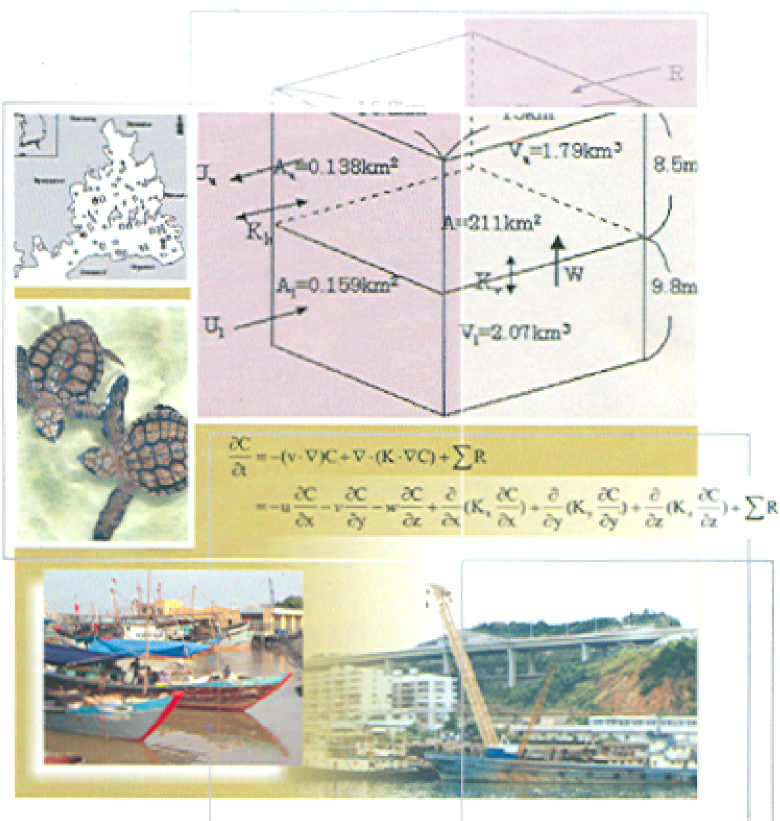
Determining Environmental Carrying Capacity of Coastal and Marine Areas: Progress, Constraints, and Future Options

Edited by

Huming Yu and Nancy Bermas



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A Workshop Proceedings

Kuala Lumpur, Malaysia • 12-15 May 2002

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Organized by:

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National Oceanography Directorate, Ministry of Science, Technology and the Environment, Malaysia
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MISSION STATEMENT

The Global Environment Facility/United Nations Development Programme/International Maritime Organization Regional Programme on Building Partnerships in Environmental Management for the Seas of East Asia (PEMSEA) aims to promote a shared vision for the Seas of East Asia:

> "The resource systems of the Seas of East Asia are a natural heritage, safeguarding sustainable and healthy food supplies, livelihood, properties and investments, and social, cultural and ecological values for the people of the region, while contributing to economic prosperity and global markets through safe and efficient maritime trade, thereby promoting a peaceful and harmonious co-existence for present and future generations."

PEMSEA focuses on building intergovernmental, interagency and intersectoral partnerships to strengthen environmental management capabilities at the local, national and regional levels, and develop the collective capacity to implement appropriate strategies and environmental action programs on self-reliant basis. Specifically, PEMSEA will carry out the following:

- build national and regional capacity to implement integrated coastal management programs;
- promote multi-country initiatives in addressing priority transboundary environment issues in sub-regional sea areas and pollution hotspots;
- reinforce and establish a range of functional networks to support environmental management;
- identify environmental investment and financing opportunities and promote mechanisms, such as public-private partnerships, environmental projects for financing and other forms of developmental assistance;
- advance scientific and technical inputs to support decision-making;
- develop integrated information management systems linking selected sites into a regional network for data sharing and technical support;
- establish the enabling environment to reinforce delivery capabilities and advance the concerns of non-government and community-based organizations, environmental journalists, religious groups and other stakeholders;
- strengthen national capacities for developing integrated coastal and marine policies as part of state policies for sustainable socio-economic development; and
- promote regional commitment for implementing international conventions, and strengthening regional and sub-regional cooperation and collaboration using a sustainable regional mechanism.

The twelve participating countries are: Brunei Darussalam, Cambodia, Democratic People's Republic of Korea, Indonesia, Japan, Malaysia, People's Republic of China, Philippines, Republic of Korea, Singapore, Thailand and Vietnam. The collective efforts of these countries in implementing the strategies and activities will result in effective policy and management interventions, and in cumulative global environmental benefits, thereby contributing towards the achievement of the ultimate goal of protecting and sustaining the life support systems in the coastal and international waters over the long term.

Dr. Chua Thia-Eng
Regional Programme Director
PEMSEA

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LIST OF ABBREVIATIONS AND ACRONYMS

BOD	Biological oxygen demand
CEMS	Coastal environment management system
CHL	Chlorophyll concentration
COBSEA	Coordinating Body on the Seas of East Asia
COD	Chemical oxygen demand
DEFRA	Department of Environment, Food and Rural Affairs
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DOF	Department of Fisheries
DON	Dissolved organic nitrogen
DOP	Dissolved organic phosphorus
EC	Environmental capacity
ECC	Environmental carrying capacity or effective/permissible carrying capacity in tourism
ECOS	Ecotourism opportunity spectrum
EFDC	Environmental fluid dynamics code
EIA	Environmental impact assessment
EU	European Union
FCZ	Fish culture zone
GIS	Geographic information system
HEM	Hydrodynamic eutrophication model
ICM	Integrated coastal management
IMO	International Maritime Organization
INCO	International Cooperation
IOC	Intergovernmental Oceanographic Commission
LAC	Limits of acceptable change
LPOC	Labile particulate organic carbon
LPON	Labile particulate organic nitrogen
LPOP	Labile particulate organic phosphorus
MC	Management capacity
MOE	Ministry of Environment
MOMAF	Ministry of Maritime Affairs and Fisheries
MOSTE	Ministry of Science, Technology and the Environment
MSY	Maximum sustainable yield
NH4	Ammonium nitrogen
NO3	Nitrate + nitrite nitrogen
NPS	Non-point source
PCC	Physical carrying capacity
PEMSEA	Partnerships in Environmental Management for the Seas of East Asia

PO4	Dissolved phosphate
PO4t	Total phosphate
POM	Particulate organic matter
PS	Point source
RCC	Real carrying capacity
ROS	Recreation opportunity spectrum
RPOC	Refractory particulate organic carbon
RPON	Refractory particulate organic nitrogen
RPOP	Refractory particulate organic phosphorus
S	Salinity
SA	Available silica in modeling
SA	Sustainability appraisal
SDS-SEA	Sustainable Development Strategy for the Seas of East Asia
SEA	Strategic environmental assessment
SEERA	South East of England Regional Assembly
SFG	Scope for growth
SMAs	Special management areas
STP	Sewage treatment plant
SU	Particulate biogenic silica
T	Temperature
TAM	Total active metal
TCC	Tourism carrying capacity
TMDL	Total maximum daily load
TN	Total nitrogen
TOC	Total organic carbon
TOMM	Tourism optimization management model
TOS	Tourism opportunity spectrum
TP	Total phosphorus or total phosphate
TPLC	Total pollution load control
TPLMS	Total pollution load management system
TPM	Total particulate matter
TSS	Total suspended solids
UNEP	United Nations Environment Programme
UNESCO	United Nations Education, Scientific and Cultural Organization
UWE	University of West of England
VAMP	Visitor activity management process
VERP	Visitor experience resource protection
VIM	Visitor impact management
VIMP	Visitor impact management process

PREFACE

Measuring environmental carrying capacity (ECC) is a key scientific issue for implementing sustainable development. ECC is a widely used concept particularly in the study of human populations. In the past, the concept was focused on the relationship between humans and their subsistence. This concept was easily extended to cover a more general ecological framework when population pressure and its effect on the environment has become an issue. This concept has generated multiple definitions of carrying capacity including a variety of strategies for applying it. Estimates and models of carrying capacity have therefore flourished. Finding convincing and widely agreed upon estimates and models for human populations and their activities, however, is difficult.

In the coastal and marine environment, carrying capacity analysis is met with difficulty which is attributed to the complexity of multiple and interactive subsystems that make up the marine ecosystem, coupled with their dynamic nature both in space and time. Despite the uncertainties, there are a number of studies done in the East Asian region on carrying capacity particularly on aquaculture development. Available information, however, are scattered. There is therefore a need to synthesize and analyze available information. The synthesis would hopefully point to a common measure on addressing carrying capacity in the marine and coastal areas.

ECC is one of the major environmental management difficulties or "bottlenecks" affecting policy and management decisions in the East Asian seas region. As an initial step towards this direction, a case study on ECC is being undertaken by the GEF/UNDP/IMO Regional Programme on Building Partnerships in Environmental Management for the Seas of East Asia (PEMSEA). The project aims to gather existing knowledge on the concept, approaches, and methodologies in measuring carrying capacity of bays, lagoons or semi-enclosed seas relating to sectoral development such as tourism, industries, fisheries and aquaculture.

In May 2001, PEMSEA launched an electronic forum on carrying capacity to bring together experts on carrying capacity and obtain their consensus, by sharing experiences and expertise on the issue. The forum generated varied opinions and recommendations from the experts. The present workshop was convened as part of the effort to consolidate the outputs of the forum and promote data sharing among the members of the group. Seventy percent of the workshop participants were members of the e-forum. The workshop was held in conjunction with the Asia-Pacific Conference on Marine Science and Technology in coordination with the National Oceanography Directorate of the Ministry of Science, Technology and the Environment (MOSTE), Malaysia, the Malaysian Society of Marine Sciences and the Institute of Biological Sciences, University of Malaya. The workshop specifically aims to 1) stimulate greater interest in carrying capacity studies by bringing together experts from East Asia and other regions to exchange knowledge on the concept, approaches and methodologies and application of carrying capacity in coastal and marine areas; 2) to identify gaps and constraints in measuring environmental carrying capacity; 3) to synthesize available information and propose common measures on addressing ECC; and 4) to recommend actions on how research results can be linked to policy and management decisions.

This volume presents the summary of the proceedings of the workshop and the key technical/scientific papers presented during the two-day event. It discusses the concept and approaches, methodologies, application and case studies, research needs and priorities in ECC research. The recommendations focus on the roles of the policymakers, scientists, and resource managers and provide useful insights and future opportunities in advancing ECC research and application. It is hoped that through this effort, better cooperation in terms of sharing knowledge and research outputs among the countries in the East Asian region and with other parts of the world would be promoted.

Chua Thia-Eng
Regional Programme Director

ACKNOWLEDGEMENTS

The Workshop on Environmental Carrying Capacity owes it success to the collective efforts of various individuals, agencies, and organizations. Their support and valuable contributions are gratefully acknowledged.

Sincere appreciation is extended to the organizers of the Asia-Pacific Conference on Marine Science and Technology: the Malaysian Society of Marine Sciences; National Oceanography Directorate; Ministry of Science, Technology and the Environment Malaysia; and the Institute of Biological Sciences, University of Malaya. Special thanks are given to Prof. Phang Siew Moi and Prof. Ho Sinn Chye for their efficient coordination and assistance before and during the formulation and implementation of the workshop. Sincere thanks also go to Prof. Chou Loke Ming for efficiently facilitating the discussion during one of the sessions. The active participation and enthusiasm of the ECC group are also well appreciated. The discussions certainly facilitated a flow of ideas and suggestions, which constitute valuable contribution to this effort. Without their inputs, this document would not have been completed. We owe the high quality of the papers included in this publication to Dr. Lewis Linker of the Chesapeake Bay Program of the US Environmental Protection Agency; Drs. Chou Loke Ming, Ruth O'Riordan, and Navjot Sodhi of the Department of Biological Sciences, National University of Singapore; and to selected ECC participants for peer-reviewing the papers.

Finally, the tireless efforts of the staff of PEMSEA who contributed in a variety of ways to the success of this undertaking are deeply appreciated.

PART I

SUMMARY REPORT OF THE WORKSHOP

Introduction

The ECC workshop was held in Hotel Istana, Kuala Lumpur, Malaysia from 12 May to 15 May 2002. Thirteen experts from PR China, Hong Kong, Japan, Philippines, Portugal, Republic of Korea, Singapore, Thailand, and the United Kingdom, and two from PEMSEA's Regional Programme Office participated in the workshop. The list of participants is given in Annex I.

Dr. Huming Yu, PEMSEA Senior Programme Officer, served as Chairperson during the workshop. He welcomed the participants on behalf of PEMSEA and Dr. Chua Thia-Eng, Regional Programme Director. Dr. Yu provided a brief background of PEMSEA focusing particularly on how inputs from ECC studies would improve the management approach of the Programme. Dr. Yu mentioned that there are several regional initiatives dealing with the East Asian Seas but these are largely pursued on sectoral basis. For example, UNEP-COBSEA is primarily dealing with land-based sources of pollution; IMO is addressing shipping-related issues and ship sources of pollution, while UNESCO-IOC is focusing on improved understanding of oceanographic and coastal processes. As PEMSEA is management oriented, a niche for the Programme is seen where it provides a framework that cuts accross these major sources of environmental impacts by applying available scientific and technological knowledge for improved management. In this respect, the ECC will be a very important input in improving the governance effort that PEMSEA is advocating.

Dr. Yu also discussed the Sustainable Development Strategy for the Seas of East Asia (SDS-SEA), which serves as a framework that will guide the PEMSEA participating countries in developing their national coastal and marine policies. The inputs of the ECC workshop will be very helpful to further clarify and refine common areas, particularly the scientific agenda set for the Strategy, which is an important message to communicate to policymakers.

Dr. Yu proceeded with the introduction by expounding on the objectives and expected outputs of the ECC workshop. He emphasized that the discussion will not follow a firm agenda; hence free exchange of ideas was encouraged. As ECC of coastal and marine environment can be considered a pioneering effort in the region, he encouraged the participants to exchange perspectives and experiences on carrying capacity from their own disciplinary backgrounds. It was hoped that through this knowledge exchange, relevant advice and outputs could be provided to:

- Improve the development and implementation of PEMSEA activities;
- Generate collective ideas for post-workshop activities to address ECC, such as packaging of project proposals for funding; and
- Promote effective networking.

The following questions were raised to facilitate the discussions:

- What is ECC from your perspective?
- What are the uncertainties in addressing ECC?
- What are the research needs and approaches or any suggested methodologies to address these uncertainties?
- What are the opportunities and mode of operation for some collective research on ECC?

1.0. Concept and Approaches

1.1. Concept

With the different interpretations on carrying capacity, the participants saw the need to define carrying capacity. Dr. Barbara Carroll of ENFUSION, UK, shared the work on a joint project between the Environment Agency in the UK and the Southeast England Regional Assembly for the past three years. The project examined how ECC can be linked to a land use-planning system and how results of scientific studies can be translated and made useful in decision-making. Through workshops with planning authorities, government organizations, and scientists working for environmental regulation, it was decided that carrying capacity assessment is too complex and absolute as a concept, hence, the name of the methodology was changed to threshold assessment, which is more applicable to spatial planning.

With respect to bivalve culture, Dr. Pedro Duarte of the Fernando Pessoa University, Portugal, contributed several definitions coined earlier by other authors:

- The maximum standing stock that may be kept within a particular ecosystem to maximize production without negatively affecting growth rate; and
- The standing stock at which the annual production of the marketable cohort is maximized or the total bivalve biomass supported by a given ecosystem as a function of the water residence time, primary production time, and bivalve clearance time.

These definitions are focused on target species, despite a growing tendency in Eastern countries to focus on "ecological aquaculture", which is based on multispecies culture where producers and consumers are grown together to facilitate nutrient recycling. The objective is to maximize production and optimize species combinations and distributions to reduce the environmental impacts of aquaculture. The growing appreciation that ecosystems have multiple functions, with a need for sustainable management means that ecologists are increasingly challenged to model the many interactions between and among species, including their environment, on a large-scale. Therefore, a general definition of carrying capacity at the ecosystem level could be the amount of change that a process or variable may suffer within a particular ecosystem, without driving its structure and function over certain acceptable limits.

Dr. Wong Poh Poh, Department of Geography, National University of Singapore, focused his discussion on tourism carrying capacity (TCC). He mentioned that as far as tourism is concerned, ECC is seldom used. An X number of tourists in a given place is what TCC is trying to determine. It may sound very simple because the focus is on the numbers. However, one asks if these numbers are occurring in one place at one point in time. A certain number of people can be spread over an area or concentrated in one place. If the whole gamut of studies on TCC is examined, it is apparent that there is no common definition which may range from the simplest to the most complex. Most definitions are usually grouped into various types or combinations of various types, e.g., environmental/physical/ecological, economic, social, socio-economic, cultural, and perceptual/psychological.

Dr. Chou Loke Ming, Department of Biological Sciences, National University of Singapore, considered the two major components of the marine environment - the water and the ecosystems. For water, there are threshold values, which can help in defining carrying capacity and most of these are related to human health. For ecosystems, it is very difficult to come up with some threshold limits similar to water quality. To determine the quantity of degradation it can take, one can look at the mangrove forest since the area removed can be quantified, and then relate it to human perception and the levels of goods and services provided. It is not a scientific way of assessment but can be a good way to start to prevent complete degradation.

Dr. Paul Shin, Department of Biology and Chemistry, City University of Hong Kong provided an interesting mix of definitions of carrying capacity given by his students ranging from its similarity to assimilative capacity, control of pollution loadings and a means of maintaining maximum sustainable yield (MSY). MSY has been a term often used in fisheries management but its linkage with carrying capacity may be worth looking at.

Dr. Ken Furuya, Department of Aquatic Bioscience, The University of Tokyo, mentioned that ECC can be defined in various ways because coastal waters have very diverse functions. Understanding the material cycling in nature and ecosystem components serves as important background in defining carrying capacity. Dr. Furuya's definition of ECC is quite simple, which is the maximum standing stock or production with least impact on the environment.

Finally, Dr. Park Kyeong, Department of Oceanography, Inha University, Republic of Korea, mentioned that ECC can be derived based on a good understanding of prototype processes. No matter what issues are being addressed, there are fundamental prototype processes such as water movement, chronic pollution, and physical transport, which are common concerns of most countries in the region. Whether one is looking at aquaculture or tourism, there is a need to develop common ways of quantifying the basic prototype processes.

1.2. Models

Models are very specific in terms of area and in terms of the system that these simulate. Thus, all models must be customized for a particular site. Although it is recognized that there are margins of error in modeling because of the conservative approach to quantifying variables in addition to the constraints imposed by limited knowledge in ecosystem functioning, the usefulness of models in ECC studies was highlighted, particularly for integrating spatial and temporal variability of an ecosystem. An ideal water quality model package for estuarine and coastal waters, contributed by Dr. Park of RO Korea, is shown in Figure 1.

1.3. Developing common guidelines in quantifying ECC

It was emphasized that it would be possible and advantageous to develop common ways of quantifying ECC, e.g., common protocols or generic guidelines, for specific types of resource uses, e.g., tourism, aquaculture, pollution loadings, and eutrophication in a body of water. These would assist the application and extension of the efforts in ECC measurements, particularly in developing countries. This is also seen as a worthwhile endeavor for PEMSEA or for other programmes to consider. The basic protocol should be broad enough such that it satisfies, the basic requirement of comparing water quality between a Thai and a Philippine resort. Additional processes or properties, depending on the specific need of the countries or sites, will supplement the basic protocol.

The workshop noted that there are two approaches to determine ECC. One is referred to as the active approach, e.g., modeling or other systematic research, to measure the thresholds. Second is the passive or adaptive approach, whereby some rough thresholds are presumed and subsequently readjusted according to the feedback of management interventions undertaken in response to the presumed thresholds. For example, the latter approach was applied in determining the ECC for coral reef ecosystems under the pressure of coastal tourism and other activities offshore of Indonesia. In this case, 25 percent of the no-take area for coral reefs was assumed as the level of ECC, based on past research and experience. Any figures going beyond the level of ECC would indicate the level of impact exceeding the ECC. The 25 percent threshold is subject to change if it is proven to be an under- or over-estimate. In the real world, the two approaches are applied simultaneously.

