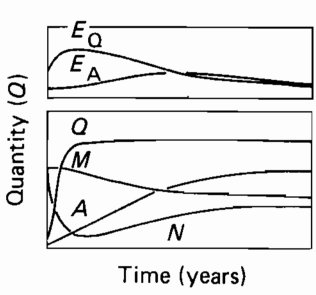
$$\dot{N} = k_3 R N - k_4 N + \frac{L_4 k_5 D}{P_1} + L_5 k_9 A$$

$$\dot{Q} = k_1 R N - k_8 Q - k_2 A Q$$

$$\dot{D} = \frac{k_2 A Q}{p_2} - k_5 D$$

$$\dot{A} = k_6 A Q + L_3 R_2 - k_7 A \frac{k_5 D}{P_1} - k_9 A$$

(a)



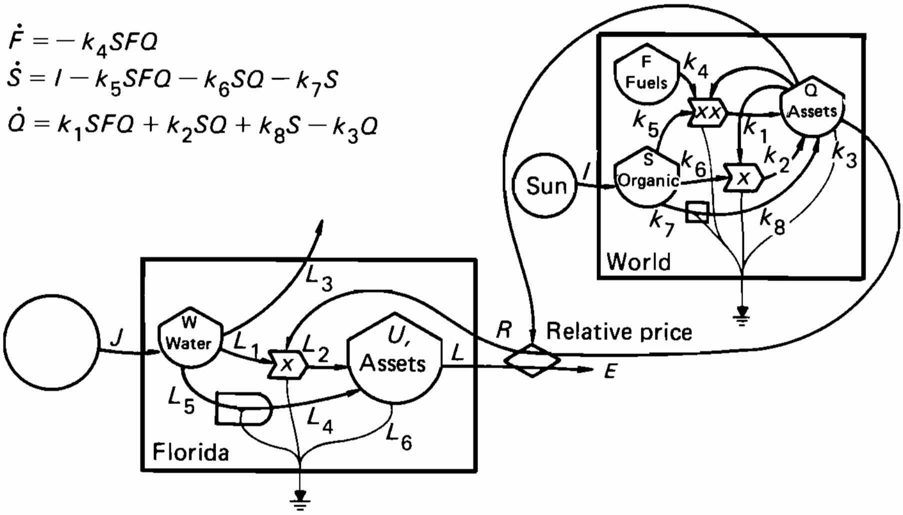
E_Q = Solar energy storage in Q

E_A = Solar energy storage in A

(b)

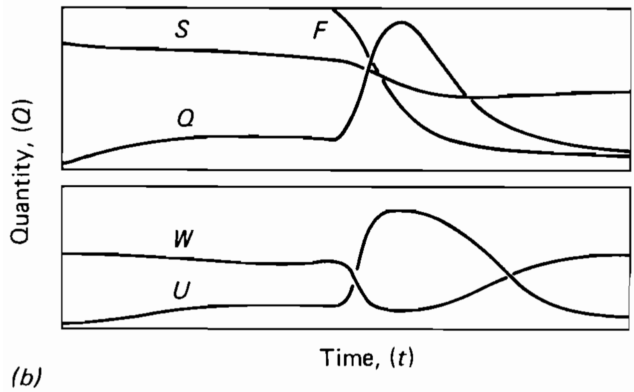
Figure 13.28. **A model of foreign trade including emergy (*EA*, *EQ*) and money (*D*, *M*). (*a*) Diagram; (*b*) simulation.**

$$\dot{F} = -k_4 SFQ$$
$$\dot{S} = I - k_5 SFQ - k_6 SQ - k_7 S$$
$$\dot{Q} = k_1 SFQ + k_2 SQ + k_8 S - k_3 Q$$



$$\dot{W} = J - L_3 W - L_5 W - L_1 WRE$$
$$\dot{U} = L_2 WRE + L_4 W - L_6 U$$

(a)



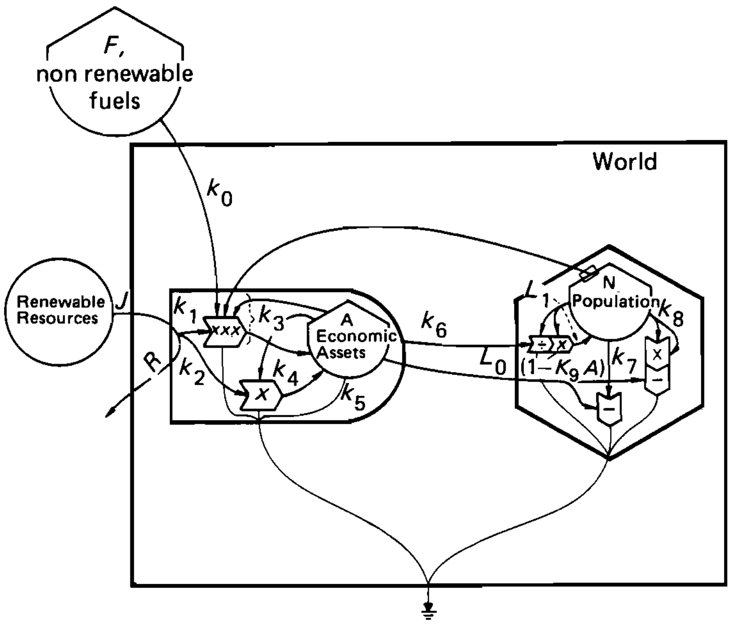
(b)

Figure 13.29. Minimodel simulation of Florida growth driven by world trend minimodel. (*a*) Energy diagram and equations; (*b*) simulation result.

organic biomass. The environmental resource tank (*W*) of the Florida minimodel is calibrated with high quality water availability. In one run the influence of the world centers were represented by a surging growth minimodel.

13.3.5. Asset-controlled population model

Whereas population is aggregated with other assets in many overview models, the model in *Figure 13.30* (Odum and Scott, 1983) has



$$\dot{F} = -k_0 RFNA$$

$$R = \frac{J}{(1 + k_1 FNA + k_2 A)}$$

$$\dot{A} = k_3 RFNA + k_4 RA - k_5 A - k_6 A - L_0 N^2 (1 - k_9 A) - L_0 N(1 - k_9 A)$$

$$\dot{N} = L_1 A - k_7 N (1 - k_9 A) - k_8 N^2 (1 - k_9 A)$$

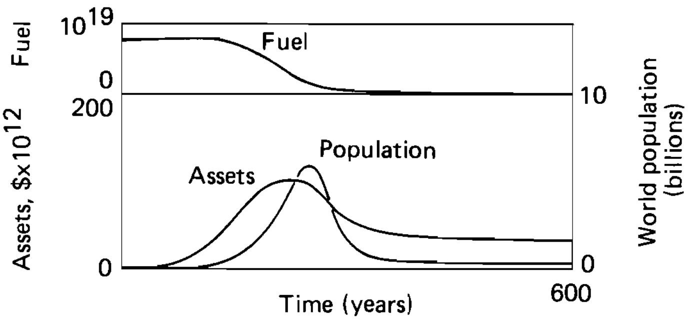


Figure 13.30. An energy-dependent world population model (Odum and Scott, 1983). (*a*) Diagram; (*b*) simulation result).

population as a higher level consumer dependent upon the development of economic assets while also contributing its feedback to that development. Reproduction is proportional to the assets per person but mortality is diminished in proportion to these assets representing the action of public health and medicine on population. The mortality pathways thus inhibited include one that is population dependent and one that is a quadratic drain representing epidemic mortality. When simulated with reasonably appropriate world figures, population crests after world assets, as might be expected. What is dramatic is the sharp turn from steep growth to population decline that is possible in one human generation.

13.4. Concluding Remarks

The test of modeling the future may be to simulate history. Modeling and simulation of history is in its infancy. Some review of archaeological simulation models was conducted in a volume of collected papers edited by Sabloff (1981) with several complex models discussed. For example, a version of Forrester's model was applied to historical Roman times by E.B.W. Zubrow. A minimodel of agricultural displacement of hunting culture was simulated by R.H. Day with another model of primitive social structure of peasants and feudal aristocrats simulating emergence of economic production and control organization.

Simple models can generate smoothed economic trends observed, particularly if sudden changes in external conditions are turned on and off on appropriate time steps. For example, a simulation model of New Zealand (Odum and Odum, 1979: Odum, 1984) simulated agricultural assets and urban development with a colonial period, a period of favorable export–import period, and a recent period of poor balance of embodied energy. In the model a policy to use more of exported agricultural products at home improved the economy.

After two decades of trying to use large complex models for policy and decisions, minimodels are beginning to emerge as compatible with policy overview. Since their simple content is diagrammable, it is understandable, usable, and its limitations apparent. Costs of use are negligible and third parties are not required. Simulation programs are so simple that principles can be thoroughly explored at little cost by busy decision makers on their own microcomputers as if they were giants conducting controlled experiments. More precise and transferable than verbal models, the simulation minimodels may become a new way to educate citizens in public affairs concerning the future.

In this chapter, similarities and differences between economic and ecological systems have been presented in energy diagrams, mathematical equations, numerical simulation graphs, and verbal language. First, we have reviewed the way resources control basic growth and development patterns. In addition, a view of economic–ecological systems as production–consumption chains with feedback properties has been explicated and illustrated. It is apparent from the models in this chapter that economic and ecological growth and development patterns are quite similar. They are all based on some combination of renewable and nonrenewable resources. It has further been demonstrated that modeling the ties between economic and ecological systems in a number of logically different ways, produces several of the often observed behavioral phenomena, such as pulsing and steady states. It has been indicated in several instances in this chapter that a number of cases traditional mathematical modeling approaches in both ecology and economics have worked from premises which do not comply with those laws, however simple they are in a descriptive sense. They are restricted in one particular sense, though, which is relevant for the issues discussed. In macroscopic minimodels the number of connected state variables determines whether the models are capable of producing the more realtistic pulsing graphs. Single state variable models tend to lead to straight line steady states.

A modeling technique has been proposed which keeps track of changing embodied energy and energy transformation ratios during system development simulations. The models in which emergy and transformity are measured demonstrate the change in the quality structure of systems over time. From this we may learn to assess more adequately the potential for, and limits to, growth.

PART III

Policy and Modeling

Introduction to Part III

L.C. Braat
W.F.J. van Lierop

Next to the technical problems of building economic–ecological models for environmental and resource management, as generally described in Chapter 4 and more specifically elaborated for various fields of application in Part II, practical and institutional constraints can be distinguished in applying the models and/or implementing their results. These constraints include bureaucratic and political circumstances and differences in objectives and views between model builders and users. A big gap often exists in the concepts of the real world between academic modelers and modelers for direct policymaking, planners, policy advisers, and the actual policymakers (*see*, for instance, Biswas, 1975; Frenkiel and Goodall, 1978).

The key concept in policy modeling is *effectiveness*. This means that a model must contribute as much as possible to solving the problem for which it is built. The objective of academic modelers, however, is *adequacy*. This is the degree to which a model corresponds with that part of the real world system it is supposed to represent (Majone and Quade, 1981). Striving for adequacy requires striving for a comprehensive model, which then tends to become large and complex and consequently costly. This leads to a trade-off problem in policy modeling studies. A model will definitely not be effective in solving a particular problem if it is not adequate at all. It is also not effective to keep improving the model *ad infinitum* and not use it to contribute to the problem solution. One may conceive of an optimum where the model is adequate enough to produce realistic results and is completed within the constraints of time and financial resources so it can be effective in the policy analysis at hand.

This suggests that when modeling for general policy analysis, one should try to keep the model "simple". This implies striving for a model with a clear, and limited, purpose. Also, there is no excuse for hiding uncertainty in complex detailed models. It is more effective to make uncertainty explicit in alternative versions or scenarios, to design and run rough models early in the project, and to pay attention to articulate documentation. Staley discusses some of these items in his Chapter (14) on the practice of modeling.

In addition, the effectiveness of a model may be increased by involving policy advisers in the model design. In the last few years it has become clear to many in both the academic and the policy world that a major problem is lack of communication before, during, and after the modeling project. The communication problem results from the inability of modelers to translate their views, biases, and products into a format which is comprehensible for policy analysts, and from lack of or obsolete and outdated professional training in the policymaking environment (Biswas, 1975; Holling, 1978; Klaassen, 1980; Environment Canada, 1982; Dror, 1983, 1984). This summary of institutional constraints to multidisciplinary modeling indicates the need for improvement of interaction channels between academic modelers of different disciplines and between them and policymakers. Pearse and Walters present some suggestions to that end in Chapter 15 on the application of economic–ecological models.

ECONOMIC-ECOLOGICAL MODELING
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CHAPTER 14

The Practice of Resource Modeling

M. Staley

14.1. Introduction

This chapter discusses the practice of ecological modeling in the context of economic systems. In this discussion we will give examples of some approaches to ecological modeling that have failed to capture the important dynamic behavior of the problem because critical links with economic systems have been left out of the analysis. This chapter will also outline some important uncertainties in ecological systems and describe the problems those uncertainties create in ecological modeling and analysis. Finally, suggestions for improving the ability of ecological modeling to provide input into environmental and resource management and policy design are discussed.

14.2. Ecological Models

Ecological models have developed along two lines. One deals with the harvest of biological populations; the other with ecosystem responses to pollution and other habitat disturbances. In the area of population harvest models, fisheries models have received a great deal of attention and have had perhaps the most significant impact on management and policymaking in the field of ecological modeling.

14.2.1. Renewable resource models

Animal population models fall into two broad categories: stock-recruitment models and cohort models. Stock-recruitment models derive from the assumption that the recruitment of young animals into the harvestable population is significantly influenced by the stock of adults left to reproduce after harvesting. These models are used to study the appropriate harvest rate to ensure sustainability of the populations as well as to provide better, or perhaps optimal, yields. The populations are represented either by numbers of individuals or by the weight or biomass of the living population. Sometimes the populations are disaggregated according to age or size to represent more realistically the biological processes of birth, death, growth, and aging. However, recent results (Ludwig and Hilborn, 1983) indicate that aggregate measures of the total population are as good as detailed age-structured models for estimating optimal harvest rates.

Cohort models have been used with those species of fish for which there is little correlation between the size of the adult population and the resulting recruitment. These models are used to estimate the best time or season to fish and the best body size to harvest. In both the stock-recruitment and the cohort models the most common assumptions include a constant environment. All the processes related to the animals' physical habitat, food resources, competitors, and predators, are embodied in a few parameters of growth, fecundity, and mortality that usually remain constant through time. These assumptions can lead to erroneous predictions and possibly dangerous recommendations for management. Another assumption that is usually made is that the process of harvesting by humans is totally controllable. There is ample evidence that social and economic pressures often make it difficult if not impossible to hold hunting or fishing effort at an optimal or even a sustainable level.

Forestry and agriculture are areas where a significant amount of ecological modeling has been done. Forestry models have been used to simulate and predict the behavior of anything from a single leaf up to an entire forest. The focus of most forest models is to assess the best age of harvest and to predict the effects of various silvicultural practices. In agriculture, the major areas of interest for modeling include nutrient dynamics, fertilizers, crop rotation, and cultivation techniques. Pest control has also been the subject of modeling in both forestry and agriculture.

14.2.2. Environmental quality modeling

The other major line of ecological modeling deals with ecosystem responses to pollution and other habitat disturbances. In this area there is enormous diversity in purpose, approach, and quality of the models. Almost every major industrial or urban development that has been proposed during the past decade in North America has had associated with it at least one ecological model to help assess the environmental impacts. In addition, the major classes of environmental issues (point source and nonpoint source air and water pollution, estuary and wetlands development, water impoundment, entrainment, and diversion, etc.) have stimulated the development of many different, often contradictory, models of biological and physical systems. Some of these models have found their way into the court room in the USA. This has led to what is known as the battle of print-out. In cases where models have been used to argue controversial points of view, it is difficult to say whether the models have helped to clarify the issues or have merely added more confusion to difficult situations.

14.2.3. Summary

Approaches to ecological models tend to cluster around the characterization of the state or stock of the ecological resources. Most ecological models use the following concepts or currencies to represent stocks and flows:

(1) Populations – the populations are represented by numbers of individuals or total biomass.
(2) Energy – the amount of energy embodied in a trophic level in the ecosystem or passes between levels. Energy is sometimes measured in physical units (calories) or in mass of carbon.
(3) Nutrients – many models assume energy and carbon are in abundant supply and that the limiting factor in many ecosystems is the supply of nutrients. This is particularly true in some aquatic and forestry systems and most agricultural systems.

Each of these approaches is best suited to different systems and the different questions that are asked of these systems. In practice, ecological models now use a mixture of these systems. In practice, ecological models now use a mixture of these ideas to build on the strengths, where possible, and avoid the weaknesses.

14.3. The Practice and Limitations of Ecological Modeling

In a short review, it is impossible to assess fully the successes and failures of ecological models. Therefore, only a few major problems with ecological models will be described here. These problems will be illustrated with case examples of applied ecological models.

14.3.1. Fisheries response

A common problem with ecological models is that the modelers ignore or play down the economic system to which the ecological system being modeled is intimately tied. This is an inevitable problem when the information for the model comes from a narrow set of disciplines. One example of this problem involved a detailed biological model of the recreational fishery for chinook and coho salmon on the west coast of Canada (Argue *et al.*, 1983). This model was developed to help establish new regulations for the sport fishery. It was thought that this fishery was catching a large and growing proportion of the fish and endangering the sustainability of the catch and the viability of the stocks. This analysis included a detailed look at the effects of various limits on the minimum and maximum size of fish that could be caught, limits on the number of fish allowed per fisherman, and the limits on the seasons in which fishing could take place.

Originally, the analysis assumed there was a constant pool of fishermen that would fish in historical patterns under any changes in regulations and that resulting changes in the quality of fishing would therefore not change the amount or timing of fishing. For example, if a set of regulations reduced the catch a fisherman could expect in a day's fishing, he would not change his attitude and would continue to fish as he had been observed to do historically. Furthermore, if a set of regulations were successful in saving young fish from capture, so they could grow to become larger fish later, fishing effort on those bigger fish would not change. It should be pointed out that these assumptions of constant or unresponsive fishing effort are normally made in most fisheries models. Under these assumptions, the model of the chinook and coho fishery predicted that significant reduction in harvest could be achieved by relatively minor changes in size limits and bag limits.

Fortunately, during the course of the study, these assumptions were questioned by an avid sport fisherman involved in the analysis. Would the fishermen change the amount of time they were willing to fish in response to changes in the perceived quality of fishing? Some evidence existed to indicate that the number of fishermen fishing was correlated with the success or catch per unit effort [CPUE (*Figure 14.1*) corrected for the effect of weather]. When this relationship was included in the model, a very different picture emerged. Those sets of regulations which appeared good with no fishing effort response failed to produce significant increases in the numbers of fish escaping the fisheries when the assumption of effort response was included in the model. By ignoring the unregulated economic behavior of the fishermen, the conclusion of the modeling exercise would have been grossly in error and dangerously misleading to the managers responsible for the fishery resource. There is still substantial debate about the exact nature of the economic response to the quality of fishing. However, salmon fishery models that ignore this important process are not regarded as credible for policy development.

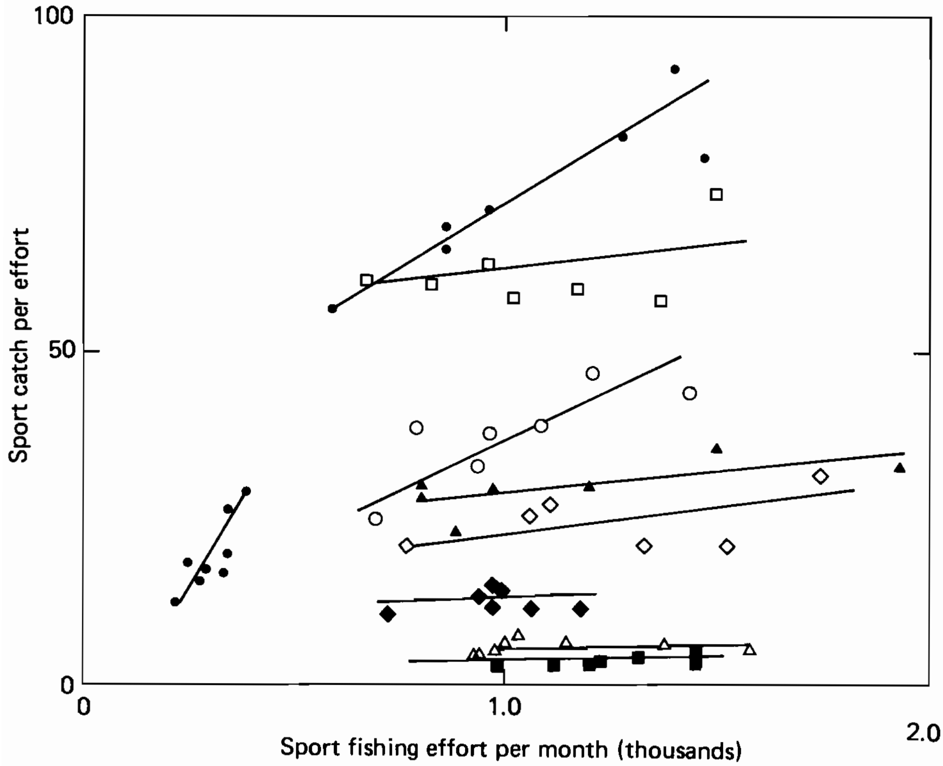

Figure 14.1. Sport fishing effort per month (thousands).