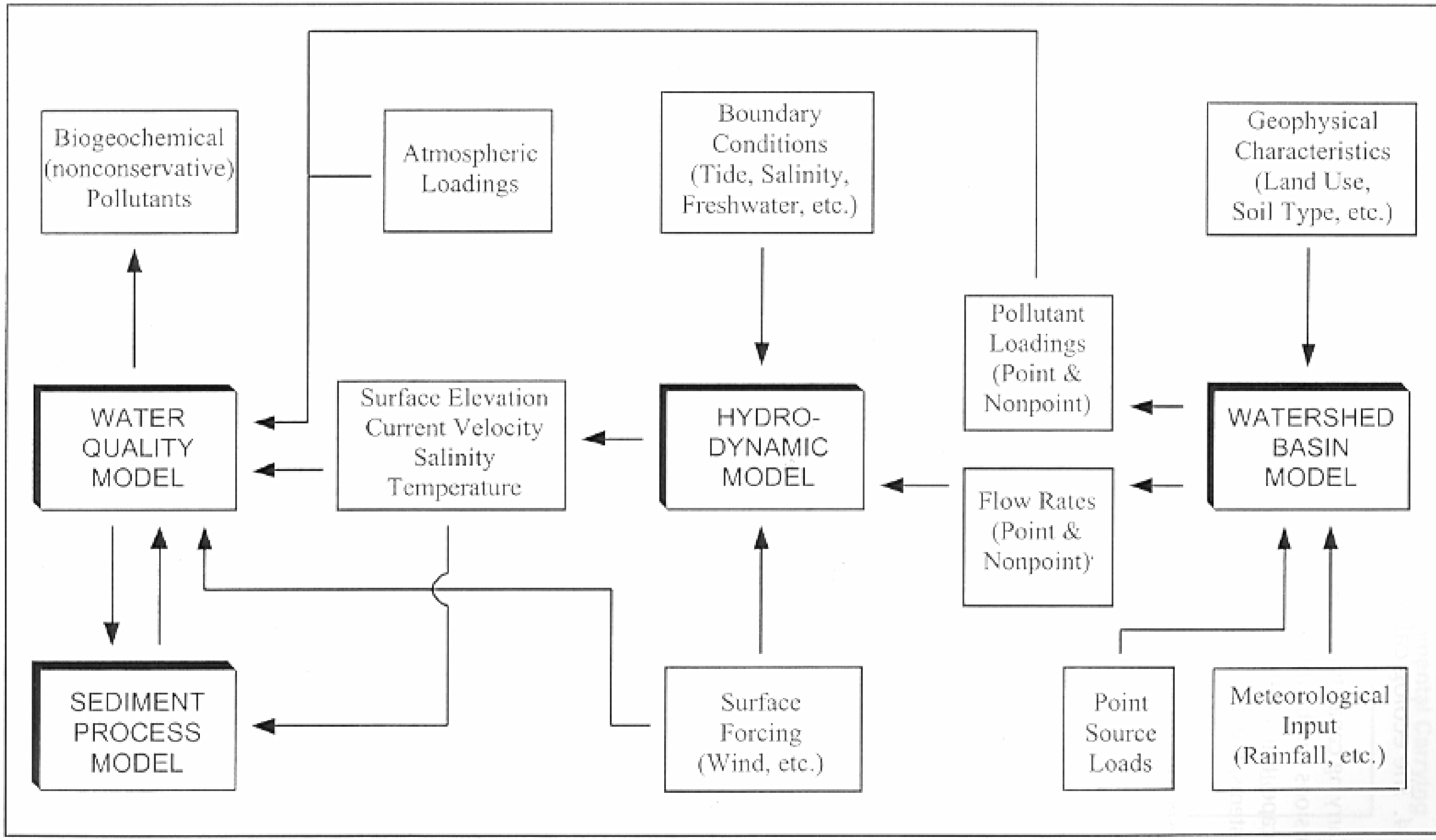


Figure 1. An Ideal Water Quality Model Package for Estuarine and Coastal Waters.

1.4. The ecological wheel

Carrying capacity was viewed as a multi-dimensional concept. The number of dimensions can be different among ecosystems. In an ecosystem with a beautiful landscape, the aesthetic value dimension is very important for tourism. In other ecosystems, the landscape may not be so important compared with production.

Based on the above premise, an ecological wheel was developed where the axes stand for the different sectors or dimensions of the ecosystem and the thresholds or acceptable level of impacts are indicated for each sector/dimension (Fig. 2).

It was suggested that the core of the wheel could conceptually represent sustainable development. Values can be assigned to each of the axes that separate the different sectors/dimensions. The values can be in the form of a formula, energy, or dollars and cents. A relative scale or range in each of the axes to indicate the thresholds can be included.

The synergies of the different sectors/dimensions, however, should be considered. If the thresholds are exceeded per sector, the synergistic effects will have a more serious impact.

An alternative way of presenting the concept of the ecological wheel was given where ecosystem attributes, maximum yields, and levels of acceptable change with corresponding thresholds serve as axes of the wheel. The axes can be extended since they represent an aggregation of maximum yields from several aspects. When the three thresholds are linked, the carrying capacity is reflected. More axes can be added to identify the acceptable thresholds (Fig. 3).

1.5. Relevance of ECC to national economic policy

The workshop participants believed that the ECC concept and its application are of paramount importance for national development policy and economy. Some participants pointed out that the ECC was not considered in traditional national economic policy, which could be characterized by the formula:

$$Y = C + I + G + X - M$$

where, Y = national income; C = consumption; I = investment; G = government expenditure; X = export; and M = import. To stimulate the economy under traditional economic policy, efforts would be made to augment one of these terms, e.g., consumption, investment, government expenditure and export, without considering ECC. However, certain economic stimulus package fails to achieve its

Figure 2. Multi-dimensional Concept of Environmental Carrying Capacity.

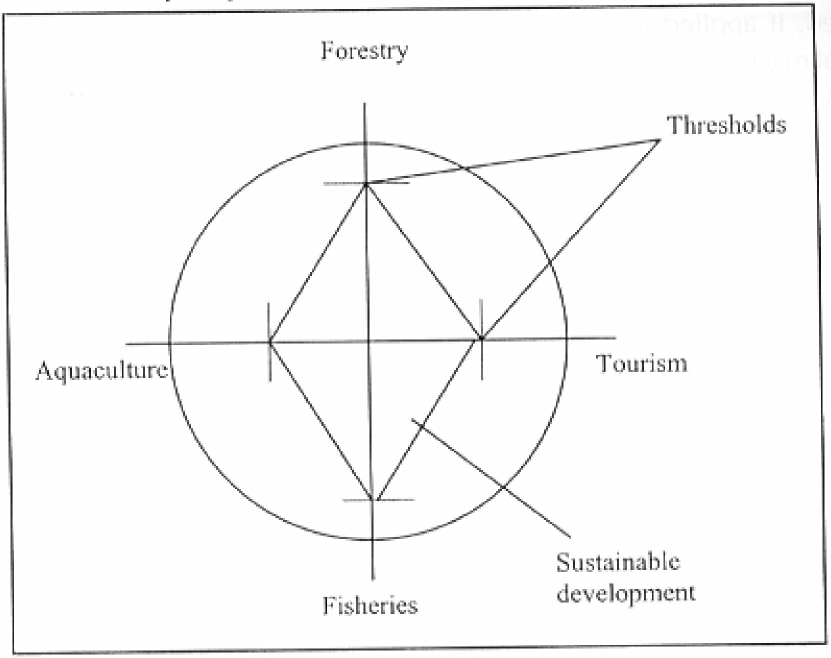
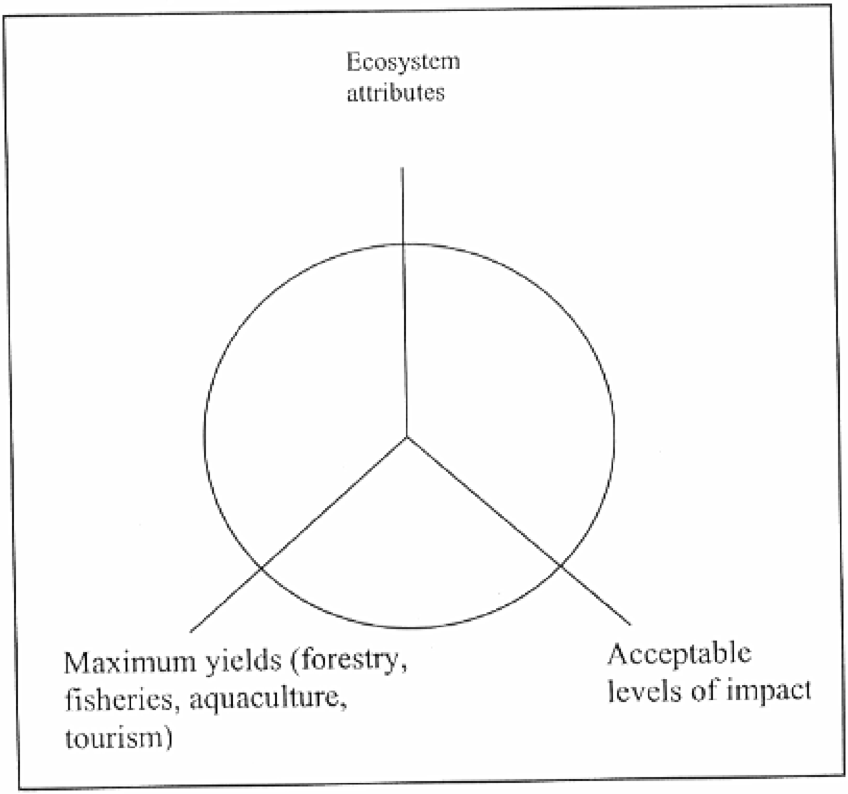


Figure 3. Multi-dimensional Concept of Environmental Carrying Capacity.



objective, as more investments might not necessarily lead to more income. For example, more fishing vessels would not lead to higher yield if the resources are depleted. If applied appropriately, ECC can contribute to the formulation of a national macro economic policy for sustainable development. The right application fields would include ECC in the rate of certain type of economic growth for human settlement in a given area. The top policymakers should be alerted to the workings of ECC for national socioeconomic development. In this regard, PEMSEA and other programmes again have a role to play.

2.0 *Methodologies, Applications and Case Studies*

A number of approaches in measuring ECC were contributed ranging from its application in aquaculture in Europe, Hong Kong, Korea, and Japan; in coastal tourism; in spatial land use planning in England and Wales; in determining pollution loadings and estuarine/coastal water quality in PR China and RO Korea; and in determining the entrainment impact of cooling water intake of a coal-fired power plant in the southern coast of Thailand. The different approaches and methodologies, which were developed and utilized for the different sectors and countries, are described below.

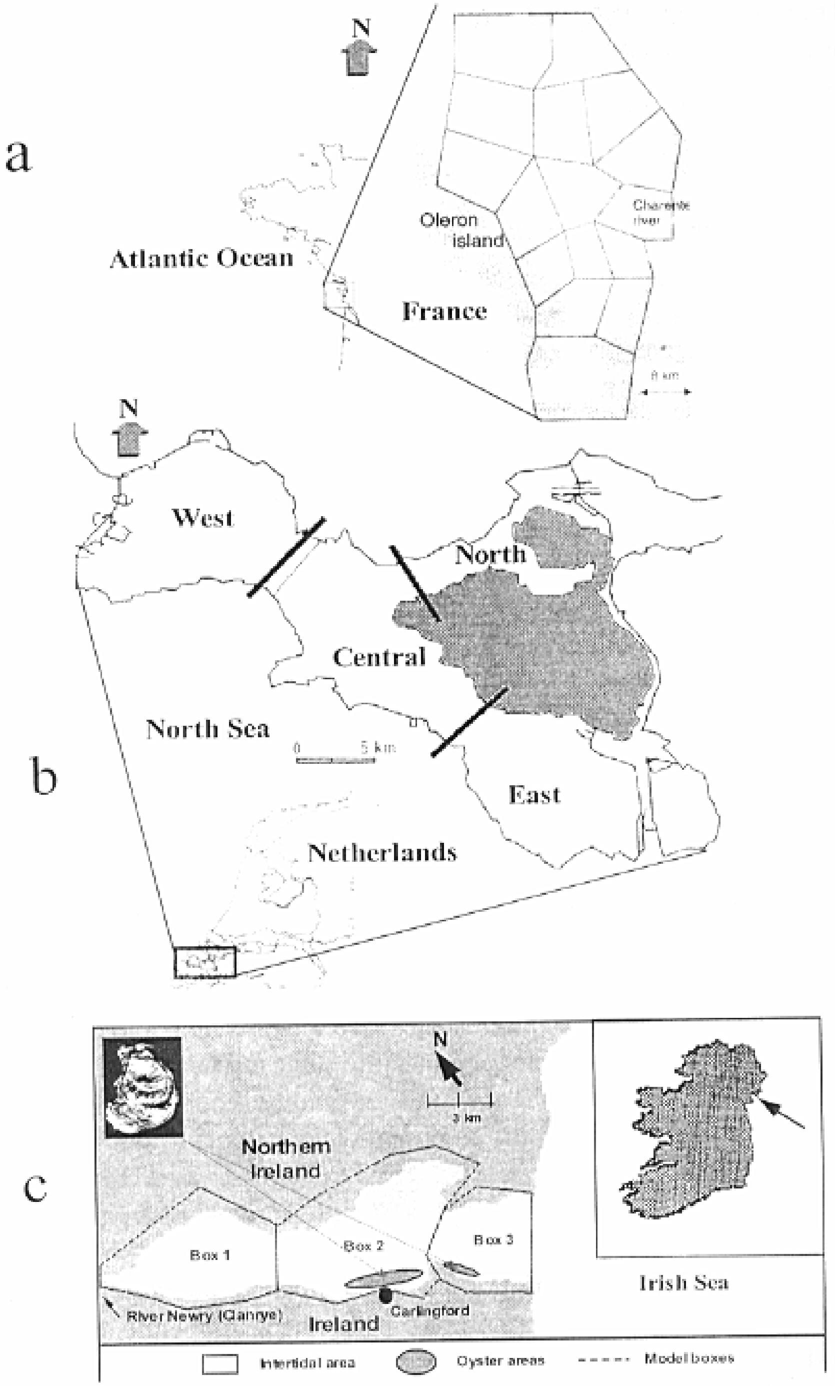
2.1. *Aquaculture*

Several methods were proposed in Europe to estimate carrying capacity for bivalve aquaculture. These approaches may be divided into two main categories: calculation budgets and mathematical models. Mathematical models included box models, coupled physical-biogeochemical models, and local depletion models.

Calculation budgets are based on the comparison between the time scale for phytoplankton biomass renewal, calculated from the biomass-production ratio, the time scale for water renewal or the water residence time, and the time scale for bivalve filtration - the time it takes for the bivalves to filter all the water within the ecosystem.

In box models, culture systems are viewed as a distinct compartment or state variables. Flows of energy or material between compartments are quantified based on internal biological fluxes mediated by external forcing functions. To account for spatial heterogeneity, the ecosystem may be divided into model boxes. Box size determines the spatial resolution of the model. Box models were developed for Marrennes-Oleron (France), the Oosterschelde ecosystem (Netherlands) and Carlingford Lough (Ireland) to estimate the carrying capacity for bivalve culture and/or to evaluate the impacts of projected reductions of nitrogen loads on ecosystem carrying capacity for bivalve growth. (Fig. 4) In the case of Carlingford Lough, the model suggested that the area was being exploited below its carrying capacity.

Figure 4. (a) Marénnes-Oléron Model, (b) Oosterschelde Box Model, and (c) Carlingford Lough Box Model.



In coupled physical-biogeochemical models, physical and biogeochemical processes are computed simultaneously over the same temporal and spatial framework. Such a model was implemented in Sungo Bay (PR China) under the Program INCO-UE. The results suggest that Sungo Bay is already being exploited close to its carrying capacity for scallops.

Local depletion models, on the other hand, are applied to smaller spatial scales (i.e., cultivation unit). In the case of Sungo Bay, a software tool has been developed that integrates a local depletion model with a geographic information system (GIS) interface which allows the user to choose a particular area on the GIS and run the model to analyze its production potential.

The various methods discussed in this work for carrying capacity estimation require different degrees of knowledge about a particular ecosystem and different calculation methods. The budgeting approach is relatively easy to apply, requiring a limited quantity of information. Modeling approaches are more demanding in terms of data and computational tools but may be more accurate because they include feedbacks between culture systems and the ecosystem. They also consider spatial and temporal variability. It is assumed that the fully coupled physical-biogeochemical models are more accurate representation of the real systems than box models. Their high spatial resolution makes it possible to analyze different aquaculture scenarios not only in terms of densities but also in terms of spatial distributions, mostly in multi-species culture systems. The major drawback however is the considerable time required in the calibration and validation processes. Ideally, a model should be applied to a particular ecosystem, using similar equations and parameters but different spatial and temporal resolutions to choose the proper spatial and temporal scales. Finally, local depletion models may be used to parameterize effects at larger scales giving more realism to larger scope models.

In Hong Kong, a simple and robust methodology for assessing the carrying capacity of fish farms was developed through numerical tracer experiments to determine the flushing rate of marine fish farms located in semi-enclosed shallow embayments such as Sok Kwu Wan, Ma Wan, and Tolo Harbour. Robust 3D hydrodynamic and mass transport models provided the flow field and computation of the tracer mass concentration. This computed flushing time for the wet/dry seasons is coupled with a quasi-steady diagenetic water quality model that includes the nutrient kinetics in the water and the sediment-water-pollutant interactions. The long-term average water quality in the fish farm can be assessed by reference to key water quality indices, e.g., chlorophyll-a, dissolved oxygen, organic nitrogen and potential lowest dissolved oxygen level on a day of negligible photosynthetic production. The overall modeling framework is illustrated in Figure 5. Results show that the carrying capacity is dependent on both the flushing rate, the pollution loading, and water depth and volume of the fish farms. The methodology could be possibly applied for the environmental management of mariculture in other sub-tropical Southeast Asian region.

In another study in Hong Kong, two simulation models, *viz.*, a 2-D, 2-layer hydrodynamic and a 3-D tide averaged water quality model were used to simulate the effects of marine fish farming on ambient water quality in Three Fathoms Cove including their applications to assist management in estimating environmental carrying capacity for fish culture operations. Figure 6 shows the schematization of the water quality model applied to the study site. Each model segment was divided into two elements vertically by a horizontal interface. Model segment 2 represented the fish culture site. In segment 3, located at the seaward boundary, seasonal variations of water quality parameters for the model were obtained from field observations. The results of the simulation revealed changes in water quality around the mariculture site due to fish farming activities. The models also demonstrate the effects of variation of feed loads on ambient water quality in the water column. By using both models, one can predict the upper level of fish stock and hence the pollutant loadings that can be maintained at a culture site without violating acceptable water quality objectives defined by the authority. By comparing the output of water quality data under different scenarios of stocking density, the models can serve as effective tools to derive scientifically sound management decisions

Figure 5. Schematic Framework of Modeling of Carrying Capacity of a Fish Farm.