14.3.2. Cultural response

Another case example involves large-scale energy development in northern Canada. For more than a decade, several government agencies, energy corporations and public interest groups have been trying to assess the environmental and social impacts of major oil and gas developments in the Mackenzie delta and Beaufort Sea area of the Canadian Arctic. Much of the concern surrounds the native people of the area and their traditional resources and life-styles. Several models have been built to assess the impacts of increased transportation activity on whales and birds, dredging of harbors on fish and other planktonic and benthic animals, as well as the effects of oil spills and their clean-up on every conceivable biological resource. Much of the focus of the analysis has been on the direct and indirect effects of habitat disturbance on fish, birds, and marine mammals that are harvested by the native people. So far, only marginal effects of habitat disturbance associated with the oil and gas activity have been demonstrated.

In concert with the analysis and modeling of the ecological systems, there have been studies of the social impact of development on the native people. One of the dominant issues is the effect on traditional life-styles of employment of native people in the oil and gas activity. There is considerable concern on the part of the natives and others that much of the northern culture and heritage will be lost to the southern industrial life-style. Traditionally, much of the native people's time has been spent in hunting and fishing for subsistence. In some population studies of fish and marine mammals and birds, it is indicated that the major agent of mortality is due to hunting and fishing. Any significant change in the hunting and fishing could have substantial effects on the populations of whales, seals, birds, and fish. These effects could overshadow the impacts of habitat disturbance and related industrial activity.

One unanswered question is whether employing the natives in the oil and gas industry will increase or decrease the amount of hunting and fishing they do (*Figures 14.2* and *14.3*). On the one hand, working in an industrial job reduces the time available for hunting and fishing. With the money earned from employment, food, and other goods can be imported from southern markets thereby freeing these people from dependence on hunting and fishing. However, there is another side to the coin.

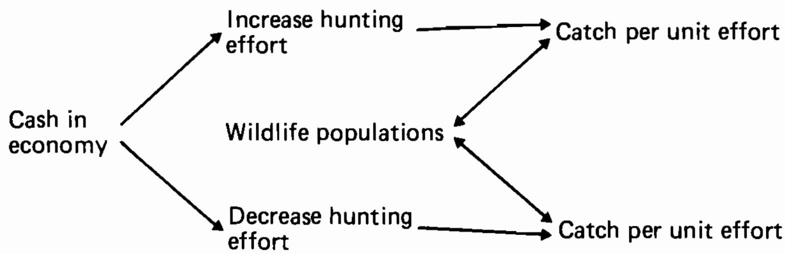

Figure 14.2. Alternative hypotheses about the effects of increased cash in the local economy on wildlife populations.

Stories are told about working natives hiring others not employed in the oil and gas industry to do their hunting and fishing for them. The earnings from the job have allowed some people to buy equipment such as snowmobiles to increase their efficiency of harvesting and perhaps the total harvest. One story tells of a native who worked on a

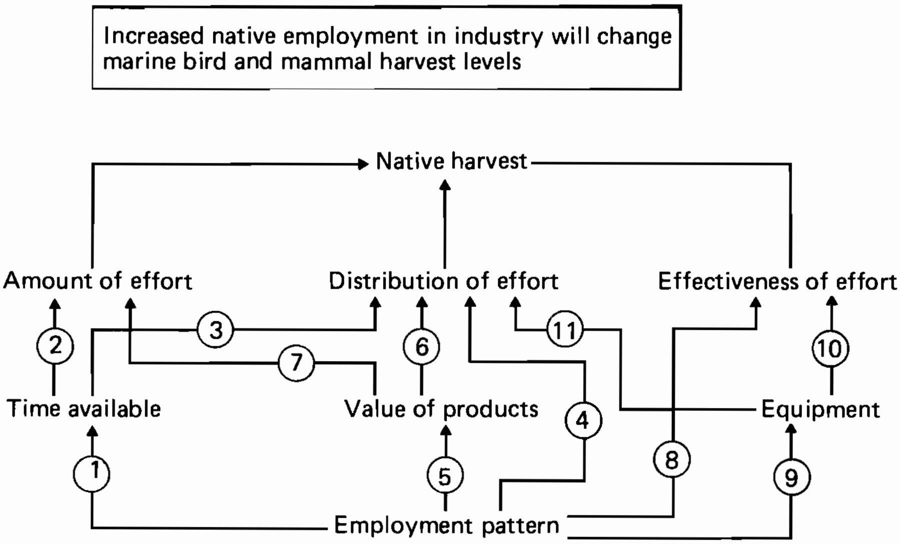

Figure 14.3. Potential effects of increased native employment in industry on marine bird and mammal harvest.

drilling rig all summer. One weekend in the fall he chartered a large twin-engine aircraft. He was able to do all his hunting for the year in just a few days. The story goes on to say that he was able to fill the aircraft with geese and caribou several times over, resulting in a higher harvest than he would have been able to achieve in an entire year by traditional means.

It may be that the major environmental impact from oil and gas development in Arctic Canada will come from the unregulated human response and economic behavior of the native people. The ecological models used to predict the effects of pollution and habitat disturbances could not have included this important link between the development and the environment.

14.3.3. Ecosystem response

Significant difficulties with ecological models surround fundamental uncertainties about the behavior of organisms in situations that they have not experienced or have not been observed to experience. Predicting the behavior of systems in new and perhaps exotic conditions is the primary reason for building models. Most models, and particularly those used in environmental impact assessment, are meant to be laboratory worlds that try to simulate the behavior of a system under circumstances that may be present after some disturbance or alteration to the environment. This is done to shed some light on the hazards and benefits of venturing into the unknown. Unfortunately, as the basis for the models we have only data and assumptions generated by our experience and imagination. The latter is often constrained by the former.

Uncertainties show up in ecosystem models as questions of whether a population will survive or flourish when restricted, due to pollution or construction, to habitats that traditionally were less preferred by the population. During a three-year research program on the effects of oil and gas development on the north coast of Alaska, a large ecosystem model of a barrier island lagoon system was developed to help coordinate the field research and to predict impacts (*Figure 14.4*; Truett, 1980). Several times the modeling exercise ran into difficulty, because the data and observations that were available included habitats that the organisms (fish and birds) had been seen to use. There were no data on what the animals really ***needed***, only what they appeared to like.

This problem led to two quite different recommendations for the deployment of drilling operations. These shallow lagoons are very abundant in fish, birds, and the food organisms that support them. The major food resources are small marine invertebrates (shrimp-like creatures) that are very abundant both in the lagoons and offshore. If the animals are dependent on the lagoon for their feeding, then any oil spilled from drilling inside the lagoons would be disastrous for the fish and birds. However, if the birds and fish could feed outside the lagoon as they are known to do elsewhere, then it may be more important to protect the bulk of the invertebrate resource that lives and breeds offshore of the islands. One suggestion was to concentrate all the drilling activity inside the lagoons and use them as a natural catchment for oil spills. None of the models that were built of the ecological interactions of birds, fish, and invertebrates were able to assess the best option conclusively. No matter how good the models are, there is no way around such basic uncertainties as the habitat requirements for animals.

14.3.4. Problems of extent and detail

Another difficulty ecological modelers face is setting the bounds of the problems and the models. For models to be useful in assisting management or setting policy, they have to appear pertinent and meaningful to the people who are responsible for management or policy development. Usually these people have a broad set of concerns. These concerns include economic and social impact of actions, administrative feasibility of implementing conclusions of models, and political viability of conclusions when presented to constituents.

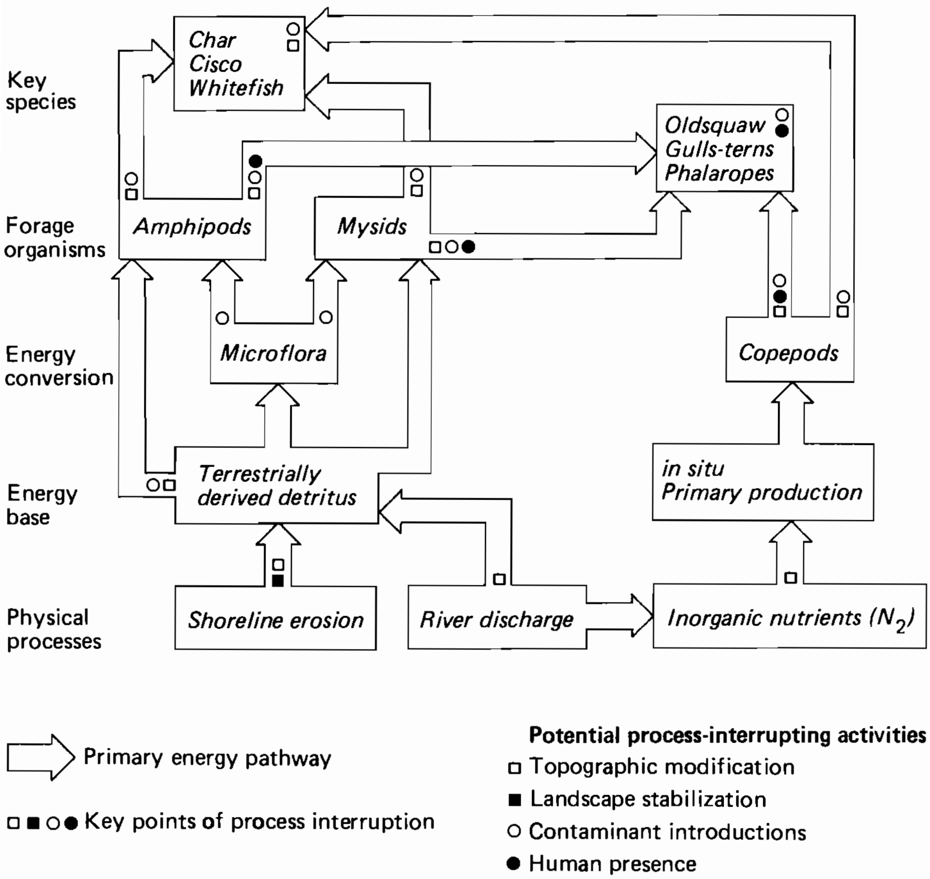

Figure 14.4. Known and suspected critical energy pathways that support the key species in the area of Simpson Lagoon, Alaska. Places where energy pathways and system components are likely to be affected by OCS development activities are shown.