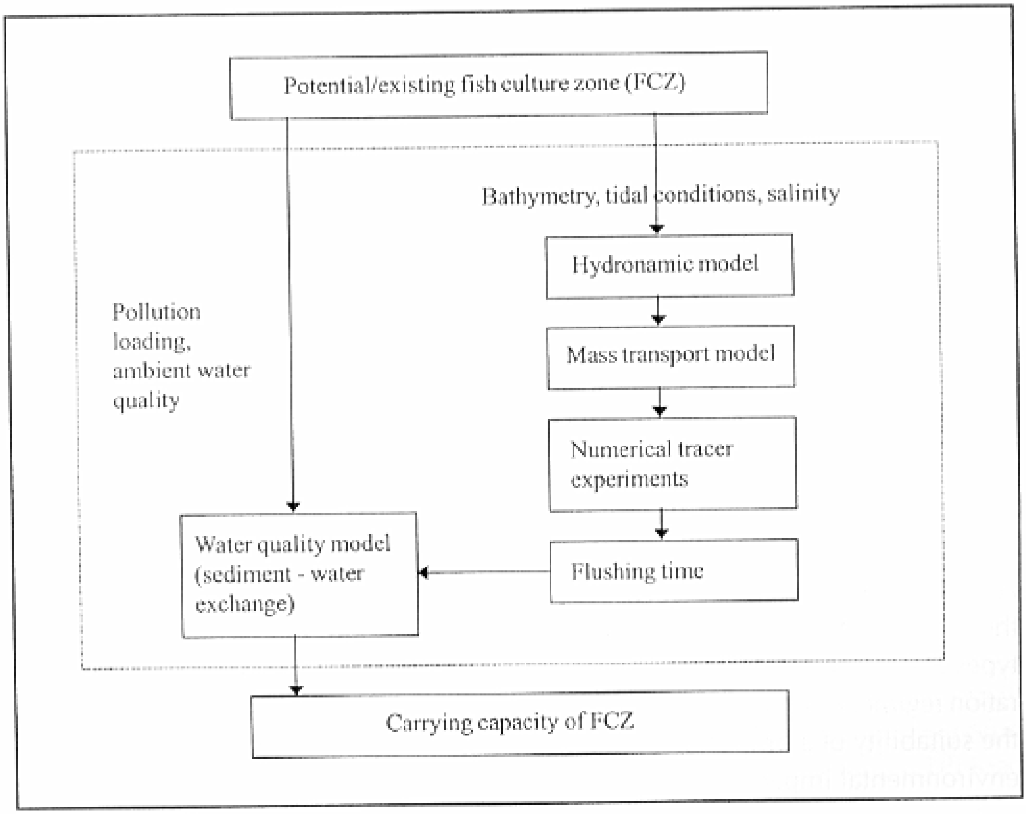
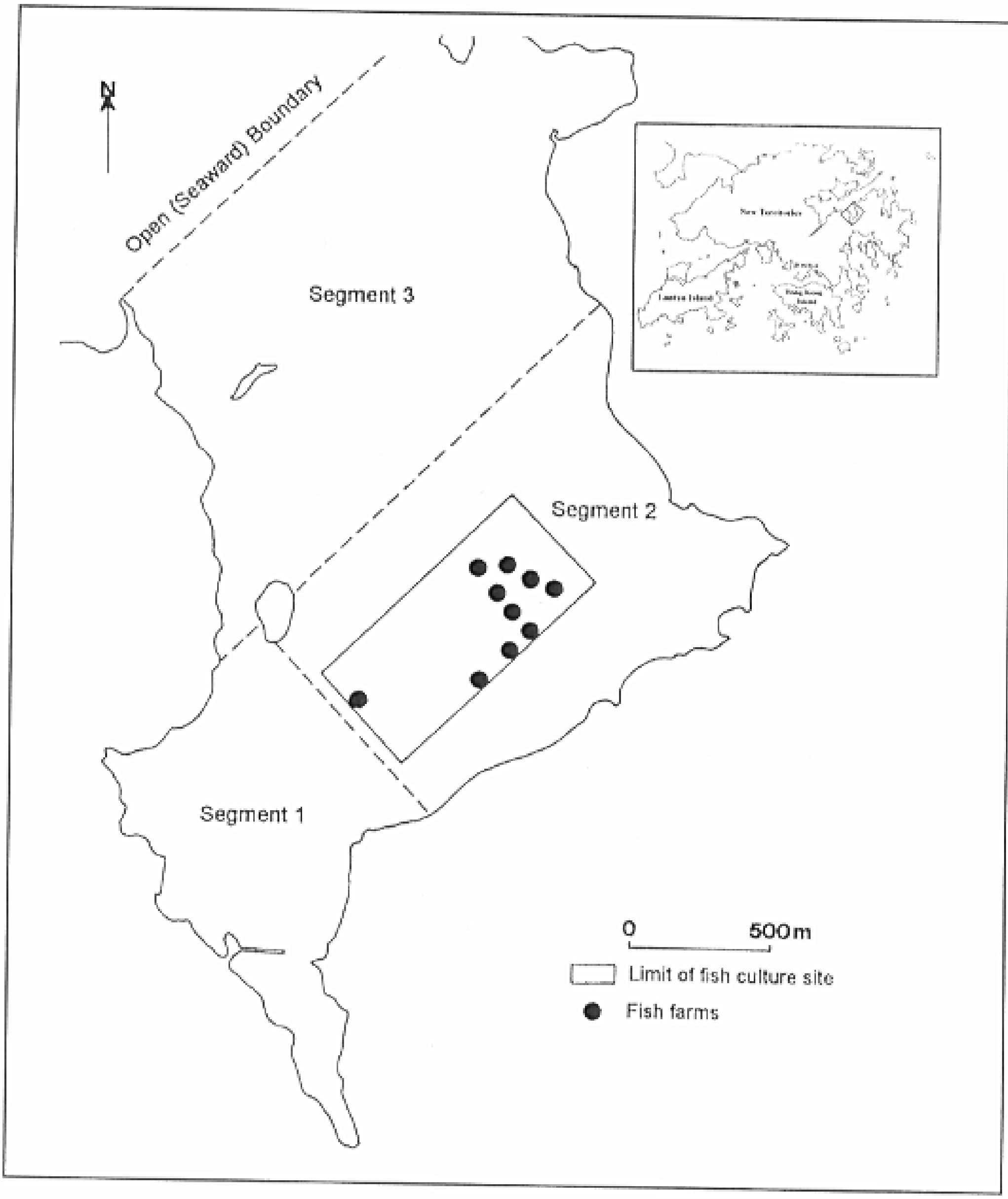


Figure 6. Location of Mariculture Site, Three Fathoms Cove, Hong Kong, and Schematization of the Water Quality Model.



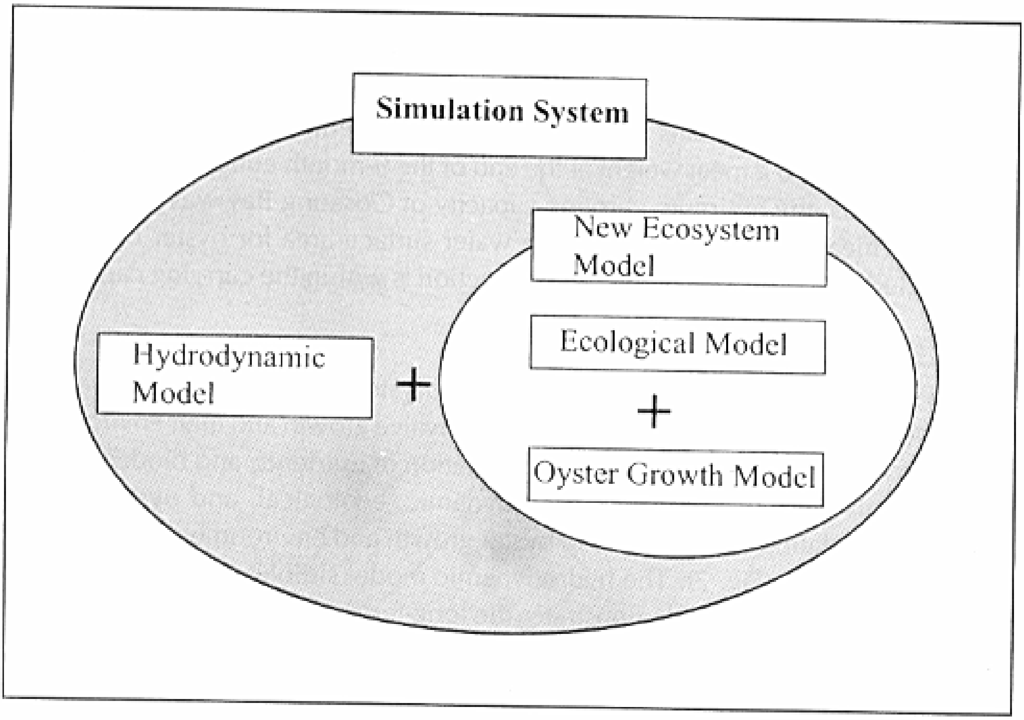
on the maximum fish stock permissible at a particular fish culture site so that acceptable water quality objectives can be met for the sustainable development of the industry. The models may also be used to test the effects of changing feed types and culture species on water quality under the same growth stage and food ration regime. Moreover, the models can serve as a useful planning tool for assessing the suitability of a proposed marine fish culture site prior to its operation, e.g., for environmental impact assessment.

In a modified ecosystem model in Goseung Bay, Republic of Korea, an oyster growth sub-model coupled with EUTROP II was developed and applied to estimate the carrying capacity of oyster culture. The model was specifically designed to simulate interactions between oyster growth and their environment including the physical and biochemical processes in the shellfish system. The simulated results suggested that 16 individuals m^{-3} was a reasonable seeding density to obtain the marketable size of 6 g meat weight at the end of the 9-month culture period. Based on these results, the optimum carrying capacity of Goseung Bay was estimated at 1,500 M/T meat weight considering the water surface area for oyster culture. It was concluded that the present oyster production is within the carrying capacity of the bay.

In a similar study in Goseong Bay, a new ecosystem model was developed and applied to determine the relationship between bivalve growth and their environment through consumption of phytoplankton, excretion of nutrients, and biodeposition. Three models were utilized, *viz.*, hydrodynamic, ecological, and oyster growth models in simulating the dynamics of oyster growth and environmental conditions in shellfish system (Fig. 7). The hydrodynamic model simulates the 3-D physical field in the coastal bay and demonstrates the long-term variability of flow field, salt and heat transport. The ecological model, on the other hand, simulates the flux of carbon, nitrogen and phosphorus elements plus oxygen production and consumption in the pelagic system. Finally, the oyster growth model is based on the scope for growth (SFG), which is calculated as the net result of energy gain by feeding, energy loss, maintenance (respiration and exception), and reproduction. The new ecosystem model was designed to simulate interactions between oyster activity and environment including the physical and biochemical processes in the shellfish system. Time series of biological and environmental observations from the shellfish system are used to calibrate the model. The simulation clarified that the oysters in the shellfish system play an important role in removing phytoplankton and releasing nutrients for regeneration of phytoplankton in the water column. Moreover, the sensitivity analysis suggested that physiological parameters of phytoplankton and absorption efficiency of oysters are the most critical factors to oyster growth.

In Hiroshima Bay, Japan, a year-to-year variation in the lower trophic level ecosystem including oyster culture was undertaken from 1984-1996 using a numerical ecosystem model to verify the causes of decline in oyster harvest in the bay. Results show that when no oyster is cultured, the concentrations of chlorophyll-a, dissolved organic phosphorus and detritus increase in the upper layer, and dissolved oxygen (DO) concentration decreases in the lower layer. This means that oyster culture plays an important role in preserving the marine environment of Hiroshima Bay. The product of oyster culture was highest when the concentration of chlorophyll-a in the upper layer was 7 ug/l and the total phosphorous (TP) load

Figure 7. Simulation System Utilized in Assessing the Relationship Between the Oyster Population and the Environment.



from the Ohta River was 0.5 ton/day. Thus, it is necessary to keep the TP load from the Ohta River at 0.5 ton/day for the sustainable oyster culture in Hiroshima Bay.

In Otsuchi Bay, Japan, a three-year project is currently in progress to understand material cycling in shellfish and seaweed aquaculture ground. Non-feeding aquaculture of seaweeds and shellfish functions as recovery of terrestrial nutrient load as well as food production. Since the culture organisms compete with natural populations for resources, viz., nutrient salts and food particles, evaluation of carrying capacity of coastal waters is crucial for the sustainable exploitation of biological productivity. Two major criteria are proposed: (a) an accurate estimation of phytoplankton primary production, which governs the magnitude of total biological productivity; and (b) an understanding of oxygen dynamics. In addition to the kelp (*Undaria pinnatifida*), intensive aquaculture of the scallop (*Patinopecten yessoensis*), and Japanese oyster (*Crassostrea gigas*) is a major activity in the bay. Cultured kelp and shellfish are incorporated in a 3-D physical-biological model. Results show that phytoplankton primary production consistently exceeded that of the cultured kelp. Therefore, herbivorous consumption of phytoplankton by zooplankton and cultured shellfish is important in evaluating the carrying capacity of the bay. While rapid consumption of dissolved oxygen was observed in the bottom sediment below the culture rafts due to intense accumulation of fecal matters, no anoxic water was formed. Observations indicated that a continuous supply of dissolved oxygen through water flow along the bottom was considerable which was attributed to wind-induced circulation, density current, and internal tide. This suggests that the topography of the Otsuchi Bay indicates an active water exchange and is suitable for aquaculture from the point of view of oxygen supply.

2.2. *Tourism*

In general, the concept of tourism carrying capacity carries several implications: a) a certain number of tourists that a destination area can accommodate; b) the notion of a level beyond which conditions are unacceptable; and c) the impacts on the physical environment and on the psychological attitude of tourists and the social acceptance level of the host population. Three to five major categories or capacities are identified to include the destination or area visited, the tourists themselves and the local or host population (Box 1).

The various capacities are related or commensurate with various tourist impacts in the area and the interaction between tourists and the host population. These carrying capacities need not reach their limits or be exceeded for problems to emerge. The various types of carrying capacities form a basic step in tourism and recreation planning to determine the upper limits of development and visitor use. Various criteria are used to determine the capacity levels, many of which are minimum or basic standards specified for facilities and uses established by planning authorities.

Box 1. Categorization of Tourism Carrying Capacity.

- Biophysical, ecological, environmental (level beyond which an area is degraded or compromised)
- Psychological, perceptual (level beyond which the tourists are no longer comfortable in the area)
- Sociocultural, behavioral (level beyond which the local population do not want tourists)
- Economic (level beyond which tourism is more of an economic liability)
- Physical-facility (level beyond which tourist facilities are saturated); managerial (level beyond which tourism management is no longer effective)

Various attempts have been made to establish the methodology for tourism carrying capacity (TCC) where numerous factors and complex interrelationships need to be considered. It is concluded, however, that the implementation of TCC in nature areas to highly developed destinations faces a wide range of conceptual and methodological difficulties. The usual or traditional approaches to carrying capacity management have little success due to various reasons: 1) the unrealistic expectation of a magic number; 2) untenable assumptions such as direct relationship between tourism and impacts;

3) inappropriate value judgments; and 4) insufficient legal support. This has led to alternative approaches with the shift from establishing use limits to issues of identifying environmental, social, and economic conditions desired by the community, and the creation of growth management strategies for managing tourism's carrying capacity challenges.

Various alternative frameworks have been developed to include more appropriate concepts and processes in recreation management strategies (Box 2).

There are a few examples of carrying capacity in coastal/marine development control. A tourism specialist and a coastal geomorphologist have proposed a generalized model for relating tourism impacts in four different zones of the coast to various types of carrying capacity. Moving in a seaward direction, the four zones with their corresponding tourism use and type of carrying capacity are as follows: 1) hinterland (accommodation and service sector) physical carrying capacity; 2) dunes (transit zone) environmental carrying capacity; 3) beach (recreational activity zone) social carrying capacity; and 4) sea (recreational activity zone) environmental carrying capacity. Carrying capacity for coastal tourism, however, has not been widely applied reflecting a similar situation in tourism: complexities of the coastal environment and the multifaceted character of tourism; continued unplanned tourist expansion along the coast; and difficulty in specifying limits or thresholds that can be applied.

The determination of TCC is therefore extremely difficult and no single formula is applicable to the different types as well as similar types of tourism in various parts of the world because of geographical, ecological and even political, economic, social and cultural conditions. Despite this difficulty, TCC should not be ignored, which is a challenge for tourism researchers.

Box 2. Alternative Approaches to Carrying Capacity Management.

- Limits to acceptable change (LAC) – attempts to solve some problems of identifying maximum use levels and is the most widely used framework
- Visitor impact management (VIM)
- Recreation opportunity spectrum (ROS)
- Tourism opportunity spectrum (TOS)
- Ecotourism opportunity spectrum (ECOS)
- Visitor activity management process (VAMP)
- Visitor impact management process (VIMP)
- Visitor experience resource protection (VERP)
- Tourism optimization management model (TOMM)

2.3. *Ecosystem/Habitat*

An environmental capacity method, linking the science of environmental management with spatial land use planning, was developed for England and Wales for better and informed decision making and to lessen environmental impacts. The method involves a combination of spectrum approach and threshold assessment. The spectrum approach follows a sequential process set within the framework of environmental objectives and appropriate environmental boundaries. Relevant environmental issues are scoped and thresholds at which an environmental impact occurs are identified together with a score. The Spectrum Score ranges through five categories from red (absolute constraint to development) to blue (development encouraged because it will resolve an existing environmental problem). For the orange and yellow categories with predicted impacts on the environment, management options may be identified in order to offer choices for mitigating impacts and thus increasing environmental capacity. The approach was tested and refined through workshops and pilot studies. Figure 8 shows an example of a working matrix from flood risk experts.

The assessment against each criterion recognizes five levels of environmental capacity for proposed development represented by colors as shown in Figure 9.

The first stage of the threshold assessment method is to scope the relevant environmental issues (e.g., energy, waste, air quality, biodiversity, soil quality, flood risk, coastal erosion, groundwater vulnerability, river water flow/quality and fisheries) and identify appropriate boundaries for the particular study, which could be assessing different sites for proposed development or assessing the form of development for a chosen site. For each relevant issue, objectives are identified. This recognizes regional differences and environmental characteristics such that the process is dynamic and can reflect changing capacities. For each issue, thresholds are identified at which there is a consequence or an impact on the environment, requiring options to avoid or mitigate. The thresholds may include both qualitative

Figure 8. Example of Working Matrix.

Criterion:	Flood Risk Management	
Thresholds	•	Flood risk area; clear evidence of risk
	•	Flood risk area; options to orientate development
	•	
	•	Flood risk area; compensation possible
	•	No flood risk; no surface water discharge problems; existing flood problems resolved by new development
Management Options	Provision of compensatory floodplain storage Sustainable drainage systems River restoration and habitat creation	
Interactions	Water quality Biodiversity Amenity and recreation	

Figure 9. The Spectrum Approach.

Red	Absolute environmental constraints to development, e.g. national designations
Orange	Problematical and improbable because of known environmental issues, e.g. area of water shortage; mitigation or negotiation difficult and/or expensive
Yellow	Potential environmental issues; mitigation and/or negotiation possible, e.g. provision of Sustainable Drainage Systems
Green	No environmental constraints and development acceptable
Blue	Development actively encouraged as it would resolve an existing environmental problem

and quantitative measurements. For assessing the relative capacity of alternative sites to accept proposed development, the score for each site is identified as shown in Figure 10.

Development requirements may be accommodated if there are enough scores of green and blue. If not, the approach identifies the further work needed to overcome problems.

The approach presented thus provides a systematic method of informing local planning authorities and developers about the capacity of the environment to absorb land use change.

In Dalian Bay, PR China, the total pollution load control model (TPLC) was applied to mitigate pollution from land-based sources. TPLC deals with the control of environmental quality targets, permitted discharge, targeted discharge from land-based sources, and total discharge from point sources. It is designed to focus on the total quantity of pollutants discharged into the sea from the individual pollution sources at an acceptable level, where socioeconomic conditions are not significantly affected. Permitted discharges from land-based pollution sources are based on the Standards of Wastewater Discharge of China. Under TPLC, the standards used include seawater quality standards, marine organisms quality standards, sediment quality standard and permitted contents of pollutants in wastewater discharge. To predict the total quantity of industrial wastewater, including the trends in variation of seawater quality, a gray and equal-dimensional model was utilized. Water exchange in the bay was studied using the Euler-Lagrange method. The particle model, on the other hand, simulated the transport and diffusion of pollutants in the bay. Results show that the exchange rate of seawater in Dalian Bay is calculated at 14.05 percent, which indicates low exchange capacity. The western part of the bay in particular was identified as highly polluted since most of the pollutants

Figure 10. Example of Spectrum Scores for Comparison of Alternative Sites for Proposed Development.

Criteria	Development potential/locational options						
	A	B	C	D	E	F	G
Transport	•	•	•	•	•	•	•
Energy	•				•		
Waste Management			•	•	•		•
Water Resources					•		
Flood Risk	•		•	•	•	•	•
Biodiversity	•	•		•		•	•
Air Quality	•			•		•	
Contaminated Land		•	•	•	•	•	•
Amenity and Recreation	•	•				•	•
Access							•
Fisheries				•		•	
Heritage		•		•		•	

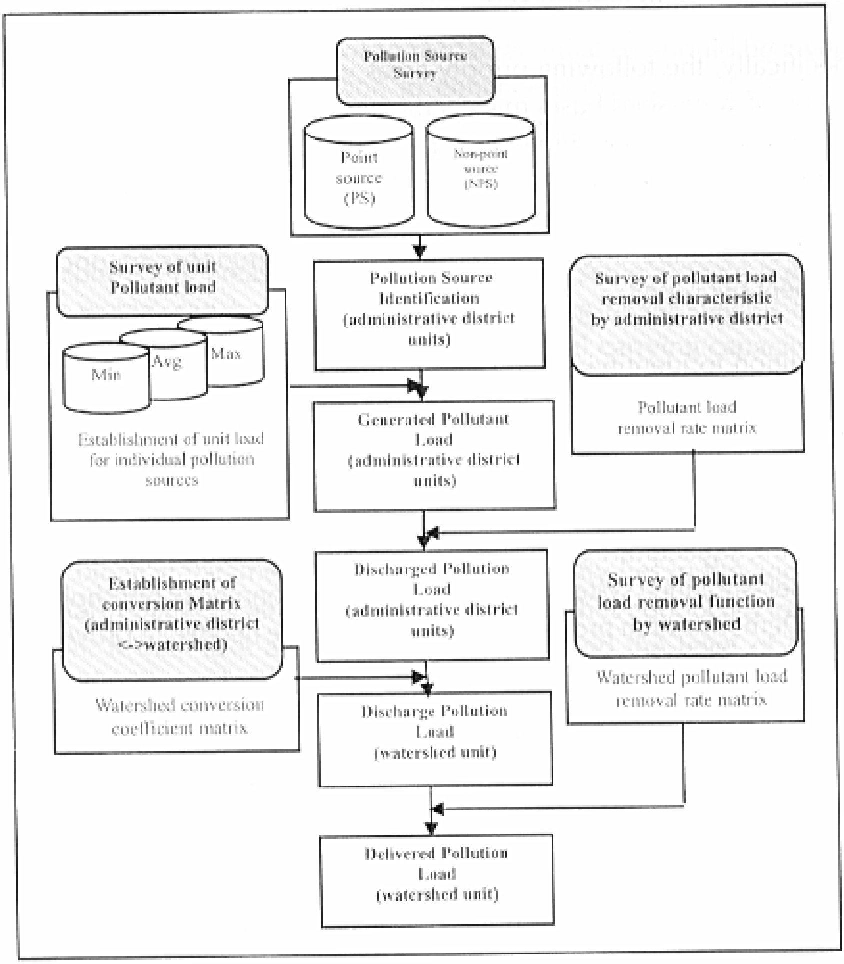
converged in this part of the bay where it is characteristically shallow and the exchange capacity is low. This area was therefore identified to be a potential focus for TPLC studies for purposes of improving the water quality.