Often these concerns are not those of the ecologists and ecological model builders. As a result, the models tend to address the problems on a scale of time or space that is too narrow or a level of detail that is too complex to be useful within the broader context of management and implementation of policy. For example, models of a single tree or of one square meter of tundra are of great interest to disciplinary research but are difficult to apply to regional forest management or land use conflicts.

Many models fail to produce outputs on indicators that are relevant to the economic or political context of the problem being modeled. For example, many environmental monitoring programs are being proposed where models are used to evaluate changes in organisms. These changes are easy to measure or easy to attribute directly to changes in the environment being monitored. Unfortunately, the organisms being monitored are not of economic or political concern. The respiration of sea worms around a sewage outfall site may be of profound biological interest. However, connecting the worms to migratory fish or birds that are either visible or commercially harvested is difficult to do. Therefore, a sound ecological model of a potentially important component of the ecosystem will not necessarily succeed in influencing management or policy because the output and conclusions of the model are not germane to debate over management or policy.

14.4. Improvements for Ecological Models

The ecological model problems discussed so far fall into two classes:

(1) The problem of uncertainty about the behavior of ecological systems that is, in principle, irresoluble with available data and sophisticated modeling and analysis.
(2) The connections between ecological and economic models.

14.4.1. Adaptive management

Adaptive management, or more precisely, adaptive control has been suggested as one of the best ways of coping with problems of uncertainty (Walters, 1986). The general concept of adaptive control is to use management or industrial and urban development as experiments explicitly designed to generate information and resolve uncertainties by direct intervention into the system. The choice of management action has three parts:

(1) The best certain equivalent – or, what would be the best thing to do if the current best estimate of the response of the system were in fact the certain response?
(2) Conservative risk taking – this is part of the policy or management plan that is sensitive to the uncertainties and tends toward caution in the face of the unknown.
(3) Active probing – this is the adaptive part of the solution: when the system's state or behavior is so uncertain that knowledge, which can only result from trial, is more valuable than caution.

Fishery management is an appropriate area for the application of adaptive management. In fisheries, gross uncertainties about the basic productivity of the stocks is common. Furthermore, no amount of detailed biological research or sophisticated modeling is going to be able to reduce the uncertainties and predict the behavior of the populations very far away from historically observed levels. Only by probing with significant changes in harvest levels and practices can sufficient contrast be observed to make reliable estimates of important parameters such as productivity.

Adaptive management suggested a different approach and style to modeling than has been practiced traditionally, particularly in fisheries. Instead of modeling to build the best simulation of the system and solve for an optimal policy, modeling with adaptive management in mind means looking for opportunities for management to provide experiments (experimental design) and learning to live and cope with uncertainty and the inability to predict.

14.4.2. Ecological–economic connections and policy relevance

The second major problem area with ecological modeling involves poor connections between ecological models and economic models and the failure of ecological models to address appropriately the scope of problems for management and policy design. The path to improvement lies in restructuring the process of model building. The process of model building must maintain a broad perspective that keeps the management and policy issues in perspective at all times. Also, the process must be able to engage all the relevant disciplinary actors and provide a common framework for communication and cooperation.

One approach to model building that has had some success at maintaining interdisciplinary involvement in the modeling process is Adaptive Environmental Assessment and Management (Holling *et al.*, 1977). In this process, a series of workshops are designed and used to build the model. In the workshops, an explicit attempt is made to involve both the various disciplines from all appropriate areas of expertise and the policy and management people responsible for decision making in the design and construction of the model. This process of model building has been very successful in a number of situations (Environment Canada, 1982) at building the links between ecological analysis and economic dynamics as well as maintaining the interest and involvement of the real actors in the modeling and analysis.

14.5. Conclusion

Applied ecological modeling has played and will continue to play a significant role in management and policy design for renewable resources and environmental impact assessment. New techniques and approaches coming from mathematics and computer science will continue to enhance the capabilities of ecological modeling. There are at least three improvements to the modeling process and the concept of environmental and resource management that will further enhance the usefulness of this important tool.

Firstly, there is a need to ensure that the modeling process maintains a focus on its objectives, be they policy design, research planning, or scientific curiosity. This will ensure relevance of the results to the problem.

Secondly, there is a need for an open process of model development that involves all the important actors and expertise. This will minimize the risk of missing important connections such as the behavior of the economic system to changes in the ecological system. And finally, there is a need for a more innovative and experimental approach to management and policy testing for ecological systems. The major uncertainties and risks will never be understood without major experiments.

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CHAPTER 15

Perspectives on the Application of Economic–Ecological Models

P.H. Pearse
C.J. Walters

"The more he looked inside, the more Piglet was not there"
(Pooh, looking for Piglet at Piglet's house)
A.A.Milne

15.1. Introduction

Most of the models reviewed in this volume were motivated by policy problems, and were intended to assist in resolving them. Many were actually "applied" in the sense that their predictions were discussed and debated in real decision-making contexts. However, it is difficult to assess just how much impact such model applications have had, because the complicated interplay among many interest groups and policy-making authorities that surrounds research and environmental issues means that model prescriptions are never followed precisely or without compromise. Moreover, in the absence of a clear empirical track record of past performance, it is difficult to provide a credible perspective on how this field of study is progressing. At present, the best we can do is to comment on the spectrum of applications that have been attempted to date, and speculate about how future efforts might be made more effective by encouraging model builders and analysts to examine more carefully how their products are seen by those involved in decision making.

Too often we see modeling efforts that have apparently been aimed to influence some mythical, omniscient, and all-powerful decision maker, who can and will respond rationally to the logic of a model.

Usually there is simply no such person: all the carefully reasoned argument falls not on deaf ears, but on ears that are attuned to only part of the findings, or on no ears at all. Modelers must learn to live with and take advantage of this state of affairs, or they will continue to have frustratingly little impact on decision making.

In this chapter we begin with a rough classification of the IvM/IIASA survey responses in terms of the detail of model predictions or prescriptions and the complexity of the decision environment in which the models might be applied. This classification indicates that most of the work so far has been aimed at what are probably the most difficult situations for decision making, namely those involving strategic choices by many individuals, agencies, and institutions interacting in concert or, more often, in conflict. We then identify some obstacles to acceptance of models in such circumstances, with reference to the attitudes and behavior of the people involved, and we discuss some modest steps that model builders can take to gain for themselves a better audience in the courts of decision.

15.2. The Range of EEM Applications

For any economic/ecological "problem", it is obviously possible to design a wide range of models that might be of some value in decision making. At one extreme, very simple calculations might demonstrate basic constraints and trade-offs among performance measures, and thereby stimulate a search for more realistic or imaginative policy options. At another extreme, a very detailed model might try to represent how a host of operational tactics (local, short-run decisions, investments, regulations, etc.) act in concert (or competition) to produce overall results that might be quite different than would be expected from considering each tactic in isolation. Between these extremes, models may ignore various tactical details about how to achieve certain results (such as particular exploitation rate in a renewable resource) in order to explore more readily the consequences of various strategic decision options.

Models of differing detail are likely to appeal to different actors involved in decision making. Thus a very general model may serve the needs of high-level policymakers who would be bored by a very detailed model which would excite those responsible for daily management. Failure to identify precisely whose interests are to be addressed is perhaps the most common error made by academic or inexperienced model-builders.

A second basic dimension of any problem is the complexity of the decision-making environment, measured in terms of the number of people and institutions that are implicitly party to decisions. At one extreme in this dimension are problems of single entrepreneurs, such as farmers or ranchers, who have wide decision-making authority on a restricted spatial area. At the other extreme are global problems such as acid rain, where many governments and many more agencies must interact with industrial and public interests to make and implement a policy change.

But the most awkward cases to model well are problems of intermediate spatial scale, such as regional development planning, where lines of authority and influence are usually poorly defined and may shift rapidly as various groups clarify how their own interests may be affected. Even within government agencies, that supposedly have well-defined objectives and policy instruments, there are usually conflicts of interest, competition for limited resources, divided responsibilities, and so forth. At intermediate spatial scales, there is often no clearly identifiable "client", or rational decision maker, who can embrace a model for his decision making. Instead, the model may be used in varying degrees, and at surprising times, by all sorts of actors. It is worth noting, incidentally, that this description of the range in complexity of decision making environments is equally valid for market and centrally planned economies. The latter may have larger bureaucracies and hence appear to function with clearer lines of authority and responsibility, but decision making may in fact involve complicated bargaining among players whose interests go far beyond the problem at hand.

An attempt to classify roughly the models covered by the IvM/IIASA survey responses according to their predictive or prescriptive detail and their decision complexity is shown in *Figure 15.1* (*see* Appendixes A and B). The models are concentrated in two clumps. One is a collection of rather detailed models for management of agricultural enterprises; these models are clearly intended for use by individuals in detailed planning of operational decisions such as when to plant crops and how to apply pesticides. We shall not comment further on such applications in this chapter, since it is easy to see how they can and will be used extensively in the future.

The other clump of models deals in much less detail with larger, regional problems, such as fisheries, water management, and pollution control. These have often been developed for the use of single government agencies, but in fact involve decision variables that can be influenced by or are the responsibility of other agencies and interest groups.

Figure 15.1 reinforces an impression one gets from reading many reviews about models being developed; namely that the model development process often involves first picking a basic spatial scale, then working toward an increasingly detailed representation until various technical constraints (such as computer time available) are reached.

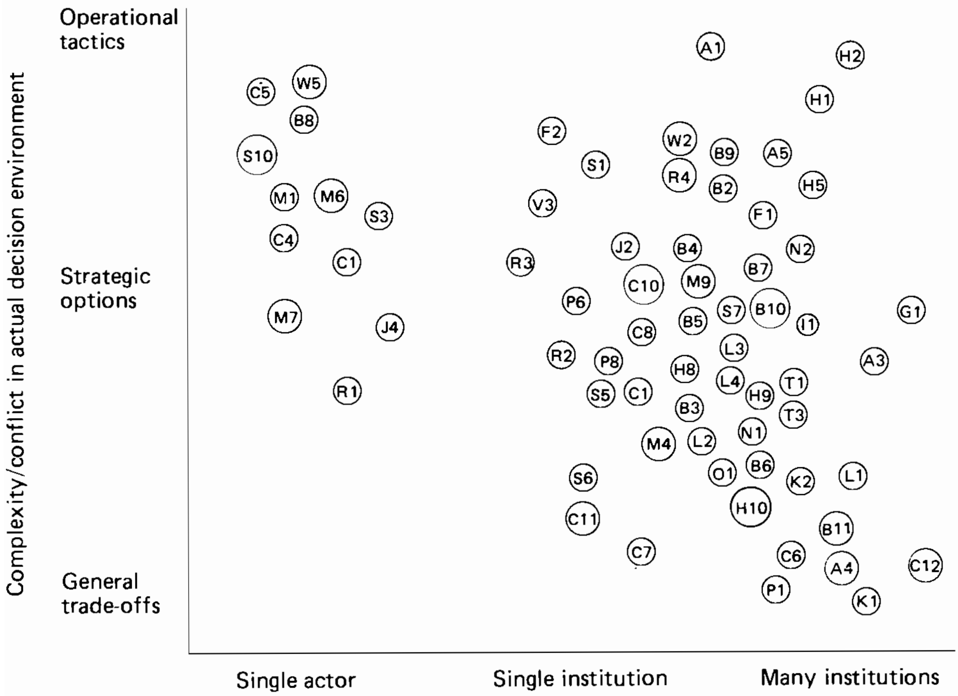

Figure 15.1. A rough classification of survey responses in terms of their detailed predictions and number of players who might use the model results in decision making (*see* Appendix B).