In Lake Shihwa, Republic of Korea, the total pollution load management system (TPLMS) was used to estimate pollution load and the assimilative capacity of the lake. TPLMS is a water quality management system where total pollution loads from watersheds and point and non-point sources are controlled within the assimilative capacity of the receiving water. In Lake Shihwa, conventional and toxic pollutants, particularly mercury was recognized as major water quality problems. The major management target, however, was confined to organic materials indicated by chemical oxygen demand in the first phase of TPLMS implementation. This was due to technical difficulties in quantitatively estimating pollutant load and lack of basic information to identify detailed transport passages of the toxic pollutants. For the main part of Lake Shihwa, TPLMS management target was set at 2 ppm in terms of chemical oxygen demand (COD) since it is the required level to support swimming and other water-related outdoor activities as well as the growth of some insensitive marine organisms under the national seawater quality criteria. Pollution load estimation involves identifying the pollution sources (e.g., information from point and non-point sources) and estimating three different kinds of pollution loads, *viz.*, the generated pollution load, discharged pollution load, and delivered pollution load (Fig. 11). Generated load reflects the pollution source at source level before any treatment is done. Since the generated load is significantly decreased before it is finally discharged into public waters, the identification of detailed treatment processes and efficiency of the applied treatment methods are required to calculate the discharged load from the generated load. Considering the natural purification process in streams, which is highly variable and site-specific, measurements are made directly on the stream flow and on pollutant concentration under different flow conditions to estimate the delivered load and delivery ratio, i.e., measured delivered load/ discharged load in the upstream watershed. In coastal water quality management, assimilative capacity is usually indicated by the targeted pollutant load. Thus, it can be defined in terms of generated load, discharged load, and delivered load in accordance with management requirements. In Lake Shihwa, assimilative capacity was indicated by the delivered load because it was used as the model input to predict the lake's water quality.

In Kwang-Yang Bay, RO Korea, the establishment of a modeling framework as a management tool has been suggested due to recent active development in the area. A three dimensional hydrodynamic-eutrophication model (HEM-3D) consisting of hydrodynamic and eutrophication models, which has been listed as a tool for total maximum daily load (TMDL) development in the USA, has been applied. The hydrodynamic model, often referred to as environmental fluid dynamics code (EFDC), is a real time model based on continuity, momentum, salt-balance, and heat-balance equations. The model simulates density and topographically induced circulation, tidal and wind-driven flows, and spatial and temporal distributions of salinity and temperature. The water quality model, on the other hand, is based on mass-balance equations for 22 water quality variables in the water column including suspended algae, dissolved oxygen, and cycles of carbon, phosphorus, nitrogen and silica. Since numerical models of hydrodynamic and water quality conditions are dependent on understanding the prototype processes, the importance of data is emphasized. A comprehensive data set, including simultaneous measurements of hydrodynamic and water quality parameters and external loads are essential for reliable water quality modeling. A general water quality model applicable to any system and to any problem is not available yet. Characteristics of the system to be modeled, specific problems areas and data availability affect the modeling of physical transport biogeochemical processes. Hence, water quality modeling, as it stands now, is site-specific, problem-dependent and data-dependent. Advances in our understanding of physical transport processes would eventually result in a general hydrodynamic model applicable to any system. Advances in biogeochemical processes would result in the refinement of kinetic formulations and in the explicit incorporation of the components of the high trophic levels into the numerical model.

In the southern coast of Thailand, the carrying capacity concept was applied in determining the entrainment impact of cooling water intake of a coal-fired power plant. The term ecological carrying capacity was arbitrarily adopted as the capacity that a unit volume of water could carry certain biomass of aquatic animals. Data

Figure 11. Flowchart of Pollution Load Estimation.



on fishery production of every type of fishing gear from both commercial and artisanal fisheries were used. The method entails counting the animal larvae present per unit volume of water. Considering a fishing area of 11,000 km^2 with an average depth of 15 m, the total volume of fishing water is about 165 billion m^3. If taken against the total animal production of 1.5 million MT, each cubic meter of seawater in the area can carry as much as 0.009258 kg of animals. The total loss in situ of aquatic animals per year is therefore estimated at 2.2 million kg/yr. Mitigation measures proposed to compensate for this loss include the installation of artificial reefs and establishment of a sea ranching program.

3.0. Research Needs and Priorities in ECC Research

3.1. Research needs and priorities

The following research areas were identified for ECC: a) carrying capacity for fish stock assessment; b) ECC for aquaculture in different environments, in enclosed and open systems; c) ECC for human settlements in coastal island systems; d) ECC for tourism and multiple uses; and e) watershed-based modeling.

Specifically, the following priority areas were identified for ECC research: a) integration of watershed basin management with coastal water quality modeling; b) carrying capacity for multiple uses; c) nutrient loading; d) ecosystem modeling - integrating the biological and physical attributes; e) linking harmful algal blooms in the perspective of ECC; f) shellfish culture as a nutrient sink/remover; f) multi-species/multi-compartment modeling; and g) principles of uncertainty for management decisions.

In addition to identifying the priority areas mentioned above, the group also contributed case studies for consideration in ECC research.

a) Case study on ECC for feeding aquaculture practices

In feeding aquaculture practices (e.g., marine fish culture), the major issue is to achieve a self-sustained system that is below the environmental threshold or carrying capacity. Demonstration projects in coastal waters are needed to prove the concept is feasible and practical in the local context.

In addition to applying modeling techniques to estimate the maximum permissible stock density, the introduction of integrated culture practices is highly desirable. One such integrated culture approach is to deploy artificial reefs, which act as biofiltration units to assimilate the extensive nutrients generated by marine fish culture. To further enhance the biofiltration function/efficiency of the artificial reef units, growth of non-feeding aquaculture species such as mussels, scallops or oysters can also be integrated with the system. This system provides the following benefits:

1) removal of excessive nutrients by non-feeding aquaculture species, e.g., bivalves, and organisms attached to the reefs;
2) aggregation of fish to the artificial reefs, which can further enhance the removal of excessive feed from the culture practices; and
3) development of recreational fishing activities hence, deriving income from the reef fish population.

Disciplines required for demonstration projects:

Marine biologists and ecologists (ecophysiologists), modeling experts, economists (to assess marketability of the cultured bivalves), marine engineers (to design artificial reef structures) and local fishermen (cooperation).

b) Aquaculture for bioremediation

The ecosystem approach to environmental remediation should be given close attention as a biological intervention to enhance ecological carrying capacity. Species perform specific roles in nature. Some species serve as biofilters and are effective in filtering suspended sediment, absorbing nutrients, and fixing pollutants such as heavy metals. Multi-species culture can help to overcome or reduce impacts resulting from a mono-species culture. In enclosed areas such as bays, this strategy should be explored. In areas such as marinas, which serve a single purpose, i.e., boating, the waters can be improved by introducing target species to remove nutrients and/or pollutants and increase the ecological carrying capacity.

c) A practical solution in determining limits of acceptable change for coral reefs and mangroves

Research is still needed to be able to accurately quantify ECC in terms of biological/ecological parameters. Much of the difficulty lies in the wide range of environmental variables operating in nature. If research is conducted to determine quantification, it may take another 30 years or so and the level of satisfactory resolution will still not be reached. In view of the extreme pressures for such quantification, examples currently practiced in the region, e.g., for coral reefs, a 25 percent no-take area appears to be emerging for systems where demand comes from its village community can be considered. This exemplifies a Limit for Acceptable Change (LAC) and with further research, the level can be altered in either direction.

For mangroves, studies have shown positive correlation between areal extent and adjacent shrimp production, which can support decision-making. The Matang mangrove forest reserve, where sustainable logging over a 30-year cycle has been practiced for the past 100 years is already a clear indication of sustainable utilization of a mangrove forest.

d) ECC application for development planning and strategic environmental planning and management

1. Identify a mechanism to integrate the scientific details as ECC is defined, measured, and assessed differently for various environmental issues and show interactions.
2. Establish means of communicating ECC using the same language as policy and decision-makers, and complement their own methodologies, i.e., need to provide the information that can be understood by users and stakeholders.
3. Investigate the procedure in which ECC would be used by policy and decision makers to assess how it might affect the continuing development of ECC methods and to identify priorities and timescales.

4. Study interactions between ECC and socioeconomic issues to ensure integration (not trade-offs) within sustainability appraisals used for monitoring progress to sustainable development.
5. Identify indicators which are compatible with those used on sustainability appraisal (SA).

e) Ecological modeling for carrying capacity assessment

Ecological models for carrying capacity assessment should be coupled with physical-biogeochemical models where physical, chemical, and biological processes are simulated within a common spatial and temporal framework. The models should include sediment-water-atmospheric interactions and trophic-chain dynamics in terms of mass and energy transfers. Trophic groups or biological species should be considered, depending on available knowledge about a particular ecosystem and also on the relative importance of different trophic groups and species.

The success of such models depends mostly on field and experimental data for the processes being modeled. Of special relevance is the accurate representation of boundary conditions in terms of loads of pelagic variables, such as nutrients and suspended matter. Ideally, watershed basin models should provide land boundary conditions, while monitoring data or larger scale models should provide sea boundary conditions.

For a particular ecosystem, it may be necessary to develop smaller scale models. For example, in the case of aquaculture, local depletion models may be implemented to analyze the scale of the cultivation unit. Ecophysiology or demographic models may also be implemented to analyze individual growth and population dynamics of particular species. These models may then be used to parameterize larger scale ecosystem models or else, the smaller scale models may be nested within the structure of a larger scale model.

It is important that ecosystem scale models are designed to accommodate different types and quantities of loadings, cultivation strategies, harvesting strategies, and other uses that may be quantified in terms of water quality or biological variables. This will allow their usage to analyze different management scenarios and corresponding carrying capacity levels for different variables/activities.

f) Watershed basin model

Most water quality problems in the coastal waters are caused by excessive loads of organic and inorganic materials. Loads (causes) have a primary effect on the coastal water quality (response). Accurate estimation of land-derived loads is critical for the management of water quality in coastal waters. However, accurate estimation of loads, particularly non-point source loads, has not been in place. There is a need to establish generally accepted methodologies or approaches for loading estimation. Information on loading is the very basis for the definition of carrying capacity, in the context of water quality management.

A case study with the following steps is proposed to establish a sound protocol for the methodology for loading estimation:

1) Select a case site in which both non-point and point source loads are present.
2) Choose one or several watershed models and set them up (GIS-based) for the case site.
3) Design and conduct field surveys to collect data for model application.
4) Calibrate and verify watershed model(s) using the data.
5) Integrate watershed models with coastal water quality models.
6) Establish and recommend the overall processes of watershed modeling.

g) Water quality model

The assimilative capacity, as well as external loads, affects the spatio-temporal distributions of pollutants. The assimilative capacity may be a direct indicator of the environmental carrying capacity of a coastal system. A reasonably calibrated and verified water quality model has been used as a useful tool for water quality management. There have been substantial advances in the theoretical development in coastal water quality modeling. However, such modeling activities have not been accompanied by comprehensive monitoring program, particularly for the coastal systems in East Asia.

A case study with the following steps is proposed to establish a sound protocol for the methodology for integration of water quality modeling with monitoring programs:

1) Select a case site that exhibits water quality problems such as eutrophication and/or hypoxia.
2) Choose a model including hydrodynamic and water quality appropriate for the chosen site.
3) Design and conduct field surveys to collect data for model application.
4) Calibrate and verify the model using field data.
5) Establish and recommend the overall processes of water quality modeling.

Some of the issues that are necessary to be included in the project are:

1) Exploring the water-sediment interactions to examine the role of benthic sediments.

2) Exploring the exchange fluxes of pollutants across the open boundary.
3) Exploring the internal cycling processes of pollutants.

h) Effects of material transport from the open ocean for the carrying capacity of the coastal water

It is important to clarify the effects of material transport from the open ocean to estimate the threshold of nitrogen and phosphorus loads from land. Otherwise, one cannot estimate the necessary cut of the N and P loads from land to prevent red tide occurrence and/or oxygen deficient water mass in coastal areas. Material fluxes through the sea surface and/or sea bottom must be clarified qualitatively.

i) Tourism carrying capacity

There are many issues to discuss on TCC, if taken from the viewpoint of management. One is the physical characteristics of coastal ecosystems, second is infrastructure facilities to boost tourism, and the third is the tourists themselves. The usual problem that arises from tourism carrying capacity is determining when TCC is reached. Information required by TCC from the scientists include showing various standards on environmental quality adopted by the Blue Flag (an exclusive eco-label awarded to beaches and marinas in Europe and South Africa that symbolizes high environmental standards, and good sanitary and safety facilities at the beach/marina).

The following research needs are identified for TCC: a) Research is recommended on coastal tourism, especially marine ecotourism, which is expanding in Southeast Asia. Many popular coastal resorts such as Boracay (Philippines), Kuta (Bali), Phuket and Ko Samui (both in Thailand) have reached their carrying capacity, resulting in environmental degradation, shortages in water, among others. It would be necessary to conduct various TCC studies in coastal tourism to prevent similar negative impacts, keeping in mind their relationships/ interaction to other coastal uses/sectors. b) Many Southeast Asian countries are not ready to adopt eco-labels for the beaches and marinas because of the implementation costs and the argument that the eco-label is not required as only few beaches are polluted. It was suggested that this is where national governments and organizations like PEMSEA can work through and influence policymakers on the importance of environmental quality of the coastal environment. A regional eco-label can be developed to suit the coastal conditions.

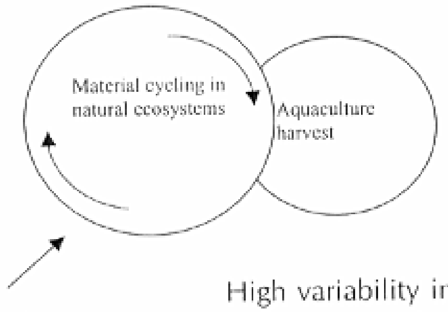
j) ECC study for aquaculture

Functions of aquaculture

1) Food security: protein and oil production
2) Economy aquaculture: market dependent
3) Recovery of nutrient loading: shellfish and seaweed culture

Conflicts with capture fisheries and natural populations

Carrying capacity of coastal waters



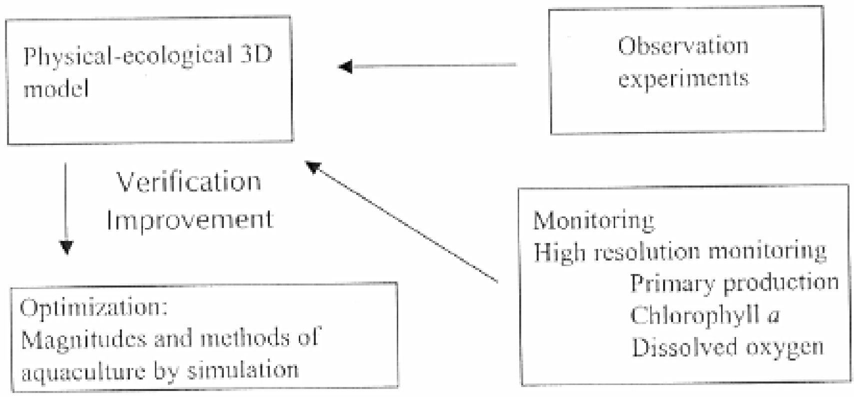
Physical forcing

High variability in primary production and ecosystem structure

Carrying capacity for aquaculture grounds

Requirements:
Understanding of material cycling and ecosystem functions
Identification of critical parameters and key species for monitoring

Strategy for optimizing local aquaculture



3.2. Framework for a case study on ECC

It was suggested that a possible way of addressing ECC research is to have a couple of study sites where different development, e.g., fisheries, tourism or aquaculture are present. Models or any other useful methods can then be put together to integrate the different effects and determine a compromise for the different uses, without going beyond certain thresholds. The usefulness of putting up generic guidelines for ECC measurements representing different environments, levels, and intensities of use was brought up. For instance, one can check on the aquaculture sites in Manila Bay and other areas, the water quality and production rates, and compare them with other bays in the region using the same guidelines.

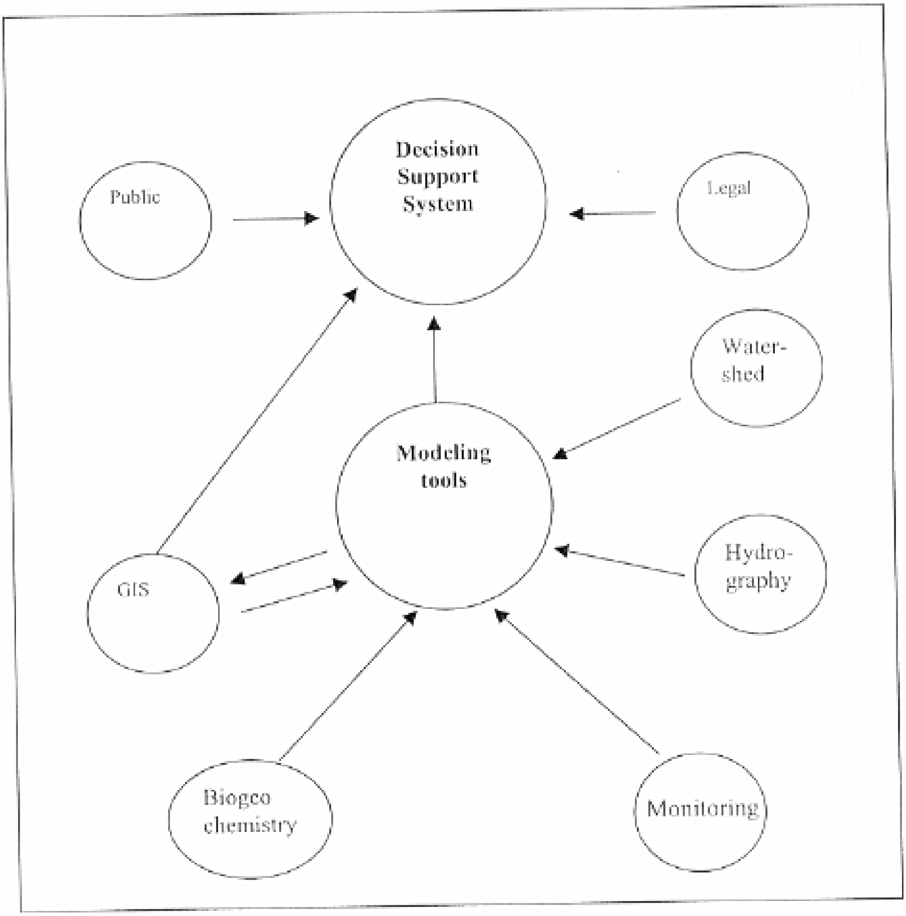
A schematic diagram representing a framework for a case study on ECC was developed integrating a decision support system with modeling tools (Fig. 12). Dr. Duarte cited that there were a number of projects funded in Europe to help define the concept of ECC. They are about to start an EU-funded project that involves some 10 European countries, several universities, and research institutions where ECC is part of the project although the focus is largely on management. Aquaculture, tourism, and pollution loads are included. The idea is to have an integrated management system based on GIS and modeling, to provide managers the basic information for decision on certain development goals. The link between managers and scientists in such decision support systems was emphasized, including the application of the concept to other areas. Interestingly, the proposed project would include not only scientists, but decision makers and end users with management responsibilities as well. The project would facilitate a good communication network between science and decision makers.

4.0. Conclusion and Recommendations

4.1. Conclusion

In summary, the workshop noted that ECC refers to the capacity of the environment and the resources therein to carry the load of human activities, i.e., "development" in particular. To determine the ECC is to find out the levels at which development is sustainable within environmental constraints. In operational terms, ECC may be expressed as a set of thresholds to indicate the break down of harmony between environment and development. ECC is dependent on a specific environmental compartment, its uses by people, and the time the uses occur. It is impossible to identify one threshold for all types of resource uses in different environments at all times. That is why there is no single way of measuring ECC under different circumstances. The right approach is to determine the thresholds for each type of development in given temporal and spatial scales. ECC for a large ecosystem would be the proper aggregation, if possible, of measured

Figure 11. Framework for Case Study on Environmental Carrying Capacity.



thresholds for various types of development occurring in that system at a given period of time.

Although it is difficult to define the ECC in general terms, some basic understanding on the ECC concept may be useful. For this purpose, it may be sufficient to consider the ECC as a level within which the achievement of a certain objective can be maximized on a sustainable basis, without generating unacceptable impacts on the environment, taking into account ecological, economic, cultural and other criteria. However, primary efforts should focus on measuring the thresholds on given temporal and spatial scales, which can be used to induce appropriate and effective management interventions.