Thus we see no models in the lower left corner of the figure. There are two possible conclusions: either individuals and single institutions have few strategic concerns worth modeling, or modelers are not very good at recognizing and responding to such concerns.

15.3. Model Application

There is often an implicit judgment that the extent to which a model is used to influence policy decisions is a measure of its success. In other words, the model builder has succeeded if his model is used by policymakers and, by implication, has failed if it is not. But whether a model is a good one and whether it is used are two separate questions. A good model will characterize a problem accurately, and in such a way that it can be better understood and analyzed. But this does not guarantee that it will be used in making policy decisions. It may be inadequately exposed, hence ignored altogether; or it may have been developed before its time (e.g., Ricker's stock–recruitment model); or it may become a valuable building block for future models (e.g., Holling's experimental component analysis). Conversely, there are many examples of poor or inadequate models which have had considerable influence on government policies (e.g., the economic–environmental theories of the eighteenth century Physiocrats).

In other words, the merits of different models cannot be judged by comparing their relative acceptance by politicians or policymakers. Nevertheless, scientists are not indifferent to the ultimate use of their work: widespread or influential application is gratifying, and it brings recognition as well. Moreover, models of economic/ecological systems often pose hypotheses that are testable only within the framework of carefully designed policies. Thus most model builders are eager to see their work applied.

Why then are most model building efforts greeted with far less enthusiasm than their proponents had hoped and expected? Most models end up buried and forgotten in academic reports, after perhaps serving as a focus for a few spirited debates. Much more rarely, some general or qualitative model conclusion becomes the basis for a strategic policy guideline or basic decision. Almost never are detailed prescriptions adopted comprehensively. To many modelers, having their models "actually applied" has come to mean little more than "presented for discussion".

The failure of models to be welcomed by policymakers often stems as much from the model builder and the human context of the problem, as from the model itself. To begin with, the model builders, expectations may not be entirely reasonable; as mentioned above, the circumstances may not be ideal for the model to shine as a policy tool. Secondly, modelers often deserve the reactions they receive because they have not worked with enough sensitivity to the concerns of their audience or the structure of the decision-making environment. Finally, any model dropped into a complex policymaking setting will threaten many of the people already involved. Improving the model itself may only make it more threatening, and efforts to respond to criticism may aggravate the situation by raising a number of obstacles not even related to the quality of the model. The following section addresses some of these problems.

15.4. Obstacles to Acceptance of Models

Our review of modeling experience suggests several common obstacles to the acceptance and successful utilization of models by policymakers. In identifying these, we also offer a few suggestions for developing a more constructive relationship between modelers and clients.

15.4.1. Resistance to new technology

There is, undeniably, widespread apprehension about new-fangled computer technology. Those who are not familiar with it understandably view it with anxiety and suspicion, despite efforts to reduce the mystique of computer models by referring to them as "glorified accounting systems" or "tools for simply keeping track of interactions". Indeed, even knowledgeable people are sometimes anxious about someone else's "black box", especially when it is intended to analyze a problem in which they have an interest. This reluctance to accept strange technology is to be expected. After all, the kind of model-building described in this volume was unknown a generation ago. For some time, key decision makers will continue to be drawn from a generation that never learned to use computers, and they are unlikely ever to have the time or inclination to do so.

One way to alleviate apprehension of computer technology is to involve the potential users as much as possible in the model building process. This enables them to see first hand what goes into the making of a model, and often transforms their initial resistance into enthusiastic acceptance. Such participation in the model creation makes it easier for users to have some direct input to the model, which further promotes their acceptance of it.

15.4.2. Opposition to change

Models are usually designed to help analyze the consequences of policy changes, and hence to assist in making changes. But typically some groups will be threatened by change, and will see their interest best served by preventing change. An obvious tactic for them is to deliberately confuse or deflect orderly debate about options for change, hoping that if confidence in progress is undermined and uncertainties heightened there will be no decision at all. So we find the examination of model results relating to the more contentious policy issues marked by apparently random attacks on the decision process, including such matters as the composition of the groups brought together for the purpose, the competence of the analysts and, most commonly, the reliability of the data used in the analysis. (The authors of this paper were recently involved in a policy review of a highly disputatious fishing industry, which vividly illustrated these tendencies.) The capacity of computer models to compile and analyze large volumes of data is undoubtedly one of their great advantages, but it also leaves them more vulnerable to critics bent on exaggerating their empirical imperfections.

This problem, like some of the others listed below, has no specific solution. The key point is that the modeler who is sensitive to the possibility of such entrenched opposition can look for and take advantage of every opportunity to defuse it during the course of the analysis. Every effort should be made to involve opposition groups in the study in the hope that an inside look at what is going on and a feeling of participation in the project will enable them to find advantages in cooperating. Flexibility on the part of the modelers is also important. The ability and willingness to change models quickly enables them to respond effectively to the range of changing questions a staunch critic is likely to pose.

15.4.3. Opposition to compromise

Many regional and other types of models purport to balance conflicting interests in search of the maximum aggregate or social advantage. This inevitably involves compromises and trade-offs. But the main concern of interested parties in these situations is not in being weighted reasonably; that will be regarded simply as an incidental result of their attempts to elevate, strengthen, and protect their particular positions, even at the expense of others. Because the interested parties cannot usually be expected to act in concert to maximize the collective benefit, their approach to the policy issue is incongruous with that of the analyst and his model.

Here the best prescription is to involve representatives of potentially conflicting interests in the model building process right from the start. Once everyone has had a chance to state their positions and to display their intransigence, they have to cooperate at least to the point of helping design a model and a set of policies to test it with. This process gives participants a better understanding of the dynamics of the system and of other points of view, which in turn decreases much of the initial inflexibility to compromise.

15.4.4. Suspicion about objectivity

A related, but separate matter is the genuine difficulty that parties involved in conflict have in accepting the objectivity of any analyst. Models of the kind discussed here typically deal with issues involving conflicting interests, and the parties to the conflict have often persuaded themselves, through prolonged debate, of the strength of their claims and the reasonableness of their (conflicting) proposals. For them, an analyst's assertion of scholarly objectivity in his findings will be accepted cautiously if at all. They look for allies and are wary of opponents; in this atmosphere, interest groups respond to analysis mainly according to which of these groups it supports.

It hardly needs to be added that modelers sometimes justify suspicions of bias, and that it is too much to expect of public interest groups to distinguish between those that do and those that do not. Any effort on the part of the model builder to better communicate the assumptions within his analysis and the implications of those assumptions will help the user judge the objectivity of the model. Even more effective is to involve potential users in the building of the model. Then they can not only see why certain assumptions were adopted, but also try to incorporate some of their own assumptions.

Clarification of advisory and advocacy roles also helps to remove suspicion about objectivity. For example, an analyst can limit himself to an advisory capacity by addressing only the concerns under discussion by his client, or he can take on an advocacy role by trying to convince his client to address different issues. Either role may be valuable, but modelers should decide explicitly which to take since the tactical requirements are quite different: the active advocate may be forced to use the same tricks of showmanship and persuasion as his competitors. Analysts who explicitly and deliberately take an advocacy role will be least vulnerable to suspicions of hidden bias on the part of public interest groups, but they face obvious limitations in aiding policy formulation.

15.4.5. Commitments to established concepts

For any large-scale problem there are bound to be some who have struggled to understand it (often over much of their careers), developed what they consider to be satisfactory ways of dealing with it, and are proud of their achievements. Forest management agencies throughout North America offer many rich examples in their pursuit of sustained yield principles. New approaches and concepts are often seen by such individuals as personally threatening. Moreover, the use of analytical models for policy analysis, and the means of responding to them, often do not fit into the established bureaucratic structure and its procedures for decision and action. And there may be no one willing and able to introduce such changes. Few modelers, approaching a policy problem anew and often briefly, seem to appreciate how tightly the entrenched institutions and bureaucracies cling to their established concepts, approaches, and procedures, even about relatively innocuous issues.

Many decision situations deteriorate into "either–or" debates over extreme options or basic uncertainties. Model building directed only to the exploration of recognized options is not likely to be of much value, since the problem is usually not just to quantify a selection criterion or measure of risk. A much more valuable approach is to use the modeling process and its products as means of stimulating a search for more imaginative policy options, using previously unnoticed policy instruments or combinations of established actions and procedures that have surprising effects when used together.

15.4.6. Truncated perspectives

Regional models are usually designed to reveal results in the form of aggregate statistics that incorporate heterogeneous collections of economic and ecological microsystems, such as firms, spatial patches, and so on. In contrast, the direct experience of most people in private interest groups and bureaucracies is limited to one or a few microsystems, and so they are often puzzled by findings that do not reflect, or even run counter to, their own experience. Thus policy debates may degenerate into exchanges of anecdotal evidence about fragments of the problem, even though those involved may honestly try to grapple with the whole. This should be dealt with outside of the analytic process, by discussion using concrete examples to convey the perspectives of the problem. Technical language and abstract generalities should be avoided.

15.4.7. Preoccupation with distributional effects

Political decision-making is generally much more influenced by the distributional impact (who will gain, who will suffer) of new policies than with efficiency gains measured in economists' or ecologists' terms. Yet most economic/ecological modeling, especially of renewable resource problems, has dealt primarily with issues of efficiency, average performance, and risk. Politicians, especially, are much more sensitive to concerns that arise from perceived self-interest, noted in the preceding points, than to arguments about rather abstract concepts such as maximum long-term productivity or efficient resource allocation.

15.4.8. Apparent irrelevance

The classic error of policy analysts is to fail to identify what questions are of primary interest in the first place and, accordingly, how to design the analysis in order to reveal the most useful answers. For example, imagine a policymaker who wants to know how many fish should be caught. The analyst replies that he doesn't know but that he could tell him the optimum age structure of the stock instead. This failure to respond to the questions being asked of the model continues to make modelers look foolish or irrelevant to policymakers.