4.2. *Recommendations*

The following recommendations, intended for policymakers, resource managers and other stakeholders, including scientists and the public, were developed:

a) Policymakers

- Develop ECC assessment as a component of effective tools and techniques for a cross-sectoral management framework and mechanisms, e.g., integrated coastal management;

- Improve the understanding of basic ecosystem functions, structures and processes; adopt an ecosystem-based management vision, mission and approaches, particularly the anticipatory and precautionary approach, and cross-sectoral working attitude and style;

- Provide policy and financial support to ECC research and assessment to extend the results obtained for management improvement;

- Apply "adaptive management approaches" coupled with ECC research, monitoring and assessment;

- Strengthen environmental monitoring system, through improved sampling design and data quality, taking into account the ECC data requirements;

- For effective application of ECC assessment, appropriate institutional arrangements should be made, including defining of resource rights and application of proper legal/regulatory and market based instruments; and

- For the success of any field projects in ECC research and assessment, a balanced approach to stakeholder consultations and involvement should be adopted, e.g., reconciliation of public and private sector interests, foreign and local interests, rights of moving and indigenous residents, among others

b) Resource managers

- Increase corporate awareness of ECC concept and its merit in including responsible behavior in conserving the environment and resources as a capital, which would contribute to its growth and profit stream over time;

- Undertake study tours to take a look at best/wise practices in the field regarding resource conservation and rational uses, particularly where the ECC concept is being recognized and applied; and

- Support ECC research, assessment and application as part of the efforts to improve resource use strategy and policies.

c) Scientists

- Effect a perception change to undertake management-oriented ECC research and assessment;

- Strengthen communication, by "speaking their language", with policymakers and other stakeholders concerning the ECC concept and its merits and address their needs through proper application of ECC assessment;

- ECC studies should lead to the formation of development and management alternatives and the understanding of the consequences associated with these alternatives, in order to aid decision making, rather than simply providing numbers and figures without appropriate interpretation and assessment of implications;

- Conduct ECC research and assessment in a multidisciplinary manner, through addressing the interconnections, if any, among the thresholds for environment, resource uses, and economic development, e.g., the coupling of water quality model, biomass models and economic effect model;

- Improve sampling design, monitoring schemes and research methodology to enable cost-effective ECC studies and useful results for management improvement;

- Compile and review existing cases of ECC studies, distill experiences and lessons learned, and identify good/wise practices, as a guide to further advance ECC research and assessment; and

- Select an ICM site to validate and demonstrate operational (easy-to-apply) methods for ECC studies and to showcase the benefits of ECC application in management improvement.

5.0. Closing

Dr. Yu summarized the salient points of the discussions. He commended the participants for actively sharing their ideas and insights based on their own experiences and disciplinary backgrounds. Dr. Yu mentioned that the workshop was successful in evaluating the experiences in ECC research in the region and other areas where opportunities for future researches can be identified.

The sustainability of the ECC forum was discussed. Some participants suggested that the forum should be maintained. However, others agreed that a more effective way of linkage among the e-forum members is to develop a joint site study where ICM is in operation. This study should focus on one area of ECC and apply a multidisciplinary approach by looking at tourism, pollution loading, water quality, and others. In this way, networking activities would continue but with emphasis on the development of a joint field project.

The workshop suggested that PEMSEA compile and review available ECC case studies, identify gaps and pinpoint future directions for ECC research and assessment, taking into account the recommendations arising from the workshop.

The group was also informed that the report of the workshop will be published and the draft will be circulated to them for their comments. The scientific papers, on the other hand, will be published in one volume dedicated to carrying capacity.

Finally, Dr. Yu ended with a word of thanks and encouraged the participants to continue supporting PEMSEA's efforts in advancing ECC research.

PART II

WORKSHOP PAPERS

A Review of Current Methods in the Estimation of Environmental Carrying Capacity for Bivalve Culture in Europe

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Abstract

Mollusk culture is one of the most important types of mariculture, with suspension feeding bivalves being among the most cultivated organisms. This is a passive type of culture with bivalves feeding on phytoplankton and detritus. In the last years, there has been a growing concern about carrying capacity of natural ecosystems for bivalve culture, mostly because in several situations bivalve growth was reduced due to overstocking. Mass mortalities have also been a major concern.

Several methods were proposed in Europe for carrying capacity estimation. The simplest are based on average properties of the ecosystem, like water renewal rate, phytoplankton primary production and bivalve clearance rate. If the time scale of the former two processes is larger than the time scale for bivalve filtration then, bivalve standing stock is over ecosystem carrying capacity. This approach does not take into account temporal or spatial variability. More complex approaches are based on ecosystem box modeling. These models are resolved in time and have some degree of spatial resolution. However, since transport processes are averaged, the intrinsic variability of natural ecosystems is poorly represented and spatial resolution is very low to predict local food depletion effects. A more complete approach is the usage of fully coupled physical-biogeochemical models.

The objective of this work is to discuss the advantages and limitations of the different approaches referred above in the light of several case studies. It is argued that fully coupled physical-biogeochemical models are appropriate tools for carrying capacity assessment, namely in the case of multi-species culture systems, allowing to predict appropriate species combinations towards a sustainable aquaculture.

Introduction

Carrying capacity for bivalve cultivation has been the subject of several research projects during the last years. This is because in areas where bivalves are abundant, declines of growth and survival rates have occurred and plans have been proposed to regulate the cultivated biomasses, in order to fit the carrying capacity of different ecosystems. Furthermore, in areas where aquaculture of mollusks is beginning, farmers need to know the maximal densities that may be cultivated to obtain the maximum economic benefit (Héral, 1993). Overcrowded culture conditions may also lead to increased incidence of shellfish diseases (Dijkema and van Stralen, 1989). Besides that, environmental agencies need to know how to regulate bivalve aquaculture in order to prevent ecological impacts. High culture biomass may produce a negative impact to the local environment through organic loading and increased oxygen demand beneath culture leases, phytoplankton biomass reduction and increased nutrient turnovers (Prins et al., 1998; Smaal et al., 2001), potentially leading to degradation of culture environments.

There were several projects financed by the European Union (EU) related to carrying capacity. A workshop was held in Plymouth in October 1996 based on the results of an EU-AIR concerted action (program FAR - Fisheries and Aquaculture Research) on "Trophic capacity of estuarine ecosystems (TROPHEE)". The main objective of this action was to synthesize current knowledge on carrying capacity modeling for bivalve suspension feeders. The results of the mentioned workshop were published in special volumes of scientific journals - Bayne and Warwick (1998), on bivalve ecophysiology and growth and Smaal and Héral (1998), on carrying capacity estimation.

Over the last years, there were considerable progresses on bivalve ecophysiology. Generally, bivalve growth is calculated from scope for growth (SFG). SFG depends on clearance, filtration, ingestion, absorption, respiration and excretion rates. These rates are computed as a function of food quantity and quality, temperature and physiologic parameters. In the literature, it is possible to find equations and parameters describing the ecophysiology of several species - e.g., Barillé et al. (1997) and Ren and Ross (2001) for the oyster (*C. gigas*); Hawkins et al. (1998) for the clam (*Cerastoderma edule*), the oyster (*C. gigas*) and the mussel (*Mytilus edulis*); Hawkins et al. (1999) for the green-lipped mussel (*Perna canaliculus*); and Hawkins et al. (2002) for the scallop (*Chlamys farreri*).

Since the Plymouth workshop, there were some evolutions on available tools for carrying capacity modeling. Some of these were developed under recent intercontinental studies, namely the project "Carrying capacity assessment and impact of aquaculture in Chinese bays" (Program INCO - UE), with the participation of Chinese and European scientists. Progresses in carrying capacity modeling are limited, among other things, by available knowledge on structure and function of natural ecosystems, bivalve ecophysiology and the availability of modeling tools.

The objective of the present work is to analyze the different approaches used for carrying capacity estimation for bivalve aquaculture, emphasizing works carried out in Europe or with the intervention of European scientists and the practical requirements for carrying capacity estimation.

Defining Carrying Capacity

The concept of environmental carrying capacity is very important in ecology and management, not only for species cultivation, but also for other activities such as tourism. With respect to bivalve culture, carrying capacity has been defined as the maximum standing stock that may be kept within a particular ecosystem to maximize production without negatively affecting growth rate (Carver & Mallet, 1990). Alternatively, and more recently, carrying capacity has been described as the standing stock at which the annual production of the marketable cohort is maximized (Bacher et al., 1998; Smaal et al., 1998), or the total bivalve biomass supported by a given ecosystem as a function of the water residence time, primary production time and bivalve clearance time (Dame and Prins, 1998). These definitions are focused on target species, despite a growing tendency in eastern countries for "ecological aquaculture", that is based on multi-species culture where producers and consumers are grown together to facilitate nutrient recycling (e.g., Fang et al., 1996; Grant, 1999). The objective is not only to maximize production, but also to optimize species combinations and distributions in such a way as to reduce the environmental impacts of aquaculture. The growing appreciation of the multiple ecosystems services and the need for sustainable management, means that ecologists are increasingly challenged to model the many interactions between and among species and between species and their environment. A general definition of carrying capacity at the ecosystem level could be "the level to which a process or variable may be changed within a particular ecosystem, without driving its structure and function over certain acceptable limits". Once the "acceptable limits" in terms of water quality and other parameters are established, it should be possible to manage different ecosystem uses in a sustainable way.

There are several examples where carrying capacities for bivalve cultivation have been exceeded by non-sustainable practices. These include the bay of Marénnes-Óleron (France), where oyster (*Crassostrea gigas*) growth has reduced significantly with increased stock densities over the years (Héral, 1993; Raillard and Ménesguen, 1994). Similarly, mussel (*Mytilus edulis*) growth in the Oosterschelde estuary (Netherlands) has been compromised by increased standing stocks (Smaal et al., 2001).

The cultivation of bivalve filter-feeders depends on natural food: phytoplankton and organic detritus. Both the quantity and the quality of these food items are important for bivalve growth (Bayne, 1992; Hawkins et al., 1998). Carrying capacity for bivalve cultivation depends on the rate of renewal of available food. Water temperature may also play an important role (Grant et al., 1992). Suspension feeders have a remarkable capacity to filter the water column such that they are food limited at higher culture density. Therefore, water residence times and phytoplankton doubling times may limit carrying capacity (Dame and Prins, 1998).

The problem of carrying capacity may be viewed at several spatial scales. Among those very important to consider are the scale of the cultivation unit (e.g., farms, rafts, etc.) and the ecosystem scale. The importance of the former results from its direct relevance to the farmers while the latter results from its relevance to whole ecosystem management.

The relationship between bivalve production and bivalve standing stock is parabolic, as demonstrated by the theoretical model described in Bacher et al. (1998) and the work of Ferreira et al. (1998). There is an initial increase in production, but as available space becomes filled up with stock, individual bivalve growth rate is depressed and mortality increases due to several factors associated with overcrowding. The overall result of these effects is a strong reduction of harvest yields above a certain stock threshold.

Estimating Carrying Capacity for Bivalve Cultivation

The approaches to the question of carrying capacity may be divided into two main categories: calculation budgets and mathematical models. Models may be divided into box models, coupled physical-biogeochemical models and local depletion models. All these models are based on known relationships between environmental variables and physiologic and biogeochemical processes. Most of these relationships may be found in the literature (e.g., Jørgensen et al., 1991). In the next paragraphs, each of these approaches will be discussed and case studies presented.

Calculation budgets

Calculation budgets are based on the comparison among the time scale for phytoplankton biomass renewal, calculated from the biomass-production ratio, the time scale for water renewal or the water residence time, and the time scale for bivalve filtration - the time it takes for the bivalves to filter all the water within the ecosystem (Dame and Prins, 1998). These comparisons may be performed over daily, seasonal or other time scales to determine the biomass of shellfish that can be sustained in a given ecosystem. Table 1 shows some results for various ecosystems. In some cases, the filtration time scale is lower than both water residence and the primary production time scales. This is the case of Marénnes-Oléron Bay (France), where the period of growth required for the oysters to reach commercially valuable size increased from 1.5 to 4 years from 1972 to 1985 with the increase in standing stock (Raillard and Ménesguen, 1994).

The budgeting approach is based on averaged values, usually long-term (annual), and cannot incorporate feedback from the ecosystem (Smaal, 1991). Besides that, when bivalve density is averaged over a certain spatial scale, local food depletion effects may be grossly underestimated if average density is much lower than local density (see below).

Box models

In simulation models, culture systems are viewed as distinct compartments or state variables (e.g., bivalve biomass, phytoplankton). Flows of energy or material between compartments are quantified based on internal biological fluxes (e.g.,

Table 1. Water Residence Time, Primary Production Time (biomass-production ratio) and Bivalve Clearance Time for Various Ecosystems (adapted from Dame and Prins, 1998, Falcão et al., 2000 and Duarte et al., submitted).

System	Residence time (days)	B/P (days)	Bivalve clearance time (days)
Sylt	0.5	0.8	4.0
North Inlet	**1.0**	**0.8**	**0.7**
Carlingford Lough	65.8	16.9	490.2
Marennes-Oléron	**7.1**	**10.0**	**2.7**
South San Francisco Bay	**11.1**	**1.1**	**0.7**
Narragansett Bay	26.0	1.7	25.0
Osterschelde	40.0	3.1	3.7
Western Wadden Sea	10.0	1.0	5.8
Ria de Arosa	23.0	0.6	12.4
Delaware Bay	97.0	7.4	1278.0
Cheasapeake Bay	22.0	0.9	325.0
Ria Formosa	1.0	1.6	4.0
Sungo Bay	20	5.5	10.1

Lines in bold correspond to those ecosystems where bivalves clear the water faster than food is renewed though either water renewal or primary production (see text).

grazing) mediated by external forcing functions (e.g., light intensity). The model can be represented as a system of differential equations which correspond to a theoretical perception of which factors are important in bivalve growth. To account for spatial heterogeneity, the ecosystem may be divided into model boxes. Box size determines the spatial resolution of the model. Typically, box size in coastal ecosystem models has a scale of hundreds to thousands of meters. For a description of the general structure of an ecosystem box model with bivalve suspension feeders, see Herman (1993) and Dowd (1997).

Exchange processes of pelagic variables between different boxes and between the ecosystem and its boundaries may be parameterized based on steady-state balances of conservative state variables (e.g., Ruardij and Baretta, 1988) or on simulations with hydrodynamic models (e.g., Bacher, 1989; Raillard and Ménesguen, 1994; Ferreira et al., 1998). In the last case, the results obtained with the fine grid of a hydrodynamic model are averaged over the limits of the ecosystem boxes and over a particular temporal scale to solve the transport equation (see below) usually in one or two dimensions. This averaging implies that a significant part of the system variability is not accounted for in the model. The relative importance of this variability loss should be analyzed on a case-by-case basis. A common argument is that ecosystem models rarely require the spatial and temporal resolution for accurate hydrodynamics (Bird and Hall, 1988). However, the more simplified a model is, in terms of spatial and temporal resolution, the more parameterization is needed to keep it realistic. For example, when bivalve density is averaged over the large volume of an ecosystem box model, it is likely that local food depletion effects will be underestimated. This is because in the real system, bivalves will have access only to a fraction of the total box volume, whereas in the model it is assumed that they have access to the whole volume. Besides that, part of the water cleared by a bivalve stand may have been cleared before by bivalves located upstream. These effects will be better represented in high-resolution models. In low-resolution models, they have to be parameterized using approaches similar to that described in Herman (1993) to account for vertical concentration gradients. More parameters require more measurements. Increasing the number of parameters in a model and therefore its degrees of freedom simplifies the calibration process but increases the probability of getting the right results for the wrong reasons.

Figures 1a, b, and c show the general set ups of three box models presented in Bacher et al. (1998), van der Tol and Scholten (1998), and Ferreira et al. (1998), respectively. The first model was developed for Marrénnes-Oléron bay (France), the second for the Oosterschelde ecosystem (Netherlands), and the last for Carlingford Lough (Ireland). The first and the last models were developed specifically to estimate carrying capacity for oyster culture. The second model was implemented to evaluate the impact of projected reductions of nitrogen loads on ecosystem carrying capacity for bivalve growth. According to the authors, the mentioned reductions may compromise carrying capacity.

Figure 1. (a) Marénnes-Oléron Box Model (adapted from Bacher et al., 1998), (b) Oosterschelde Box Model (adapted from van der Tol and Scholten (1998) and (c) Carlingford Box Model (adapted from Ferreira et al., 1998).

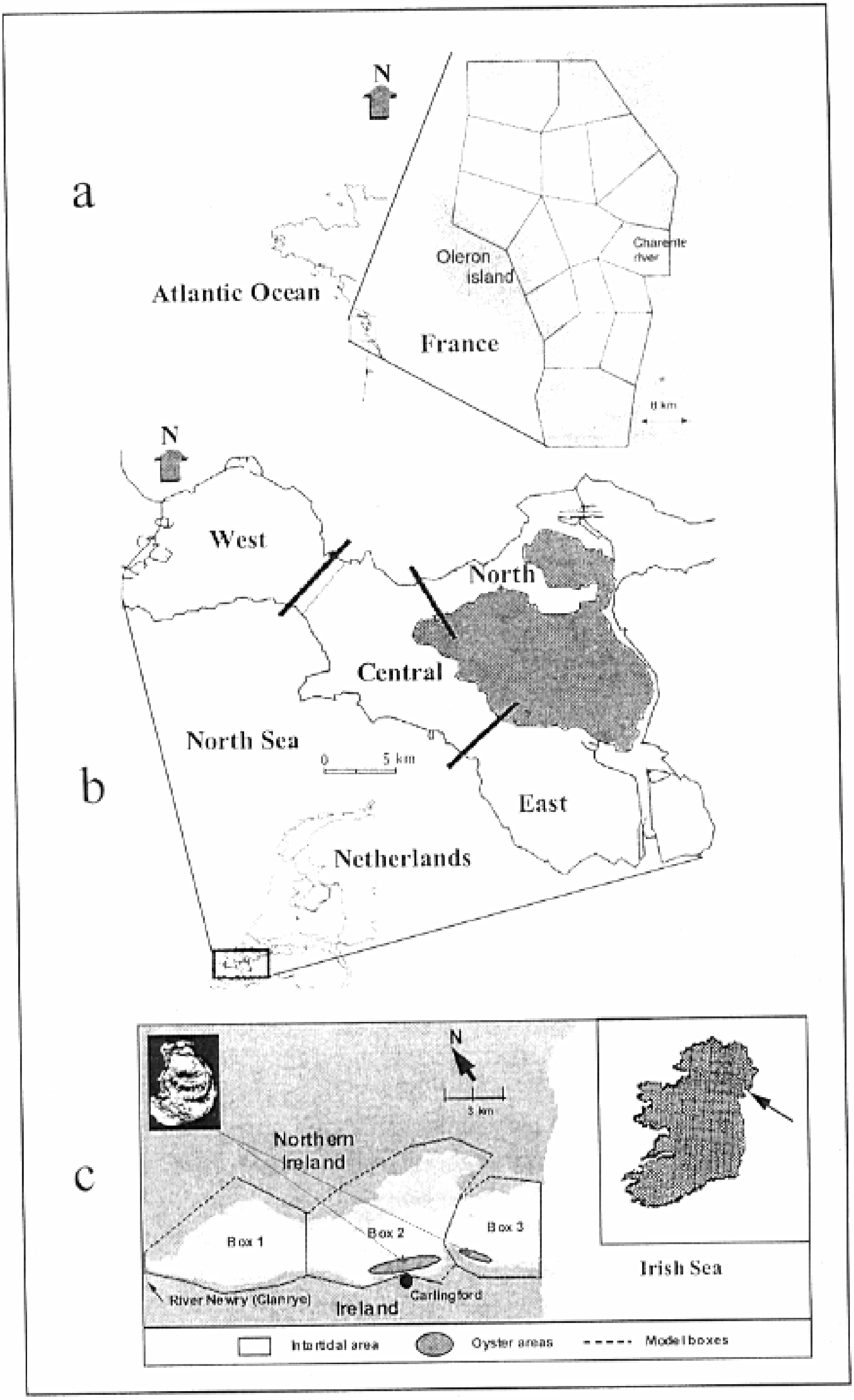
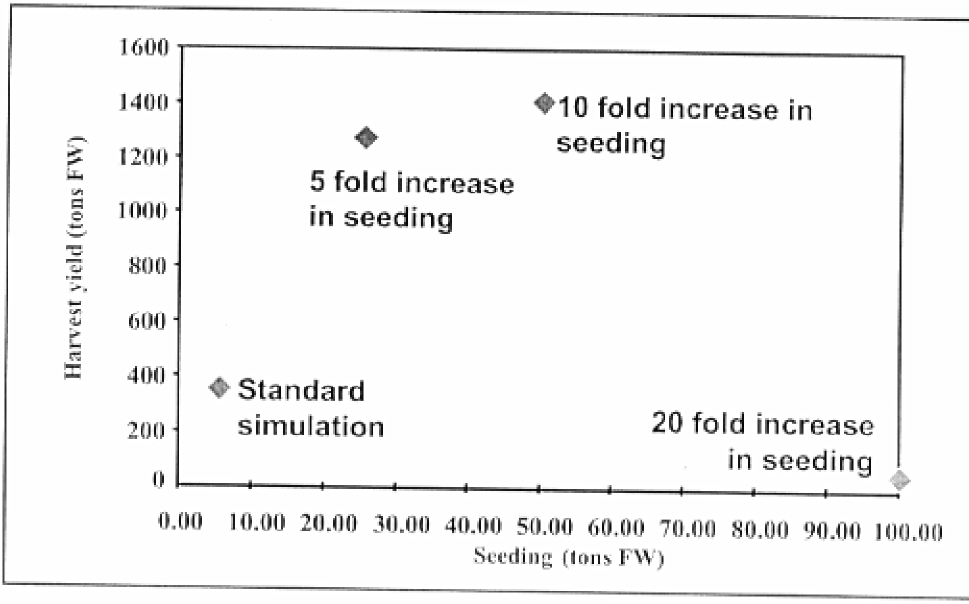


Figure 2. Oyster Harvest as a Function of Seeding in Carlingford Lough, as Predicted by a Model (see text).