In addition, models are often presented in numbing detail (reflecting a preoccupation with the model itself rather than the issue being analyzed) and findings are frequently presented with spurious precision. Sometimes technically weak modeling is disguised by these means, which nevertheless aggravate apprehensions and criticisms from those who are expected to use the results. At the other extreme, results are sometimes presented with so many qualifiers and disclaimers that the users can find no general conclusions from which they can take guidance.

15.4.9. Blurred decision-making authority

As we have said, models often mistakenly presume a single decision maker, capable of weighing all the evidence and alternatives and determining the best course of action. But responsibilities in large problems are typically divided among several authorities whose primary interests differ. Moreover, those with responsibility for decision are not likely to act without reference to subordinates, advisors, and outsiders. Even more complicating is the tendency for power and involvement in a major issue to shift as evidence reveals possible impacts on new parties, all with their own avenues for bringing their influence to bear on it. These relationships are exceedingly difficult to respond to in designing models to deal with economic and ecological issues.

However, modelers are beginning to recognize that each of their clients has limited authority, and it is necessary to look beyond overt lines of organizational responsibility in order to understand decision-making environments. This recognition has stimulated a search for novel approaches, such as modeling workshops, to involve more of the potential clients in various phases of model development and analysis. These approaches may gradually become familiar, standard operating procedures for defusing opposition and conflict before misunderstandings become too deeply entrenched.

15.4.10. Failure to communicate

Since model building requires considerable time and training, it is usually done by specialists rather than by policymakers themselves. And usually the communication between model builder and client is far less than perfect. In fact, many analysts seem deliberately to present their data and ideas in the most confusing way, with complicated equations and vast tables of statistics. The day is long past for such tactics; no one is impressed any more, and many are inclined to suspect that the analyst is trying to hide something. Moreover, such presentations are unreasonable impositions on the policymaker; they reflect a lack of consideration of his time and basic responsibility, which is to make decisions on the basis of advice stemming from the results of the analysis rather than to learn the intricacies of the models.

The importance of good communication cannot be exaggerated; it affects all aspects of model building and application, and it can help overcome every one of the obstacles hindering acceptance of models. The model builder must learn how to communicate successfully. This means learning to identify the appropriate audience for his message (i.e., policymakers for end results of model/policy analysis, and analysts or model specialists for structure and dynamics of the model itself). It also means learning to present written and oral information effectively, with more emphasis on simple statements and visual images of key results as the basic products of analysis.

These ten obstacles, outlined above, appear to us as the factors most responsible for the fact that policymakers do not rely on sophisticated models as essential tools of their trade. In *Table 15.1* we attempt to identify which of the groups involved in the policymaking process generates each of these obstacles. We hasten to add that this checklist is not an attempt to allocate blame, it is rather an attempt to pinpoint where modelers have particular problems to cope with. The list implies that modelers themselves do not give rise to many difficulties, but this is misleading. Most of these obstacles exist because modelers have not, hitherto, proven capable of dealing effectively with them.

A worrisome observation is that, while textbooks have been pointing out these pitfalls to modelers for many years, the situation does not seem to be improving. In the course of our survey, we reviewed a number of progress reports on modeling projects started since 1980. All of these emphasized how the modelers expected to be able to do a better technical job (of equation formulation, computing, etc.) than had been possible previously. But few dealt with innovations to overcome the difficulties noted here. Interactive computing facilities and scenario development workshops were mentioned as ways to improve the interface between model and clients, but rarely was it even acknowledged that, for example, multiple authorities with overlapping responsibilities might react differently than a single decision maker.

15.5. Directions for Improvement

The difficulty encountered hitherto in utilizing economic–ecological models is not alarming in view of the recent development of the discipline and the technology it uses. After all, we see the same mistakes being made by analysts from the parent disciplines after a hundred years of experience. In the previous section we offered some

Table 15.1. Source of impediments to the use of models in policymaking.

	Participants in the policymaking process			
Obstacle	*Modelers*	*Private interest groups*	*Bureaucrats*	*Political decision-makers*
Resistance to new technology		X	X	
Opposition to change		X	X	
Opposition to compromise		X		
Suspicion about objectivity		X		
Blurred decision-making authority				X
Commitments to established concepts			X	
Truncated perspective		X	X	
Preoccupation with distritutional effects				X
Apparent irrelevance	X			

suggestions for overcoming particular obstacles. Perhaps the most promising strategy, because it attacks so many of these problems at the same time, is to involve the potential users more deeply in the model building process; it will guarantee that at least the right questions are being asked, communication with users will improve, and the implications of assumptions in the model will be better understood.

In a world of imperfect communication and shifting clients, even the modeler as advisor is bound to omit some policy options and performance measures from his nominal analysis. The search for completely comprehensive models is hopeless. The alternative is to seek techniques for responding rapidly to new questions as they are raised; that is, to be able to build and modify models quickly and easily. This means learning to view models not as fixed logical structures, but rather as evolving constellations of hypotheses and relationships. Development of an adaptive viewpoint will also help to forestall "battles of models", by encouraging quick and constructive responses to criticism and experimentation with alternative assumptions rather than defense of initial ones.

The potential for future application of economic–ecological models is great, if for no other reason than the possibility of addressing a broader range of real decision-making concerns. The real challenge now is in learning how to embed the modeling process and its products more effectively in complex decision environments. This challenge will be best met by putting as much effort into the study of how models are received and used as has previously been placed in the study of how to build them.

ECONOMIC-ECOLOGICAL MODELING
L.C. Braat/W.F.J. van Lierop, (Editors)
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CHAPTER 16

Evaluation

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16.1. Introduction

The purposes of this book were, briefly, to picture problems in environmental and resource management and to describe mathematical models that can be used to assist in such management. We do not pretend to have given a complete state-of-the-art of economic–ecological modeling. Also no clear manual for how to build these types of models is provided with this book. Yet in the first part of this last and evaluating chapter an attempt is made to summarize some general suggestions and options to cope with that question of "how to build economic–ecological models?" We end this chapter with a short set of conclusions for each of the book's separate parts and with some recommendations for further research.

16.2. Some General Suggestions for Building Economic–Ecological Models

In this attempt to offer some general suggestions and options for the question "how to build economic–ecological models?" the same distinction between technical and institutional aspects has been followed as formulated earlier in this book. Obviously, each problem and each modeling effort has its unique circumstances. The given suggestions are therefore not more than reminders and will probably be more useful to the inexperienced modeler (and policy analyst) than to those who have been in the business of multidisciplinary, applied modeling already for a long time.

16.2.1. Technical aspects

Given a concise problem definition the following steps in model design may be distinguished:

(1) Assessment of the "location" of the problem and, in policy modeling, of the "policy objectives". If only an impact analysis in either one of the two systems is requested, then a monodisciplinary or "extended" monodisciplinary model should be sufficient. If, however, a dual impact or a sustainability analysis is requested then an integrated economic–ecological model is the appropriate choice.

(2) The purpose of the model in the problem solving process may be one or a combination of the following: description of the problem, prediction of consequences, search for the optimal solution, or evaluation of alternative solutions. For each of these parts in the problem-solving process a limited set of modeling techniques of models is really appropriate. Quantified, comprehensive, but structured static models are sufficient for descriptive purposes. Sophisticated analytical models are required for explanation of causes. They generally should be dynamic unless a single event can be identified as the cause. Simulation models of various forms are used for predictive analysis, and optimization techniques (linear, nonlinear, and dynamic programming) in the search for optimal solutions when quantified objectives and constraints are available. Cost-benefit, cost-effectiveness and various multi-criteria models are employed for assisting in evaluation.

(3) A choice has to be made between the "integration of submodels" approach and the "holistic" approach. In both cases, one should check whether the temporal and spatial characteristics of proposed model variables will match in the model equations. Data availability may differ. In that case not all the parameters may be equally reliably estimated. A mathematical format must be chosen, which best fits the nature of the problem and the intended use of the model (*see* (2)). Basic choices are whether to build a deterministic or a stochastic model, a linearized or nonlinear version, and whether to opt for an analytical or numerical solution.

16.2.2. Institutional aspects

From the foregoing chapters a condensed list has been compiled of suggestions to solve or remedy institutional problems in building economic–ecological models.

Within the academic world multidisciplinary modeling projects require some basic training of each of the participating modelers in each other's field. Seminars, workshops, and interactive building or "quick and dirty" models are well-known media to establish a common conceptual background. If a modeler or a team is modeling for policy applications, then:

(1) The starting point should be a "simple" model; preferably one to which more details can be added in later stages of the modeling process.
(2) The modeling process should have a clear, and limited, purpose.
(3) Uncertainty should not be hidden; it should be made explicit in alternative versions or scenarios.
(4) Rough model versions should be designed early in the project.
(5) Attention should be paid to articulate documentation.

For those policy analyists and policy advisers who are responsible for the quality and relevance, in other words the "effectiveness", of the modeling study, the following recommendations may be noted:

(1) Commitment to the project should be shown.
(2) Personal involvement in the model design seems absolutely necessary in order to be aware of (and influence) the model assumptions.
(3) They should try to learn some of the more useful jargon, concepts, and techniques.
(4) Articulate and realistic objectives for the study should be provided.

The recommended project structure presupposes participants who are committed and have paid attention to the above-listed suggestions. The major problem indicated seems to be lack of communication. The project structure therefore should stimulate and facilitate interaction and communication. Several formats are possible to assist in this, for example, interactive computer techniques for policymaking, computer graphics, the Adaptive Environmental Assessment and Management Workshop approach (Holling, 1978), various operational gaming techniques, etc.

16.3. Conclusions and Recommendations

This closing section summarizes very briefly the most important general conclusions from this book and formulates some key recommendations for future research.

The mainly historical and theoretical overview in Part I of the developments in modeling for environmental and resource management indicates that both the theories, the techniques, and the methods used so far are not yet fully multidisciplinary. This yields both for the holistic as for the compartment approaches described. Also most current existing approaches have not yet been widely accepted. A desire arises from Part I for "all-encompassing" approaches. Research into this direction seems highly recommendable. At the same time, however, the experiments with linking powerful and promising existing monodisciplinary models should continue, as well as the experiments with simple holistic approaches.

The modeling overview in the various policy fields from Part II shows by comparison that where it comes to application of models for environmental and resource management, also a tendency still exists to study the economic and ecological aspects separately. Maybe the above described sequence should, however, be completely turned around: economic–ecological models up to now have mostly evolved in an applied context.

That may explain the noticed relative technical simplicity of the models. Most practical models have an extended monodisciplinary character, whereas integrated economic–ecological attempts are usually fairly academic in nature. Where these attempts have reached most successes is in fields with rather "controlled" situations in which the engineering approach is very important (agriculture and water control).

Recommendations for further research in practical economic–ecological modeling in the various policy fields distinguished, relate to a translation of the above formulated general research recommendations for Part I to the level of each individual policy field.

As to the modeling process it can be concluded from Part III that the acceptance of models and the implementation of model results is still difficult. Increasing attention should therefore be paid to improvement of the communication between all people involved in the modeling process and to the form and presentation of models and their results. Better contacts with senior policy advisors could be of help in this respect, for instance, by means of workshops where they have to cooperate closely with the modelers in solving an actual problem. Better communication can be stimulated further by the use of multistage models, among others, in which a simple model version is used for presentation and to study and explain more general relations, whereas more detailed aspects are dealt with in specific submodels. Multidisciplinary research into this type of network models should also be stimulated.

APPENDIX A

The IvM–IIASA Survey

A.1. Introduction

In 1982 the Institute for Environmental Studies, Free University, Amsterdam, started a research project concerning the relevance of economic–ecological models for environmental policy. In August 1982, the International Institute for Applied Systems Analysis (IIASA) agreed to join IvM in this project.

The main aims of the project were defined as:

(1) An international survey of economic–ecological models.
(2) Evaluation of these models.

Within these aims a distinction was made between:

(1) Scientific aims.
(2) Policy aims.

The scientific purpose of the international survey was to make an inventory of:

(1) Types or classes of models in different problem fields.
(2) The kind of structure and specifications they have.
(3) The frequency distribution of different types.

The scientific evaluation purpose concentrated on:

(1) The levels of sophistication the models have reached.
(2) Comparison of the various models by field, in order to discover general and specific features.

(3) Problems.
(4) "Hot" research items.

Policy-related purposes of the survey were:

(1) Assessment of the actual (and potential) use of the models.
(2) To analyze who applied them.
(3) In which context.
(4) With what kind of policy objectives.

The policy evaluation purpose concentrated on the evaluation of the effectiveness of the applied economic–ecological models. The method chosen to acquire the available information on economic–ecological models and their applications has included a questionnaire survey, literature study, communication with modelers and policy advisers, and a Workshop on Economic–Ecological Modeling (December 12–14, 1983), at IIASA.

A generally accepted definition and classification of economic–ecological models was not available at the start of the project. We therefore used a *preliminary definition* with was given as: *a set of mathematical relationships describing any connections between economic and ecological systems* (Braat and van Lierop, 1982). This definition was communicated to all the participants in the project. Models in environmental economics and environmental biology were not excluded, because we could not tell in advance whether they contained anything that could be described and would be accepted as ecological and economic, respectively. (For more information about the definition problem *see* Section 4.1 in the main text of this book.)

A description and evaluation of a set of models can only be made accessible and intelligible with an effective classification system used to aggregate the individual models. We have therefore developed a simple classification system which we found effective in analyzing and evaluating the models. (This classification was introduced in Section 4.5.2. of the main text.)

A.2. Survey Response and Representation

During October and November 1982, approximately 200 questionnaires with a background paper were mailed to modelers thought to be involved in economic–ecological modeling. Additional questionnaires were sent out to people suggested by the initial respondents, the

National Member Organizations of IIASA, and other people who expressed interest, until March 15, 1983. This brought the total up to 350. Analysis of the response started after the final deadline of April 15, 1983. Additional information for the project was received in the form of detailed model descriptions in research reports and published papers, which had been requested in the questionnaire.

From the 354 scientists who received a questionnarie, 123 (almost 35%) have answered; 16 of them (5%) reported that they were no longer involved in economic–ecological modeling, 19 others (5%) showed interest in the project but did not answer the questionnaire for various reasons (for instance, because of being a theoretician in the field or because they felt that their model was not a truly integrated model). A total of 109 questionnaires were completed by 88 scientists (25%). Many people reported not only for themselves but represented a team; as a result 30 modelers (11%) are indirectly involved in the survey. Consequently, 36% of the scientists originally contacted are represented, while the total response is 46%. The nonresponse rate is 189 (53%), which includes those who never responded and those that responded after the deadline. The remaining 1% includes respondents from IIASA and IvM.

Unfortunately, some questionnaires had to be excluded. This was due to, among other reasons, representing theoretical model concepts only, or representing a monodisciplinary economic or ecological model. This brought the survey sample back to exactly 100. The result represented in this report are based on this number of questionnaires. However, even within these 100 questionnaires, several had to be excluded in the analysis of some of the questions. Consequently the total number of valid answers differs among questions. The extent to which these results are representative for the entire area of economic–ecological modeling is not clear.

The initial mailing list for the survey was derived from IIASA and IvM files. A second wave of questionnaires has been mailed early 1983 to people who were suggested by respondents of the first wave. In our opinion a fair representation of the area of economic–ecological modeling was obtained.

A.3. General Distribution of Answers

In the remaining part of this appendix the distribution and frequency of the answers per question of the questionnaire are shown. For more specific information, among others, resulting from analyses of combinations of questions (*see*: Braat and van Lierop, 1985).

A.3.1. Geographical distribution of the models

The geographical distribution of the survey sample is presented in *Table A.1*; 23 models came from Western Europe, 6 from Scandinavia, 15 from Eastern Europe, 40 from North America, 2 from South America, 9 from Australia, 3 from Japan, and 2 from Israel.

Table A.1. Country of origin of models included in the survey.

Country	*Number of models*	*Country*	*Number of models*
Argentina	1	Israel	2
Australia	9	Italy	1
Belgium	1	Japan	3
Brazil	1	The Netherlands	6
West Germany	6	Norway	2
Canada	10	Austria	2
Czechoslovakia	6	Finland	1
East Germany	1	Sweden	3
France	3	USA	30
Great Britain	4	USSR	3
Hungary	5		

21 countries in total participated with 100 models

A.3.2. Purpose of economic–ecological models

Models in general have the purpose of documenting and understanding systems of the real world, solving problems,and predicting consequences of human activities. This of course is also true for models in which both economic and ecological components, processes, and activities are represented. Three alternative purposes have been distinguished:

(1) *Analytical interest* – The model has been developed for academic purposes. It may of course have potential for application in a policy context.
(2) *Specific policy problems* – The model has been developed for small scale short-term policy problems.
(3) *General policy issues* – Here larger systems and long-term policy and planning are characteristic. The output will most likely be indications of trends, ranges in predictions, guidelines, and standards.

Question 2 dealt with these alternative purposes. The distribution of the answers is presented in *Table A.2*.

Table A.2. Purpose of economic–ecological models.

	Types of answers							Σ
(a) Application to a general policy issue	X			X	X		X	35
(b) Application to a specific (policy) case		X		X		X	X	46
(c) Analytical interest (only potential relevance for policy)			X		X	X	X	35
Distribution:	23	38	24	4	7	3	1	

Total valid cases: 100

A.3.3. Fields and extent of application

Fields of actual or potential application can be identified, whether the models are initially designed for academic or policy use. Twenty-five models from the survey were designed for a specific field, the 75 other models are more general and are used in various fields. The fields listed (*see Table A.3*) represent fields of planning and decision making in which economic and environmental issues have traditionally been dealt with. The list was not exhaustive, nor fully consistent as to level of detail. The option of defining additional fields of application appropriate to the modeling effort has been used 16 times.

Other fields mentioned were:

(1) Economic development and physical planning.
(2) Water pollution.
(3) Industry (especially food industry).
(4) Drinking water.
(5) Response to stress.
(6) Balance of payments.
(7) Transportation.
(8) Housing.
(9) Economic and environmental policy in general.
(10) Human ecology.

Within a field of application, models may for instance be used for identification and description, analysis of complex processes, and prediction of consequences of policies, control, or management. Since various models have multiple use capability in this respect, and because these distinctions are sometimes hard to make, these aspects have not been included in the survey. The questionnaire concentrated on the

Table A.3. Fields of application.

	Agriculture	*Forestry*	*Fisheries*	*Land use*	*Outdoor recreation*	*Energy*	*Nonrenew. resources*	*Nature conservation*	*Diseases*	*Pests*	*Water*	*Soil*	*Air*	*Other*	*Total*
1. *Agriculture*	2	8	8	30	11	11	10	17	3	11	29	21	7	7	50
2. *Forestry*		1	5	8	3	8	5	10	0	2	7	5	4	4	12
3. *Fisheries*			11	8	7	6	5	8	0	3	12	5	4	3	25
4. *Land use*				2	15	14	10	17	0	5	27	17	8	10	45
5. *Outdoor recreation*					0	3	3	12	0	1	15	4	4	6	19
6. *Energy*						2	13	9	1	1	8	7	10	5	25
7. *Nonrenewable resources*							0	8	1	2	6	6	6	4	18
8. *Nature conservation*								0	1	3	21	11	6	5	28
9. *Diseases*									0	2	1	2	0	0	4
10. *Pests*										0	6	6	2	2	11
11. *Water*											6	19	8	9	50
12. *Soil*												0	5	4	23
13. *Air*													0	4	14
14. *Other*															1

fields as such. *Table A.3* gives an overview of the frequency of application. The diagonal numbers represent the number of models built for one field of application only. For example, the first diagonal element, 2, indicates that only 2 models focus exclusively on agriculture, whereas a total of 48 focus on agriculture in combination with other fields. The 'row' total gives the number of models dealing with a specific field. For agriculture this number is 50. The various other elements of *Table A.3* indicate relationships between the fields of application. For instance, 17 models are applied (or applicable) both in land use and nature conservation. Many of these 17 models may include more fields of application. Several combinations between fields of application are quite obvious and consequently occur quite regularly. For example, agriculture with land use, fisheries with water, etc. This might imply that the number of models designed for specific fields of application is higher than 25.

Some models have been designed for many fields of application. Consequently the sum of models in a row in *Table A.3* will usually differ from the row total for a field given in the last column. None of the models applied in or applicable for outdoor recreation, nonrenewable resources, nature, conservation, diseases, pests, soil, and water were built exclusively for these fields of application.

In general, models are developed from some *conceptual* framework, often described in the form of a set of boxes and arrows (diagrams), which have no strict definitions or constraints.

These diagrams, sometimes also called conceptual models, often form the basis for the next stage of model development, in which system components, processes, and relationships are described in a mathematical format. The resulting structure is less ambiguous. These models in mathematical format, called *theoretical* by some modelers, are *operational* in that with addition of fictional or real world data, some form of quantitative analysis can be made. When these operational models are subsequently used, they may be called *applied* models. Two categories have been distinguished in the questionnaire in relation to the purpose of the model: models which have been *applied in a research context* (e.g., methodological) only, and those *applied in actual policy formulation or decision making*.

Question 3.2 dealt with the extent of application. The distribution of answers is represented in *Table A.4*. From this table we can see that models applied in an actual policy context and models applied in a research context are equally high in representation in the sample (both 36 times). The combination a–c, in which only one score is made, is probably a mistake. It should be mentioned that the total number of questionnaires represented in *Table A.4* is only 91. This is because of incomplete answers.