The Carlingford model included several state variables such as: dissolved inorganic nitrogen (DIN), chlorophyll concentration (CHL), total particulate matter (TPM), particulate organic matter (POM) and oyster (*Crassostrea gigas*) biomass and density, divided by several weight classes. Light intensity and water temperature were the main forcing functions. The model predicted that harvest yield could be maximized at a level of 1300 tons fresh weight against a seeding of 25 tons of small spat (2 mm oysters) (Fig. 2). This seeding corresponded to approximately 5 X the normal values, suggesting that Carlingford Lough was being exploited below its carrying capacity. A comparable approach was described in Bacher et al. (1998) for Marénnes-Oléron Bay. The last authors obtained a curve for the production versus stock relationship similar to that of Figure 2.

Coupled physical-biogeochemical models

The main difference between box models and coupled physical-biogeochemical models is that in the latter, physical and biogeochemical processes are computed simultaneously over the same temporal and spatial framework. It seems logical to assume that coupled physical-biogeochemical models are a more accurate representation of the real systems than box models. The major drawback of this approach is the required computing time that complicates the calibration and validation processes to a great deal. These models calculate the velocity field with the equations of motion and the equation of continuity (Knauss, 1997) and solve the transport equation for all pelagic variables:

$$\frac{dS}{dt}+\frac{\partial(uS)}{\partial x}+\frac{\partial(vS)}{\partial y}+\frac{\partial(wS)}{\partial z}=A_x\frac{\partial^2 S}{\partial x^2}+A_y\frac{\partial^2 S}{\partial y^2}+A_z\frac{\partial^2 S}{\partial z^2}+Sources-Sinks \qquad (1)$$

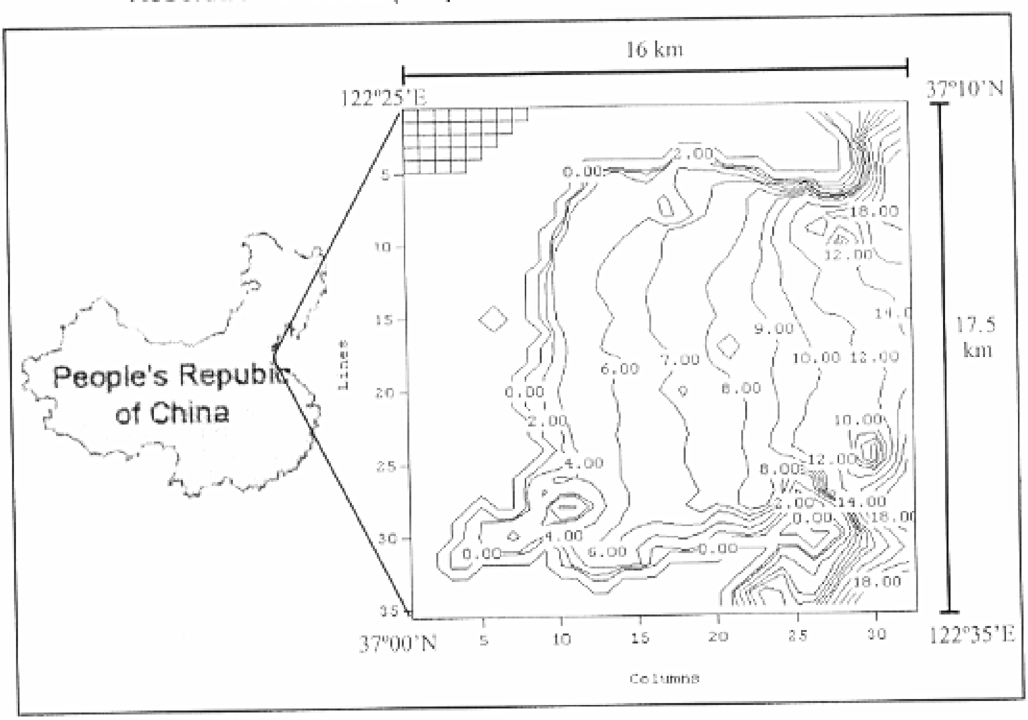
where,

u, *v* and *w* - current speeds in *x*, *y* and *z* directions (m s^{-1}); A - coefficient of eddy diffusivity (m^2 s^{-1}); S - A conservative (*sources* and *sinks* are null) or a non conservative variable in the respective concentration units.

Physical-biogeochemical models may be forced by tidal height at sea boundaries, river flow, wind stress, bottom friction, sunlight intensity, water temperature, etc. (see Duarte et al., submitted). They are based not on ecological boxes of variable shape and size, but on a finite differences, finite elements or finite volumes grid (Vreugdenhil, 1989). Typically, grid cell size in coastal ecosystem models has a scale of tens to hundreds of meters, much closer or even coincident with the culture scale.

In Figure 3, the general set up of a coupled physical-biogeochemical model implemented for Sungo Bay (People's Republic of China) within the context of the intercontinental project "Carrying capacity assessment and impact of aquaculture in Chinese bays" (Program INCO - UE) is presented. The model developed for Sungo Bay is described in Duarte et al. (submitted). It is a 2D vertically integrated

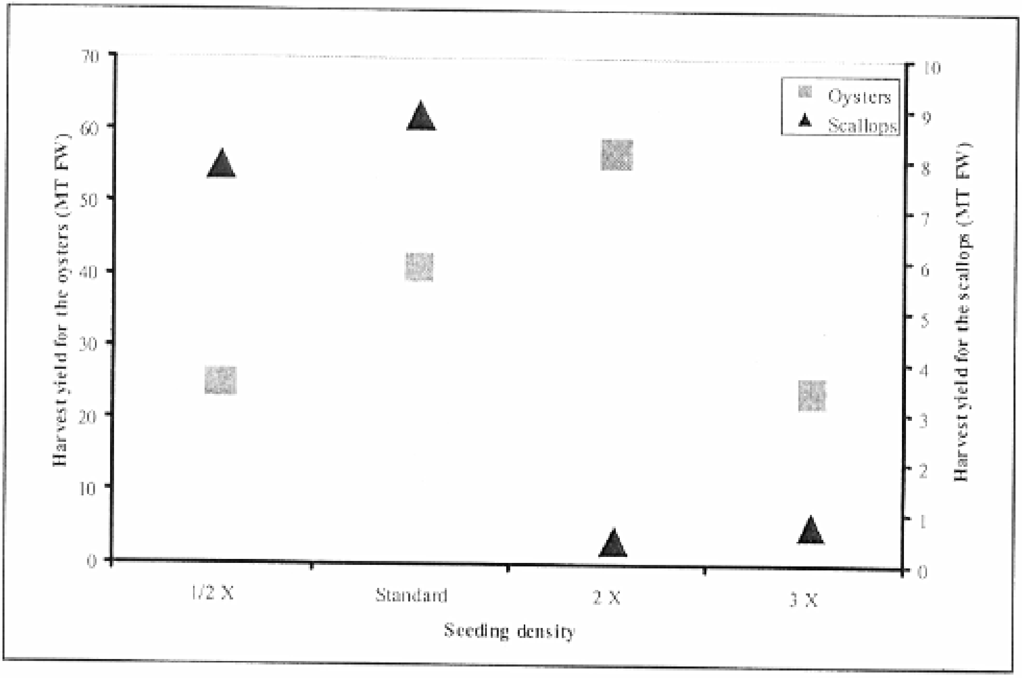
Figure 3. Sungo Bay Model Set up, including Model Domain and Bathymetry (m). Also shown, is a part of the Model Grid (upper left corner), with Spatial Resolution of 500 m (adapted from Duarte et al., submitted).



model based on a finite difference bathymetric staggered grid (Vreugdenhil, 1989) with 1120 cells (32 columns X 35 lines) and a spatial resolution of 500 m (Fig. 3). The model time step is 32 seconds. However, due to the semi-implicit method used for time integrations, each time step is divided in two semi-time steps of 18 seconds. At every semi-time step, one of the speed components is calculated semi-implicitly and the other explicitly, on an alternating sequence. The model has a land and an ocean boundary. It is forced by tidal height at the sea boundary, light intensity, air temperature, wind speed, cloud cover and boundary conditions for some of the simulated state variables. It simulates several state variables such as: water height, current speed, water temperature, DIN, CHL, TPM, POM concentrations and *Laminaria japonica*, *C. gigas* and *C. farreri* biomass and density. The model takes into account the spatial distribution and local density of the cultivated species.

The results presented in Figure 4, show that a scallop seeding reduction to half the current value has almost no effect on production, whereas a two-fold increase leads to a sharp decrease in scallop production. These results suggest that Sungo Bay is already being exploited close to its carrying capacity for scallops. This is in apparent contradiction with the results described above from water renewal, primary production and filtration time scales (Table 1). The results presented in Table 1 suggest that scallops are not food limited, since bivalves in Sungo Bay take more time to clear the water than phytoplankton to double its biomass. However, these calculations are for the scale of the whole bay, without taking into account possible

Figure 4. Oyster (Squares) and Scallop (Triangles) Harvest as a Function of Seeding in Sungo Bay, as Predicted by a Model (adapted from Duarte et al., submitted) (see text).



local food depletion at the culture scale. Furthermore, they are yearly integrated, without taking into account short time food depletion effects. Duarte et al. (submitted) discuss the application of the model to optimize bivalve production, by reducing local bivalve density and mixing up scallop and kelp cultures.

Local depletion models

These models are applied to smaller spatial scales than the previous ones - usually the scale of the cultivation unit. This may be divided into several cells, and bivalves tend to produce a decay on seston supply downstream. Examples may be found in Pilditch et al. (2001) and in Bacher et al. (in prep). Local depletion models are forced by current velocities at the boundaries, solving the transport equation (1) with those boundary conditions and local sources and sinks. They emphasize the potential importance of altering the geometry of the cultivation structures to optimize seston supply. In these models there is no feedback from the cultivation units to the ecosystem. However, they may be very useful, among other things, to parameterize local depletion effects on larger scale models. Bacher et al. (in prep.) developed a software tool that integrates a local depletion model with a GIS interface for Sungo Bay. This allows the user to choose a particular area on the GIS and run the model to analyze its production potential.

Requirements for Estimating Bivalve Carrying Capacity

Carrying capacity estimation encapsulates several scales - the ecosystem scale, the local scale (Smaal et al., 1998), and the individual scale. The ecosystem scale is accounted for in budgeting approaches. It is resolved in box and coupled physical-biogeochemical models, through the division of the ecosystem in boxes or cells. The local scale is resolved in local depletion models. The individual scale is included in all approaches in such as way that the influence of the bivalve standing stocks on the ecosystem is based on measurements carried out at the individual level - e.g., clearance and filtration rates. The upscaling of these measurements is generally based on the assumption of a linear relationship between the effect and the standing stock. This may not be true if local depletion effects are important.

Table 2 includes the requirements for carrying capacity estimation considering the different scales involved. Budgeting approaches have minimal requirements - the less accurate, yet useful methods as a first approximation. The remaining approaches demand more data. Most of the data needed for local depletion models is also needed or at least very helpful for box and coupled models. Ecosystem scale processes are needed to force ecosystem scale models. Local scale variables and processes are needed to calibrate ecosystem scale models and to force local depletion models. Individual scale variables and processes are needed to feedback larger scale processes.

Table 2. Requirements for Carrying Capacity Estimation.

Method	Ecosystem scale	Local scale	Individual scale
Ecosystem budget	Water residence time Phytoplankton doubling time Bivalve stock		Clearance rate
Box models	Exchanges across ecosystem boundaries for variables and forcing functions: Light intensity, temperature, particulate matter, nutrient loads, CHL and zooplankton	Resuspension/sediment.	Relationships between environmental variables and physiologic rates – ecophysiologic submodel, Rates include: filtration, ingestion, absorption, respiration, excretion
		Population dynamics submodel	
	Seeding and harvesting parameters	Seeding and harvesting parameters Local values for state variables and forcing functions: temperature, salinity, TPM, POM, DIN, CHL, zooplankton, bivalve density, etc.	
	Exchange coefficients for different parts of the ecosystem	Local rates: suspended matter deposition and resuspension, bacterial mineralization, phytoplankton, micro- and macrophytobenthos productivity, zooplankton grazing, bivalve mortality, etc.	Body weight, shell weight, shell length, meat weight
Local depletion models		Water depth	
Coupled physical-biogeochemical	1, 2 or 3D Hydrodynamic submodel	Bottom roughness	

Conclusion

The various methods discussed in this work for carrying capacity estimation require different degrees of knowledge about a particular ecosystem and also different calculation methods. The budgeting approach is relatively easy to apply, requiring a limited quantity of information. Modeling approaches require more data and computational tools but may be more accurate, among other things, because they include feedbacks between culture systems and the ecosystem and they consider spatial and temporal variability. Fully coupled physical-biogeochemical models seem to be the most accurate way to represent the ecosystems but demand more computing time. Their high spatial resolution makes it possible to analyze different aquaculture scenarios in terms of densities and spatial distributions, mostly in multi-species culture systems. Ideally, a model should be applied to a particular ecosystem, using similar equations and parameters but different spatial and temporal resolutions to choose the proper spatial and temporal scales. Local depletion models may also be used to parameterize effects at larger scales giving more realism to larger scope models.

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Environmental Carrying Capacity in an Aquaculture Ground of Seaweed and Shellfish in Northern Japan

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Abstract

In non-feeding aquaculture of seaweeds and shellfish the culture organisms compete with natural populations for resources, *viz.*, nutrient salts and food particles. An evaluation of carrying capacity of coastal waters is therefore crucial for sustainable exploitation of biological productivity. For this, two major criteria are proposed: accurate estimation of phytoplankton primary production, which governs the magnitude of total biological productivity, and understanding the oxygen dynamics. Preliminary results obtained using these criteria on a bay located on the northeastern coast of Japan are presented. Bio-optical measurement of both natural and active fluorescence was examined for extensive spatio-temporal coverage. This measurement proved to be useful for rapid, accurate and continuous quantification of primary production. This was exemplified in the present study by field application of ^{14}C and bio-optical techniques to monitor primary production in the aquaculture ground. Growth of the cultured kelp, locally called wakame (*Undaria pinnatifida*) and phytoplankton, was compared as competitors for nutrients during spring. Rapid consumption of dissolved oxygen was observed *in vitro* in the bottom sediment below the shellfish culture rafts due to intense accumulation of fecal matters. However, no anoxic water was formed, as wind-driven circulation and inflow of subsurface coastal waters induced by the internal tide supplied enough dissolved oxygen to prevent the formation of anoxic layer. Evaluation of the carrying capacity of the bay using a numerical physical-biological coupled model is in progress.

Introduction

Coastal aquaculture contributes more than 20 million tons of seafood, with an estimated average annual increase of 2 million tons since 1992 (FAO, 2000). Non-feeding aquaculture of seaweeds and shellfish functions as recovery of terrestrial nutrient load as well as food production. The amount of the recovery by short-neck clam fisheries is estimated to be about a quarter of the total input of nitrate in a brackish lake in Japan (Hino, in prep). On the other hand, fish culture produces considerable amount of organic residues primarily due to unutilized feed and fecal matters that accumulate on the bottom and induce anoxic waters. Nearly 50 to 90 percent of the food materials get released into the environments (Maruyama, 1999). Since non-feeding aquaculture is not accompanied by the accumulation of residual foods, this method poses less environmental threats than feeding aquaculture.

Recently, however, non-feeding aquaculture activities have suffered a setback due to environmental deterioration resulting in a decrease in the annual harvest. General trend in the exploitation of coastal aquaculture productivity has been the maximization of catch and economic efficiency. With frequent occurrences of red-tide and anoxic waters, it becomes rather evident that the trend in maximization may lead to less efficient production through environmental degradation and shifts in marine ecosystems (Barnebe and Barnabe-Quit, 2000). This conflicting situation has led to the realization that knowledge of material cycling in natural ecosystem is a prerequisite to exploit coastal productivity in a sustainable manner, as the culture organisms compete with natural populations for resources, *viz.*, nutrient salts and food particles. Nutrients, the key factor that fuels primary production, are supplied by advection, riverine input, and regeneration. In addition to phytoplankton, naturally occurring and cultured macroalgae are also responsible for the organic matter production. Not only the zooplankton and benthic organisms, but also the cultured scallops and oysters are dependent on the organic matter produced by the phytoplankton and macroalgae. Thus, it is important to quantify production at different trophic levels to evaluate the carrying capacity of coastal waters and estimate the sustainable exploitation of biological production.

Tatara (1992) studied the partitioning of organic matter produced by primary producers to heterotrophs and traced a grazing food chain similar to that of Ryther (1969). In this pioneering work, several issues were considered which control the carrying capacity of coastal waters for fisheries production. One of the major hurdles recognized was the dearth of biological data. In coastal areas, primary production of phytoplankton markedly fluctuates in time scales of hours to days, being influenced by tidal current, inflow of fresh water, and disturbance of water column. Bottle incubation has been the traditional and standard technique used in measuring primary production. However, being time consuming and laborious, it limits spatio-temporal coverage. In recent years, bio-optical approaches, such as remote sensing of ocean color from satellite (Platt and Sathyendranath, 1988), active fluorescence (Schreiber et al., 1994; Kolber et al., 1998), and natural fluorescence methods (Chamberlin and Marra, 1992) have been developed. These bio-optical approaches can provide spatio-temporally continuous data and satisfactory database on primary productivity.

Depletion of oxygen is another essential factor that determines biological productivity in aquaculture grounds. Although no external food is added in shellfish culture, natural particles are intensively fed upon and fecal matter accumulates on the bottom. Without shellfish culture, fecal matter of herbivorous animal can be distributed widely. With shellfish culture, there is localized sinking of fecal matter, as the feeding and fecal-matter production of shellfish is considerably active. Thus, organic matter deposition can cause local anoxic conditions near the bottom.

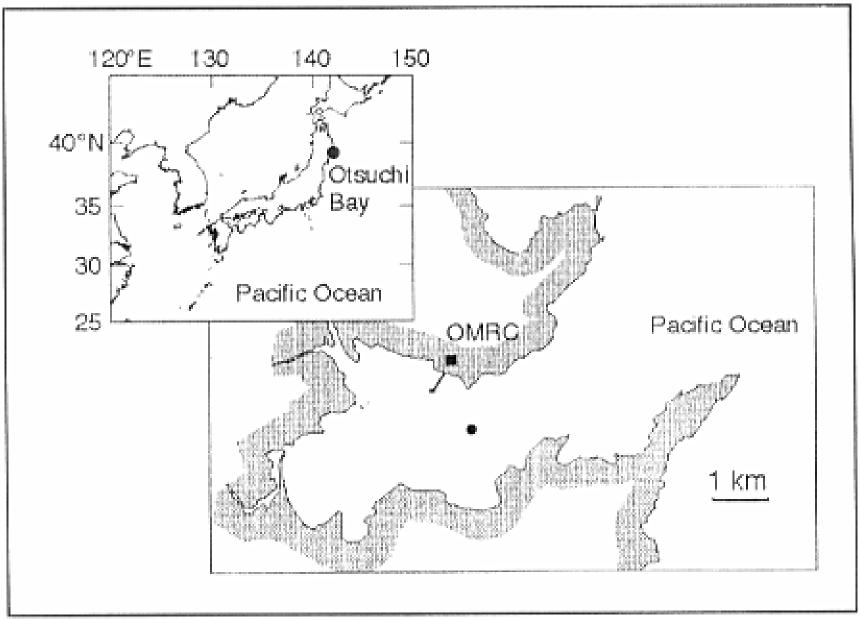
In view of organic matter distribution and dissolved oxygen dynamics, a three-year project is currently in progress in Otsuchi Bay to evaluate the carrying capacity of the bay for aquaculture by understanding the material cycling in shellfish and seaweed aquaculture ground. In addition to the kelp, *Undaria pinnatifida*, intensive aquaculture of the scallop, *Patinopecten yessoensis*, and Japanese oyster, *Crassostrea gigas*, is a major activity in the bay. Preliminary results suggested that evaluation of carrying capacity of coastal waters is crucial for sustainable exploitation of biological productivity.