Table A.4. Extent of application.

	Types of answers							Σ
(a) Application in an actual policy context	X			X		X	X	37
(b) Applied in a research context		X		X	X		X	50
(c) Not yet applied but operational			X		X	X	X	24
Distribution:	22	31	18	14	5	1	0	
Total valid cases: 91								

A.3.4. Model testing

The degree to which a model or its output represents the structure or behavior of the system it was meant to represent can be evaluated in various ways.

The relative performance of a model can be tested by comparing its results with the results of other models calibrated with the same data input. Statistical and econometrical testing techniques can be of help in this respect. A model can also be evaluated by comparing calculated (predicted) values with values measured in the real world. Of course, the measured values that have been used for calibration cannot be used as valid test data. Statistical methods (tests) may be used in deciding the significance of the difference between predicted values and measured values.

Dynamic simulation models (which occurred often in the survey) can be regarded as tested when repeated success in prediction is observed. This may be done by starting the simulation at some point in history with adequate historical initial conditions and subsequent assessment of the deviation of the present values, or by monitoring the real world systems for continuous testing.

Table A.5. The reported testing of the survey models.

	Types of answers								$\sum$
(a) Tested by comparison of performance, with other models in relation to the same set of historical data	X				X			X	17
(b) Tested against data, other than used for calibration of the world		X			X	X	X	X	54
(c) Tested by repeated success in prediction			X			X		X	16
(d) Not yet tested				X			X		32
Distribution:	5	32	3	30	7	8	2	5	

Total valid cases: 92

Question 4 dealt with the issue if and how the models have been tested. *Table A.5* gives the distribution of the answers. Apparently testing against the data, other than used for calibration of the model, is the most common way of testing economic–ecological models. Combination b–d scores twice, most likely by mistake or misunderstanding.

A.3.5. Types of economic–ecological models

Economic–ecological models are considered to consist of at least one economic and one ecological submodel. It is, however, also possible to have several economic submodels connected to one or several ecological ones. The internal *structure* of the submodels can be defined by the form of the internal relationships between the variables. Only two types have been distinguished:

(1) A submodel consists of separate, isolated, variables only ("s", simple submodel).
(2) A submodel contains a set of variables which are fully, or partially, interrelated ("c", complex submodel).

Economic–ecological models which have only one elaborately developed submodel linked to a single index (or set of independent indexes) that represent the other system, or driven by one, or several, exogenous, independent variables from the other system can be considered as a group in which these two types are mixed.

Among the 81 cases there are:

16	simple economic submodels.
9	simple ecological submodels.
65	complex economic submodels.
72	complex ecological submodels.

and the following combinations:

simple economic	+	simple ecological submodel:	4
simple economic	+	complex ecological submodel:	12
complex economic	+	simple ecological submodel:	5
complex economic	+	complex ecological submodel:	60

Other than the relative complexity of their internal structure of the submodels, economic–ecological models can be classified further by the types of *relationships* between the submodels. Three types are distinguished, based on the direction of the relationships:

(1) A one-way relationship in which the economic submodel drives the ecological submodel (10 models).
(2) A one-way relationship in which the ecological submodel drives the economic submodel (19 models).
(3) A two-way relationship, i.e., interdependent submodels (52 models).

In *Table A.6* the two classifications have been combined to produce 12 types of economic–ecological models.

Table A.6. Types of economic–ecological models.

	Econ. *Ecol.*	*Econ.* *Ecol.*	*Econ.* *Ecol.*	*Total*
Simple models	S → S 1	S ⇄ S 2	S ← S 1	4
Simple economic–complex ecological models	S → C 2	S ⇄ C 3	S ← C 7	12
Complex economic–simple ecological models	C → S 2	C ⇄ S 0	C ← S 3	5
Complex models	C → C 5	C ⇄ C 47	C ← C 8	60
Total	10	52	19	81

A.3.6. Model characteristics

Models can be described by many characteristics, their time and space dimensions, their size, and their functioning. As for *time*, the first distinction made is whether a model has time as a variable. If so, the model is called dynamic. Dynamic explicitly refers to temporal dynamics. If time is not a variable, models are called static. One class of models which does consider time, but not as a variable, is separately indicated: comparative static models. These models deal only with time in as far as they take into account the beginning and the end of the period for which they have been developed. *Table A.7* presents the distribution of answers on the time dimension of models; "En" stands for economic submodel, "El" stands for ecological submodel. Dynamic models dominate the field; both completely dynamic models and models with a dynamic ecological submodel linked to a static or comparative static economic submodel are numerous. Another way of looking at the answers is presented in *Table A.8*.

Four geographical *scales* have been distinguished in the survey: local, regional, national, and global. Global and national scales were considered to present no problems in delineation. Regional models can range from very large to rather small areas. However, it was explained in the background paper, which accompanied the questionnaire, that they should cover only part of a nation and include more than just a city or an ecosystem (the latter considered to be the local scale). *Table A.9* gives the distribution of the various geographical scales in the models.

Table A.7. Dynamics of economic–ecological models.

	En/El		*En/El*		*En/El*		*En/El*	
(a) Static	X	X						
(b) Comparative static			X	X				
(c) Dynamic					X	X		X
Distribution:	12		6		48		12	
(a) Static				X				
(b) Comparative static	X					X	X	X
(c) Dynamic		X	X		X			X
Distribution:	6		1		3		1	
Total valid cases: 89								

Table A.8. Total number of economic and ecological submodels from various time categories.

	Economic submodels	*Ecological submodels*
(a) Static	24	13
(b) Comparative static	13	10
(c) Dynamic	52	67
Total valid cases:	89	90[a]

[a]One double-count, due to the combination represented by the extreme right column in *Table A.7*.

Table A.9. The geographical scale of economic–ecological models.

	En/El		*En/El*		*En/El*		*En/El*		*En/El*		*En/El*		*En/El*	
(a) Local	X	X							X	X	X	X		
(b) Regional			X	X					X	X	X	X		X
(c) National					X	X					X	X	X	
(b) Global							X	X						
Distribution:	22		28		8		6		6		2		3	
(a) Local	X	X				X	X	X		X		X		
(b) Regional	X		X	X	X	X		X	X		X	X		
(c) National			X					X			X			
Distribution:	2		2		3		1		2		1			
Total valid cases: 86														

An alternative way of looking at time in models, different from the approach followed in *Tables A.7* and *A.8*, is from the point of view of

Table A.10. Analyzed time periods, time intervals, and prediction horizons of economic and ecological submodels.

	Covered time period		*Time interval*		*Horizon of prediction*	
	En	*El*	*En*	*El*	*En*	*El*
1 day	1	2	4	16	1	1
1 wk	1	1	1	4	–	–
2 wks	–	–	1	1	–	–
6 wks	–	–	1	–	–	–
1 mth	–	1	5	5	–	1
2 mths	–	–	–	–	–	1
3 mths	–	–	3	5	2	3
5 mths	–	1	–	–	–	–
6 mths	1	2	–	–	1	2
1 yr	17	17	37	29	5	6
2 yrs	2	1	–	–	–	–
3 yrs	3	3	–	–	3	2
4 yrs	–	1	–	–	–	1
5 yrs	2	4	6	6	4	2
6 yrs	–	–	–	–	–	1
7 yrs	–	–	–	–	–	1
8 yrs	–	–	–	–	1	2
9 yrs	1	1	–	–	1	–
10 yrs	8	5	–	–	4	3
15 yrs	2	1	–	–	1	1
20 yrs	3	3	–	–	7	8
25 yrs	3	3	–	–	3	3
30 yrs	4	4	–	–	5	6
40 yrs	–	–	–	–	2	3
50 yrs	7	9	–	–	1	2
100 yrs	5	5	–	–	2	2
200 yrs	–	–	–	–	1	1
250 yrs	–	–	–	–	1	–
Indefinite	2	3	2	2	6	7
Total valid cases:	62	67	60	70	50	58

time periods covered by the model, either in analysis or in prediction (time horizon). Additional features then are the time intervals. Since time is often treated differently in economic and ecological models a distinction was indicated in the questionnaire. Regrettably the survey did not supply unambiguous information on this point. This should be taken into account in interpreting the results which are presented in *Table A.10*. Because of the problems with this question we give the scores for each time aspect for the economic and ecological submodels separately. Combinations are not taken into consideration here. A similarity in economic and ecological submodels is evident in using a "1"-

year period in their analysis as well as for their time interval. Economic submodels seem to have slightly longer time intervals than ecological ones. The horizon of prediction varies between 20 and 30 years. It looks as if many models from the survey aim to give a forecast for the year 2000.

A distinction between optimization models (which contain an objective function), simulation models, and other models which have no internalized objectives was the basis for *Table A.11*, which presents the distribution of the combinations. The other models which have been mentioned in the questionnaire are:

(1) Input–output models.
(2) Scenario.
(3) Analytical.
(4) Statistical functions.
(5) Decision models.

Table A.11. Dynamics of economic–ecological models.

		En/El		*En/El*		*En/El*		*En/El*		*En/El*	
(a)	Optimization	X	X			X		X	X	X	
(b)	Simulation			X	X		X	X	X	X	X
	Distribution:	23		25		12		10		3	

		En/El		*En/El*		*Total El*	*Total En*
(a)	Optimization	X	X		X	44	24
(b)	Simulation		X	X	X	27	47
	Distribution:	6		1		71[a]	71[a]

Total valid cases: 80

[a]Owing to double-counting.

Because they scored only a few times, they have not been included in *Table A.11*. It should be remarked that the models in this listing are not absolutely exclusive.

Economic submodels seem to use optimization techniques more, whereas ecological submodels use simulation techniques to a greater extent. Combined in economic–ecological models they appear to be used relatively as frequent. The combination of an economic simulation submodel with a single ecological optimization model, however, does not exist.

One way to indicate the *size* of a model, relevant in both ecological and economic models, is the number of endogenous (state) variables, as presented in *Table A.12*. Small and medium-sized models appear to exist

Table A.12. Number of endogenous variables in economic–ecological models.

		En/El		*En/El*		*En/El*		*En/El*		*En/El*	
(a)	1–10	X	X					X			
(b)	10–100			X	X				X	X	
(c)	> 100					X	X				X
	Distribution:	34		19		4		17			
(a)	1–10		X				X	X			
(b)	10–100	X			X						
(c)	> 100			X		X			X		
	Distribution:	5		5		1		2			
Total valid cases: 89											

more than larger models. However, one should note that the number of variables per submodel is indicated. Two submodels of type b may imply close to 200 variables!

APPENDIX B

References for Survey Models

Code

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