Study Site

Otsuchi Bay (39°20'N, 141°56'E), located on the Sanriku rias coast in northeastern Japan (Fig. 1) is 8 km long and 2 km wide and opens into the northwest Pacific Ocean. Sanriku rias coast has an array of bays with similar topography to that of Otsuchi Bay. Due to freshwater influx into the innermost part, and the long, narrow topography of the bay, it is expected that water exchange in the bay is brought about by the seaward outflow of surface water over landward-moving inflow of denser, more saline subsurface water from outside the bay. Conversely, an inflow of surface water over outflow of deeper water is also observed during summer. This circulation pattern may alter from the former to the latter, and vice versa in summer, depending on the difference in water density between inside and outside the bay. The former circulation pattern is brought about by westerly wind stress along the axis of the bay. This circulation pattern predominates in the winter and spring (Shikama, 1980). The formation of spring bloom of phytoplankton depends on this circulation (Furuya et al., 1993) and supply of nutrients from outside the bay.

Oceanographic survey was conducted at Stn. F located in the central part of the bay (Fig. 1). Current meters (at 8 and 30 m depths) and a natural fluorescence sensor (INF-300, Biospherical; at 7.5 m depth) were moored at Stn. F.

Figure 1. Stn. F at Otsuchi Bay.

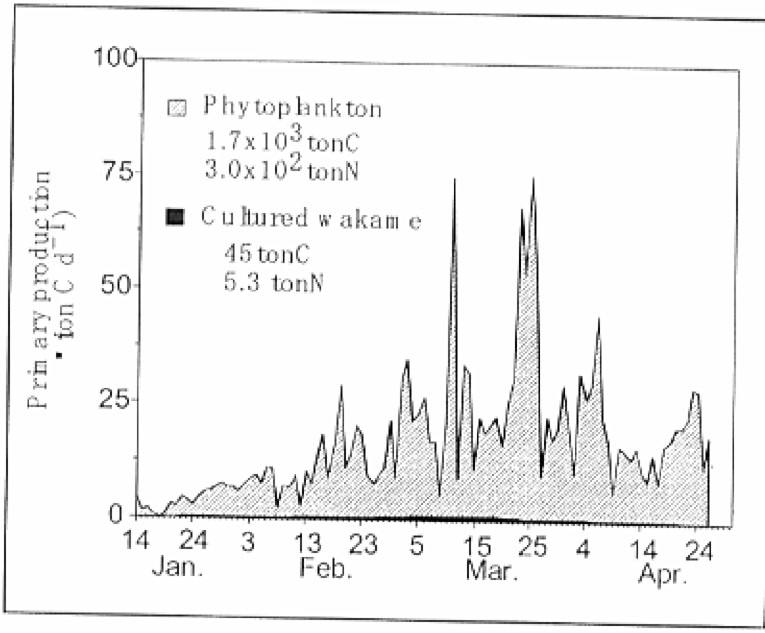


Primary Production

Primary production of both phytoplankton and the cultured kelp, *Undaria pinnatifida,* were compared as competitors for nutrients from mid-January to late April when a major portion of the annual primary production occurs in the bay. Considering the ease of instrumental handling and quick data availability, we applied the natural fluorescence technique and pulse amplitude fluorometry (Schreiber et al., 1994) to estimate primary production. Chlorophyll a and primary productivity were monitored using natural fluorescence and also estimated from the seawater samples. Bio-optical measurements were intensively conducted, and specific light absorption coefficient and quantum yield of phytoplankton photosynthesis were applied for conversion of natural fluorescence to chlorophyll a and primary productivity. Calculated chlorophyll a and primary productivity showed high correlation with the directly measured values (Furuya et al., 2002; Yoshikawa et al., in press). These time-series data on primary productivity were used with the integrated water column production derived from bio-optical spectral model to estimate primary production of the entire bay which was observed at 1.7×10^3 tC during the period of investigation (Fig. 2).

Cultured *U. pinnatifida* showed steady growth and primary production until March. Growth was much reduced in April and most part of the primary production was lost due to the removal of aged part of the thallus (Yoshikawa et al., 2001). Maximum biomass of 840 tonnes in wet weight (28.3 tC and 2.54 tN) was recorded in early March, just prior to the harvest. These figures decreased as the harvest

Figure 2. Primary Production of Phytoplankton and Cultured Kelp.



continued (Fig. 2). Production of cultured *U. pinnatifida* throughout Otsuchi Bay increased steadily in January and February, reaching its peak at 1055 kg C d^{-1} and 99.0 kg N d^{-1} in March. Erosion of the alga began in early March, and peaked at a rate of 469 kg C d^{-1} and 43.1 kg N d^{-1} in mid March. Erosion declined gradually as harvesting continued, and was comparable to production rates during April. Total biomass produced during the three-month observation period was 49.5 tC and 4.68 tN. The biomass harvested was 38.7 tC (81 percent of total) while losses due to erosion was 10.8 tC, which correspond to 19 percent of the total biomass produced. In terms of nitrogen, total losses due to erosion and harvesting were 1.56 and 3.12 tN, respectively. Averaged for the entire bay, total biomass produced during the observation period was 3.1 g C m^{-2}.

Phytoplankton primary production of 1.7 x 10^3 tC during the period of investigation consistently exceeded that of the cultured kelp (Fig. 2). Therefore, herbivorous consumption of phytoplankton by zooplankton and cultured shellfish is important in evaluating the carrying capacity of the bay.

Dissolved oxygen

Organic matter loadings and dissolved oxygen concentration in bottom waters were examined near the shellfish culture areas (Furuya, in prep.). Sinking flux of organic matter was collected using sediment traps under the culture rafts of scallops and oysters in spring (April and May 2000) and summer (July and August 2000). The sediment trap was cylindrical in shape, 14.5 cm in diameter and 50 cm in height. A honeycomb grid was installed at the opening of the trap to avoid disturbance of trapped materials and entry of large swimmers. The traps were deployed for two days and after their recovery, replaced by new ones. Trap samples were used for microscopic analysis and measurement of CN content. An *in situ* oxygen sensor (ADO8M, Alec Elec.) was fixed at 2 m above the bottom under a scallop culture raft. The sensor was calibrated once a week against DO measurement using the Winkler method. A good similarity in values obtained using both methods was consistently observed.

Sinking flux of organic matter, mainly composed of fecal matter was significantly higher under the oyster rafts than that of the scallops in both seasons ($p<0.05$). Flux under the oyster rafts was on the average 21.6 mgC m^{-2} d^{-1} for spring and summer, while that under the scallop rafts was 7.75 mgC m^{-2} d^{-1}. Mean flux outside the raft area was 5.75 mgC m^{-2} d^{-1} which is significantly lower than under the culture rafts ($p<0.01$).

Oxygen consumption rate of bottom seawater taken under the scallop rafts as determined by dark bottle incubation ranged from 0.26 to 3.07 mg L^{-1} d^{-1} with a mean of 1.49 mg L^{-1} d^{-1}. The mean rate implied rapid depletion of dissolved oxygen near the bottom in several days. However, *in situ* dissolved oxygen varying between 4.34 and 7.19 mg L^{-1} never got exhausted in summer. *In situ* continuous monitoring showed steady but slow decrease in dissolved oxygen in summer at a mean rate of 0.041 mg L^{-1} d^{-1}. This rate suggested that it took 160 days to produce anoxic water. These observations indicated that continuous supply of dissolved oxygen through water flow along the bottom was considerable and maintained the oxygen field in summer. Continuous monitoring of temperature, salinity, and dissolved oxygen showed frequent occurrences of inflow of subsurface water from outside the bay along the bottom. Wind-induced circulation, density current and internal tide were considered to be responsible for this inflow of outside water along the bottom (Otobe, in prep).

In contrast, anoxic water mass is formed near the bottom of Ofunato Bay which is located in the southern part of the Sanriku coast (Hayakawa, 1990). While the topography of this bay is similar to that of Otsuchi Bay, the presence of sill in the mouth area of the bay reduces water exchange, resulting in the formation of anoxic water, which is further compounded by the intensive culture of Japanese oysters. By comparing the water circulation and associated oxygen field near the bottom sediment between Otsuchi Bay and Ofunato Bay, it is clear that dissolved oxygen is an important factor for sustainable exploitation of biological productivity for aquaculture. Except Ofunato Bay, topography of other bays on the Sanriku coast indicates the occurrence of active water exchange. In general, these areas are therefore suitable for aquaculture from the point of view of oxygen supply.

Conclusion

Sanriku coast has an array of bays with topography similar to that of Otsuchi Bay. Except Ofunato Bay, an active inflow of subsurface water can be expected from the outside, which prevents the depletion of dissolved oxygen, indicating that the Sanriku area is suitable for shellfish culture. However, for sustainable exploitation of biological productivity in the area, evaluation of carrying capacity is crucial (Furuya, in prep), as the culture organisms compete with the natural populations for resources. As per the fishermen's experience, shellfish production of Otsuchi Bay is primarily limited by the food supply (Otsuchi Fishermen's Association, personal communication). Numerical physical-biological coupled models of material cycling within the local ecosystem of Otsuchi Bay have been developed (Kawamiya et al., 1995; Kishi et al., 2002). Kishi et al., (2002) incorporated the aquaculture activity of both wakame and shellfish. The continuous monitoring of flow field, chlorophyll *a*, primary production, and dissolved oxygen with high temporal resolution shown in this paper serves robust validation of the model performance and its subsequent improvement. Evaluation of carrying capacity of the bay is in progress using the model to understand the importance of primary production and dissolved oxygen in determining the carrying capacity of the bay and to identify other key factors. The model will be exploited to maximize aquaculture production of seaweed and shellfish with least environmental impacts in the bay.

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Estimation of Environmental Carrying Capacity for Aquaculture: A Case Study in the Republic of Korea

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CHOI, WOO JEUNG, LEE WON CHAN and KANG CHANG-KEUN. 2003. Estimation of environmental carrying capacity for aquaculture: a case study in the Republic of Korea, p. 60-77. *In* Huming Yu and Nancy Bermas (eds.) Determining environmental carrying capacity of coastal and marine areas: progress, constraints, and future options. PEMSEA Workshop Proceedings No. 11, 156 p.

Abstract

The ecosystem model (EUTROPII) based on physical factor and eutrophic processes in the coastal bay was modified to estimate the carrying capacity for oyster (*Crassostrea gigas*) culture system and applied to Goseong Bay, one of the most important oyster culture sites in Korea. Simulated values of oyster growth rate, chlorophyll *a*, and dissolved inorganic nitrogen concentrations were in agreement with the observed data. The model calculation demonstrated that intensive culture resulted in the reduction of oyster growth rate. Based on meat weight, oyster growth rate did not show significant difference between 12 individuals m^{-3} and 32 individuals m^{-3}. However, oyster growth rate clearly decreased in seeding density of more than 40 individuals m^{-3} and was very sluggish at seeding density of 300 individuals m^{-3}. We found that 16 individuals m^{-3} were reasonable seeding density to obtain a marketable size at the end of culture period. Based on these results, the optimum carrying capacity of Goseong Bay was estimated at 12,300 M/T with the assumption that oyster is cultured in the whole water volume. The available water surface area for oyster culture in Goseong Bay, however, is only 13 percent. Therefore, the carrying capacity of the bay could be re-estimated at 1,500 M/T meat weight.

Introduction

Mollusk culture particularly oyster, mussel, arkshell and fish culture are an important fishery industry in Korea. Oyster culture (*Crassotrea gigas*) in particular has a long history. The traditional oyster culture method used in the intertidal beds along the Korea coast has been replaced with suspended-culture since 1969. The use of better methods increased rapidly and maximized in mid-1980s. Oyster production began to decrease slowly which can be attributed to factors such as unstable settlement of spat, lack of available food in intensive culture (Yoo et al., 1980; Park et al., 1999), and eutrophication of the coastal areas. In some cases of suspended oyster culture grounds in the southern coastal bays of Korea, conditions for oyster culture have deteriorated from year to year. As a result, the culture period has been extended to obtain a marketable size. Estimating the environmental carrying capacity for oyster culture is therefore necessary to maintain sustainable production while protecting the environment from the impacts associated with aquaculture.

Carrying capacity is one of the basic concepts of population ecology. Errington (1934) first introduced the term "carrying capacity" from observed evidence on environmental regulation of the abundance of wild animal population under predation. According to Odum (1983), the system is said to have reached carrying capacity when bivalve biomass ceases to increase. Dame and Prins (1998) defined carrying capacity as the total bivalve biomass supported by a given ecosystem as a function of water residence time, primary production time, and bivalve clearance time. Carver and Mallet (1990), on the other hand, considered carrying capacity as the difference between food supply and food demand to the mussels, assuming maintenance of actual growth rate. Raillard and Ménesguen (1994) considered that it corresponds to the ability of the system to support shellfish production. Thus, carrying capacity is generally defined as the biomass of animals in a population that can be supported permanently by a given system (Krebs, 1978).

Oyster growth is strongly related to the physical, chemical and biological processes in the marine environment. For an evaluation of the carrying capacity of an aquaculture system, it is necessary to understand the hydrodynamic characteristics and biological features of coastal ecosystems. In addition to conducting filed surveys, the use of numerical models, which serve as powerful tools in understanding environmental processes is indispensable. In general, models used in the evaluation of carrying capacity of shellfish culture system can be classified into five categories: global model; empirical model; dynamic energy budget model; eco-physiology model; and ecosystem model (Héral, 1985; Van Haren and Kooijman, 1993 ; Raillard and Ménesguen, 1994; Grant et al., 1993; Bacher et al., 1998). Eco-physiology model can effectively estimate the growth of shellfish according to available food organisms in the aquaculture system. This model, however, does not consider the availability of food supply based on hydrodynamic characteristics, impacts of culture activities on the marine environment, regeneration of food within the shellfish system, and the special differences in biochemical and physical processes occurring in marine culture systems (Raillard and Ménesguen, 1994). Most of the ecosystem model, which has been used for environmental assessment based on carbon and nutrient cycle, does not consider the physiology of shellfish. (Kremer and Nixon, 1978; Nakata and Taguchi, 1982; Nakata et al., 1983a; Taguchi and Nakata, 1998).

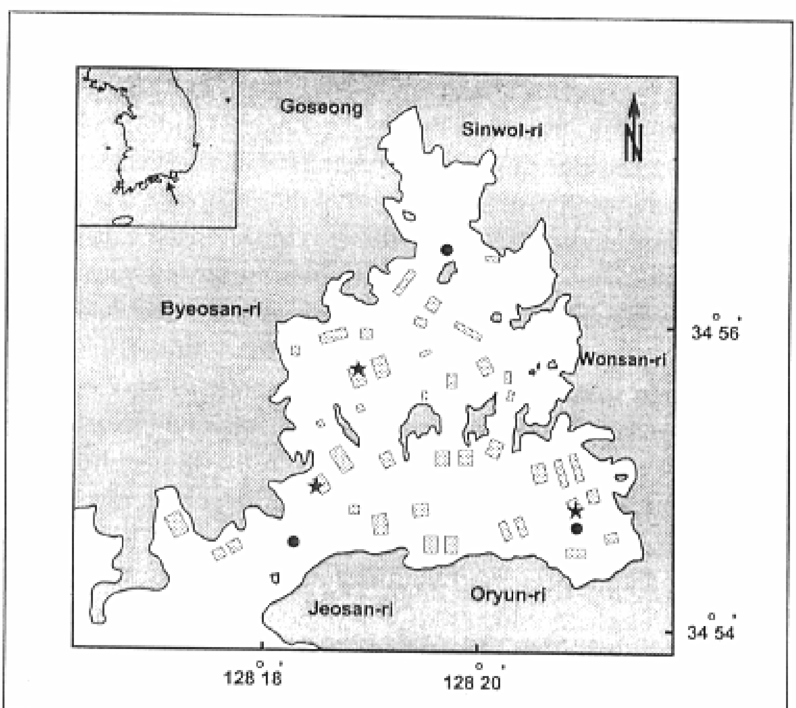
Shellfish physiology is essential to estimate the carrying capacity. Dame (1993) emphasized the coupling between the eco-physiology model and the ecosystem model to estimate the carrying capacity in aquaculture system. In this study, we developed an oyster growth sub-model and coupled it with EUTROPII model to evaluate the physical-biological interactions in the estuarine lower trophic ecosystem in terms of carbon, nitrogen, phosphorus and oxygen cycles (Nakata and Taguchi, 1982; Nakata et al., 1983a; 1985). The modified ecosystem model was applied to Goseong Bay to estimate the carrying capacity of the oyster culture system.

Materials and Methods

General description of the bay

Goseong Bay is a semi-closed bay (Fig. 1). Water exchange with the open sea occurs in the south through the narrow mouth. The area of the bay is 21 km^2 and its mean water depth is about six meters. The local hydrodynamic regime is regulated by tidal currents and the tidal range is up to about one meter. Pollutant loads from land-based sources originate from the northern part of the bay where a small town

Figure 1. Map Showing the Sampling Stations for Biological (★) and Environmental Samples from Goseong Bay, Republic of Korea.



Tetragons indicate oyster culture farms.

is located. Most of the basin consists of forestland. *C. gigas* is cultivated in the bay and annual oyster production reaches up to 1,300 M/T expressed as meat weight. Water and oyster samples were collected at three stations located in the bay for the calibration of the model during the period June 2000 to February 2001.

Description of the ecosystem model

Physical sub-model

A hydrodynamic model was developed by Nakata et al. (1983b; 1985), which includes time-dependent tidal forcing, surface wind and local density gradients together with actual coastline topography and bathymetry. Under hydrostatic and Boussinesq approximations on a rotating Cartesian coordinate system, the model employs the equations of fluid motion, flow continuity, conservation of heat and salt to determine the local distribution of prognostic variables. For the equation of state, Knudsen's expression is adopted (Taguchi et al., 1999).

Biological sub-model

The numerical model employed in this study was developed by Nakata et al. (1982; 1983a; 1985). The model was developed to evaluate the physical-biological interactions in the estuarine lower trophic ecosystem in terms of carbon, nitrogen, phosphorus and oxygen cycle. The model contains nine state variables called "compartment" consisting of four organic compartments expressed as carbon stock (phytoplankton, zooplankton, detritus and dissolved organic matter), four nutrients (phosphate, ammonium, nitrite and nitrate) and dissolved oxygen (Taguchi et al., 1999).

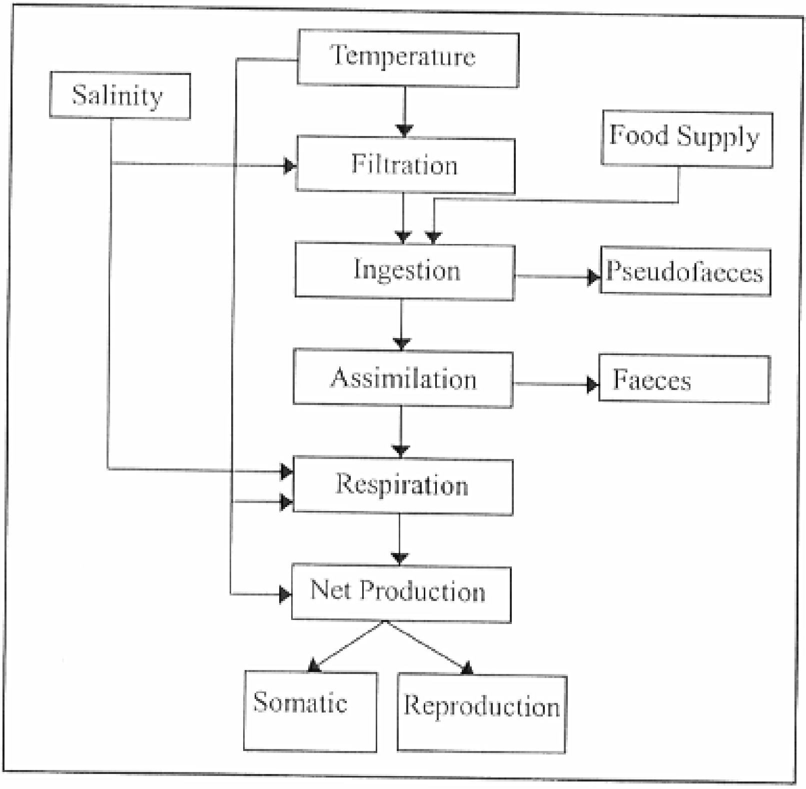
Oyster growth sub-model

Oyster growth model is based on the scope for growth (SFG), which is calculated as the net result of energy gained by feeding, and energy loss by maintenance (respiration and excretion) and reproduction (Fig. 2). Thus, the time-dependant oyster growth is the result of changes in net production (NP), which is assumed to be the difference between assimilation (A) and respiration (R), and a governing equation can be written as:

$$\frac{\partial O}{\partial t} = NP = A - R = I - B - R \qquad (1)$$

where O and B are oyster standing stock and biodeposits, respectively. Reproduction is not included in this model because culturing oysters are incapable of spawning during spawning season resulting to individuals weighing less than the spawning conditions defined by Powell et al. (1992).

Figure 2. Schematic of the Oyster Growth Model (after Hofmann et al., 1992).



Ingestion (*I*): Ingestion rate depends on the filtration rate and the ambient food concentration. One important component of the energy budget of bivalve is filtration rate. As part of this study, the filtration rate relationship for *C. gigas* was examined in order to determine a formulation for use in oyster growth model. Powell et al. (1992) adapted Doering and Oviatt's (1986) equation for filtration rate to oysters by using the biomass-length relationship of Hibbert (1977) to obtain filtration rate as a function of biomass (W) and temperature (T):

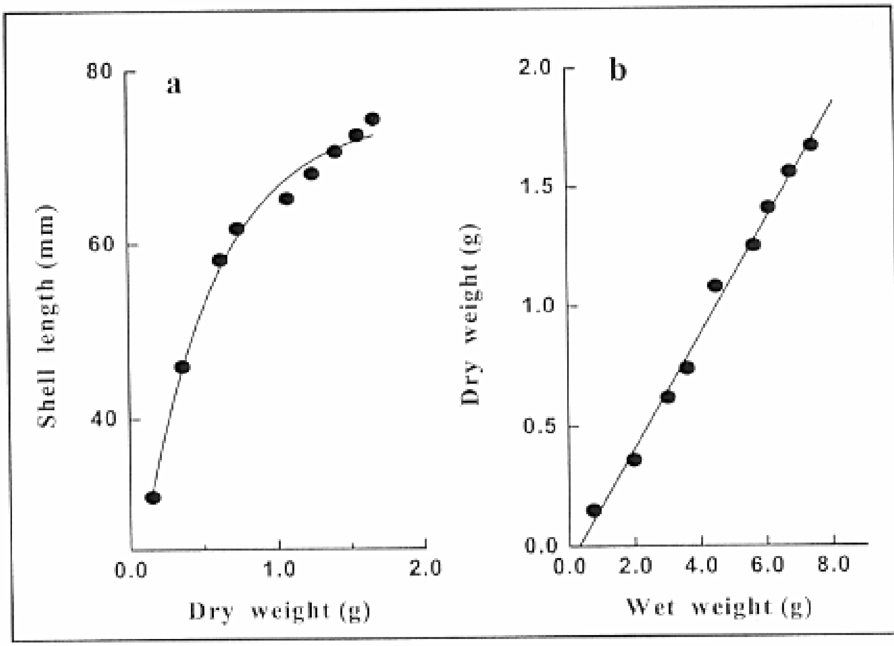
$$FR_d = \frac{SL^{0.96}\, T^{0.95}}{2.95} \qquad (2)$$

$$SL = W^{0.317}\, 10^{0.669} \qquad (3)$$

where FR_d is filtration rate (mL filtered per individual min^{-1}) , and SL is shell length (cm). However, the Equation 2 was based on the data for *Mercenaria mercenaria* rather than those for oysters (Hibbert, 1977). Therefore, a relationship between the mean shell length and the mean dry meat weight ($n=9$, $r=0.984$, $p<0.001$) for *C. gigas* was calculated as Figure 3a:

$$SL = 64.23\, W_d^{\,0.345} \qquad (4)$$

Figure 3. Relationships Between Dry Weight and Shell Length of Oysters (a), Between Wet Weight and Dry Weight of Oysters (b)



where Wd is dry meat weight in grams. A relationship between the mean wet weight and the dry meat weight ($n=9$, $r=0.996$, $p<0.001$) was derived as Figure 3b:

$$W_d = 0.237W_w - 0.069 \qquad (5)$$

To formulate filtration rate for *C. gigas*, combining Equations 2 and 4 gives a filtration relationship at 20°C of the form:

$$FR_w = 2.83\ W_d^{\ 0.331} \qquad (6)$$

where FR_w is filtration rate (L filtered per individual h^{-1}) and W_d is dry meat weight (gC). We consider the influence of temperature on filtration rate (Walne, 1972). *C. gigas* filtration rate was modeled as:

$$FR = \frac{FR_w T^{0.5}}{4.47} \qquad (7)$$

Under some circumstances, oysters reject part of the filtered material before ingestion and void it from the inhalent siphon in a less compact mass termed pseudofaeces (Haven and Morales-Alamo, 1972). Therefore, ingestion rate was expressed as:

$$I = FR \times P - PF \qquad (8)$$

. where, I is ingestion rate (mgC per individuals d^{-1}), FR is filtration rate (L per individuals d^{-1}), P is food quantity (μgC L^{-1}) and PF is pseudofaeces rejection rate (mgC per individuals d^{-1}).

Assimilation (A): Oysters excrete part of the ingested food before assimilation and evacuate it from the exhalent siphon as faeces. Therefore, the assimilation rate is calculated by subtracting the faeces production from ingestion rate:

$$A = I - F \qquad (9)$$

where, A is assimilation rate (mgC per individuals d^{-1}) and F is faeces excretion rate (mgC per individuals d^{-1}). Oysters also excrete inorganic nutrients to the water column and the excretion rate of nutrients is expressed as a function of biomass:

$$\text{Nutrient (N, P)} = E_{N,P} \times W_d \qquad (10)$$

where, $E_{N,P}$ is nutrient excretion rate (μM per individuals d^{-1}).

Respiration (R): Oyster respiration rate varies according to habitat factor such as individual size, water temperature, and salinity (Shumway, 1982). However, the rate does not depend on filtration rate variations that occur in response to the food quantity under normal physiological conditions (Clemmesen and Jørgensen, 1987). Raillard et al. (1993) generalized the relationship between respiration rate and dry meat weight for *C. gigas*:

$$R = (0.031T - 0.022)\, W_d^{-0.3} \qquad (11)$$

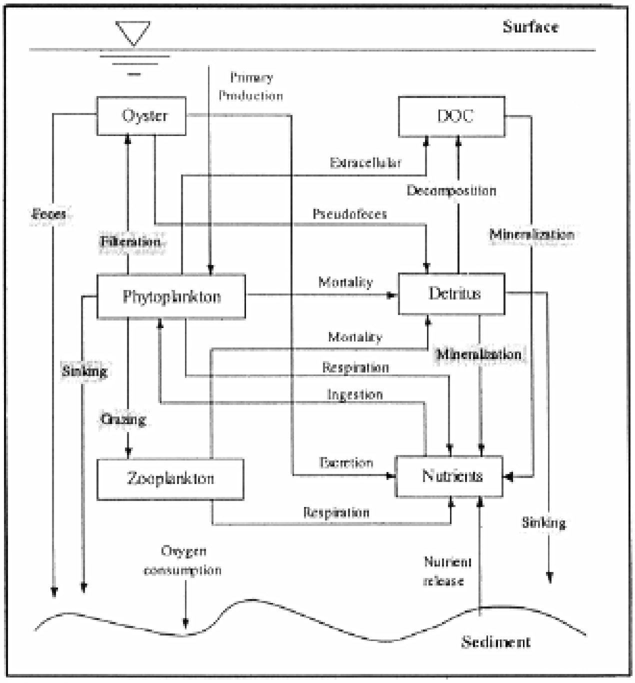
where, R is respiration rate (mL O_2 per individuals d^{-1}). In this study, salinity effect on respiration rate was neglected since the ambient salinity showed above 20 throughout the culture periods.

Coupling the physical, biological and oyster growth sub-model

Basic feature of the model

The modified ecosystem model was designed to simulate interactions between oyster growth and their environment including the physical and biochemical processes in the shellfish system. The treatment of major material fluxes in one box of the model is shown in Figure 4. This model is described by partial differential equations based on mass conservation of carbon, nitrogen, and phosphorus elements. In these equations, physical and biological processes of pelagic system are included, but the benthic variables are given as boundary conditions. Flow velocity is obtained from the calculation of the hydrodynamic model and is

Figure 4. Material Cycling in One Box of the Modified Ecosystem Model.



implemented to the modified ecosystem model. Biochemical reactions are formulated from temperature, intensity of daylight, concentration of dissolved oxygen, and concentration of compartments (C-N-P species) at every time step of calculation. The time and space dependency of these compartments are decided at each time step as a result of the interactions (physical and biological processes) among compartments or among compartments and environmental variables.

Model equations

The general equations of the coupled hydrodynamic, biological, and oyster growth model are as follows:

$$\frac{\partial C}{\partial t} = -(v \cdot \nabla)C + \nabla \cdot (K \cdot \nabla C) + \sum R$$

$$= -u\frac{\partial C}{\partial x} - v\frac{\partial C}{\partial y} - w\frac{\partial C}{\partial z} + \frac{\partial}{\partial x}(K_x\frac{\partial C}{\partial x}) + \frac{\partial}{\partial y}(K_y\frac{\partial C}{\partial y}) + \frac{\partial}{\partial z}(K_z\frac{\partial C}{\partial z}) + \sum R \qquad (12)$$

where C stands for concentration of compartments, v is the flow velocity that already has been calculated by the hydrodynamic model which includes the influence of wind, river-water, tidal force and local density, t is time (T), x, y, z are space coordinates (L), and ΣR stands for the biochemical reactions and fluxes from

Table 1. System of Differential Equations in the Model.

Oyster (mgC ind.$^{-1}$)

$$\frac{dO}{dt} = Pg + Pr = Ass - Res = Ing - Biode - Res$$

Phytoplankton (mgC m^{-3})

$$\frac{dP}{dt} = (Pgrowth - Pexu - Pmort - Pres) \cdot P - Zgraz \cdot Z - Oing \cdot O$$

Zooplankton (mgC m^{-3})

$$\frac{dZ}{dt} = (Zgraz - Zegest - Zexcr - Zmort) \cdot Z$$

Detritus (mgC m^{-3})

$$\frac{dDET}{dt} = Pmort \cdot P + (Zegest + Zmort) \cdot Z + Opf - Dremin \cdot DET$$

Dissolved matter (mgC m^{-3})

$$\frac{dDOC}{dt} = Pexu \cdot P + Dremin \cdot DET - DCremin \cdot DOC$$

Dissolved nutrient (mmol m^{-3})

$$\frac{dNUT}{dt} = (Pres - Pgrowth) \cdot P + Zexcr \cdot Z + Oexcr \cdot O + Dremin \cdot DET + DCremin \cdot DOC$$

Dissolved oxygen (mg L^{-1})

$$\frac{dDO}{dt} = Photo_P + Oair - Res_P - Res_Z - Res_O - Remin_{Org} - SOD$$

outside the system. This equation consists of the advection from water transport, diffusion from turbulent flow and biochemical reactions. Details of the system of differential equations and the functions used in the model are summarized in Tables 1 and 2.

Calculation method

The grid regime in the models was divided into 100 × 100 m in the horizontal direction and three levels in the vertical direction. Each of the three models needs boundary conditions and initial values. Boundary conditions, which are mainly used in this study, are M_2 derived from the harmonic analysis of the tide at sea level, concentration of compartments, salinity and water temperature, nutrients, freshwater discharge from river, and light intensity. Most of the data are interpolated using Spline method or linear interpolation. Initial values of compartments were set in June 2000. The simulation time covers the time-period from June 2000 to February 2001. Biochemical parameters employed in this model are shown in Table 3.

Table 2. **Equations of Modified Ecosystem Model.**

Quantity	Meaning	Formula
Pgrowth	Phytoplankton growth rate	$\mu max \cdot f(I) \cdot f(T) \cdot f(N,P)$
f(T)	Temperature effect	$e^{kt \cdot T}$
f(I)	Light effect on phytoplankton	$\int_0^{24}\int_{zmin}^{zmax} \frac{I_{z,t}}{I_{opt}} \cdot e^{\left(1-\frac{I_{z,t}}{I_{opt}}\right)} dzdt$
$I_{z,t}$	Light at depth z and time t	$I_{surf,t} \cdot e^{-k \cdot z}$
k	Light extinction coefficient	$k_0 + k_s[Chl.a : C] \cdot P$
f(N,P)	Nitrogen, Phosphorus limitation	$\min\left(\frac{N}{N+Kn}, \frac{P}{P+Kp}\right)$
Pmort	Phytoplankton mortality rate	$dp \cdot f(T)$
Pexu	Phytoplankton exudates rate	$\alpha \cdot e^{(-\beta [Chl.a:C] \cdot P)}$
Zgraz	Zooplankton grazing rate	$Rmax \cdot f(T) \cdot \{1 - e^{[\lambda(P^{*} - P)]}\}$
Zegest	Zooplankton egestion rate	$(1\text{-}e) \cdot Pgraz$
Zexcr	Zooplankton excretion rate	$(e\text{-}g) \cdot Pgraz$
Zmort	Zooplankton mortality rate	$dz \cdot f(T)$
Oing	Individual oyster ingestion rate	$F \cdot P \cdot (1\text{-}PF)$
F	Individual oyster filtration rate	$af \cdot W_d^{bf} \cdot f(T)$
PF	Proportion rejected as pseudofaeces	$PFp \cdot F \cdot P$
Oass	Oyster assimilation	$Oing - Obd$
Oexcr	Oyster nutrient excretion rate	$ae \cdot W_d^{bn}$
Dmin	Detritus mineralization rate	$MinDET \cdot f(T)$
DCmin	Dissolved organic mineralization rate	$MinDOC \cdot f(T)$
PP	Oxygen production by photosynthesis	$Rpp \cdot Pgrowth$
Rp	Phytoplankton respiration rate	$rp \cdot f(T)$
Rz	Zooplankton respiration rate	$rz \cdot f(T)$
Ro	Oyster respiration rate	$(art \cdot T + aro) \cdot W_d^{br}$

Results and Discussion

Reproducibility of ecosystem components by the model

Phytoplankton is considered as major food source of oysters because bivalves ingest phytoplankton preferentially among suspended organic matter (Prins et al., 1991). Zooplanktons, on the other hand, are considered as food competitors of oysters in addition to benthic filter feeders and fouling organisms attached to long lines in culture systems. Resuspension from the sediment was not considered in the model. Figure 5 shows the oyster growth rate, phytoplankton abundance (expressed as chlorophyll *a* concentration) and dissolved inorganic nitrogen concentrations for the culture period. The results agreed favorably with the observed data despite some uncertainties. Oyster growth increased gradually through time as the individual weight changed from 0.058 gC at the initial stage (June 2000)

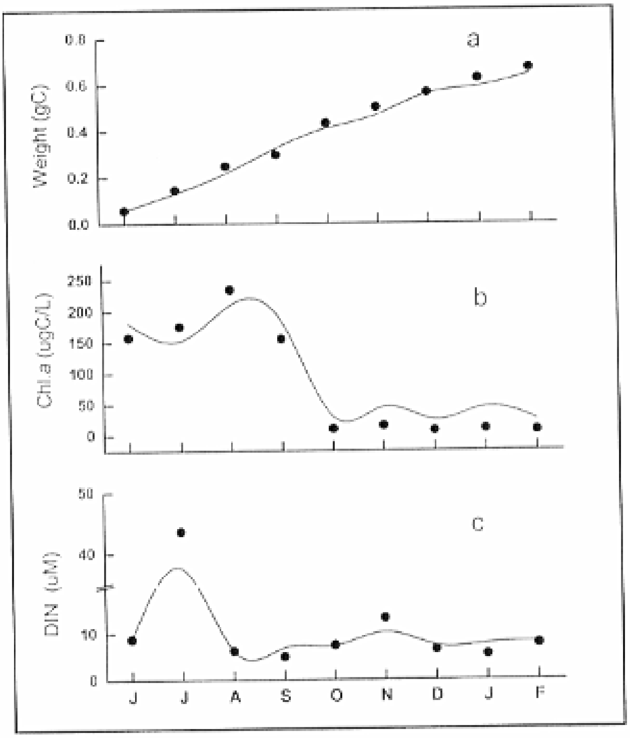
Table 3. Values of the Parameters Used in the Model.

Parameter	Definition	Value	Reference
Phytoplankton			
Mmax	Maximum growth rate	$0.6\ d^{-1}$	1
Kt	Temperature coefficient	0.0693 ? $^{-1}$	2
Iopt	Optimal light intensity	$200\ ly\ d^{-1}$	3
K	Extinction coefficient	$0.4\ m^{-1}$	OD
Kn	Half saturation constant for N limiting	$1.5\ mmol\ m^{-3}$	4
Kp	Half saturation constant for P limiting	$0.1\ mmol\ m^{-3}$	1
dp	Mortality rate	$0.015\ d^{-1}$	Tu
Rp	Respiration rate	$0.01\ d^{-1}$	5
Zooplankton			
Rmax	Maximum growth rate	$0.05\ d^{-1}$	1
A	Ivlev constant	$0.01\ m^3$? ^{-1}C	6
P*	Feeding threshold	75 ? $C\ m^{-3}$	7
E	Assimilation efficiency	0.7	8, 9
G	Growth efficiency	0.3	10
dz	Mortality rate	$0.02\ d^{-1}$	11
Oyster			
F	Filtration rate	Eq. (7)	12(OD)
PF	Pseudofaeces production	0.273· F	13
A	Assimilation efficiency	0.555	13
NP	Net production	0.205	13
art	Slope of respiration curve vs temperature	$0.031\ mLO_2\ h^{-1}\ ind.^{-1}$? $^{-1}$	14
aro	Intercept of respiration curve vs temperature	$-0.022\ mLO_2\ h^{-1}\ ind.^{-1}$	14
en	Excretion rate of N	$2\ \mu mol\ h^{-1}\ g^{-1} dry\ wt$	15
ep	Excretion rate of P	$0.57\ \mu mol\ h^{-1}\ g^{-1} dry\ wt$	16
ec	Carbon to dry weight	$0.4\ gC\ g^{-1} dry\ wt$	OD
Organic matter			
Dmin	Mineralisation rate of detritus	$0.08\ d^{-1}$	17
DCmin	Mineralisation rate of dissolved matter	$0.005\ d^{-1}$	18
Dbio	Fraction of biodegration of detritus	0.25	17
Wd	Settling velocity of detritus	$0.013\ m\ d^{-1}$	Tu

1. Jørgensen (1979), 2. Eppley (1972), 3. Ryther (1956), 4. Eppley et al. (1969), 5. Di Toro et al. (1971), 6. Frost (1972), 7. Steele (1974), 8. Marshall & Orr (1955a), 9. Marshall & Orr (1955 b), 10. Suschenya (1970), 11. Tetra Tech (1980), 12. Powell et al. (1992), 13. Kim (1980), 14. Raillard et al. (1993), 15. Boucher & Boucher-Rodoni (1988), 16. Asmus et al. (1990), 17. Ishikawa & Nishimura (1983), 18. Ogura (1975), OD. Observation data, Tu. Tuning.

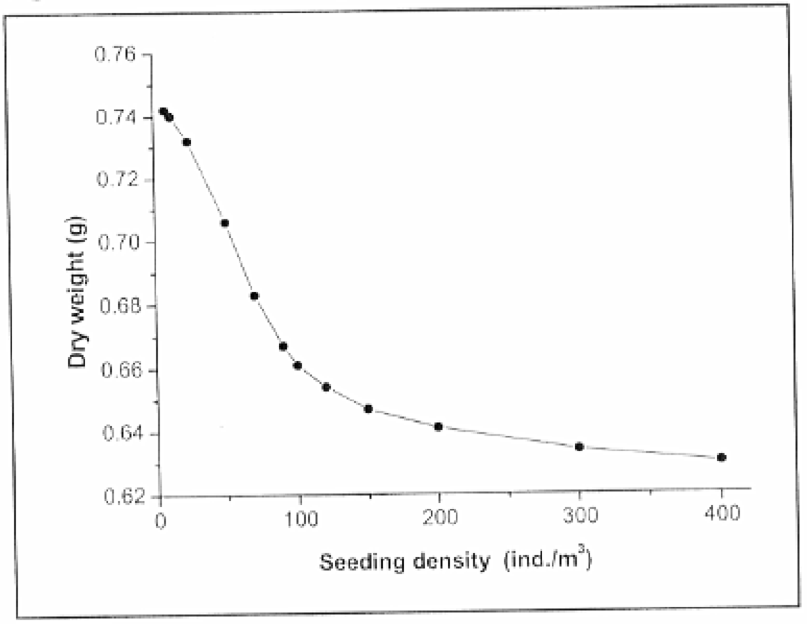
into 0.64 gC at final stages (February 2001). The annual variations of phytoplankton abundance seem to be well reproduced. The phytoplankton concentration in summer increased due to high temperature, high nutrient levels, and sufficient light, which together served to maximize the growth rate during this period. During

Figure 5. Comparison Between Simulated Result (solid lines) and Observed Ones (black circles).



(a) Individual Oyster Weight, (b) Phytoplankton, (c) Dissolved Inorganic Nitrogen.

Figure 6. Effect of Seeding Density on Oyster Growth.



winter, phytoplankton concentration was decreased due to poor environmental conditions. The dissolved inorganic nitrogen, which is a known limiting factor for phytoplankton growth, was simulated lowly in summer compared to observed values, while that in winter similar or a little low.

The effect of seeding density on oyster growth

The sensitivity analysis was performed through the estimation on the effects of seeding density to meat weight. The seeding density may be the most important biological component on the estimation of the economic feasibility of farm operation. The effect of oyster seeding density on individual oyster weight is shown in Figure 6. The running time for simulation was 30 days in summer. The oyster seeding density varied from one individual m^{-3} at the initial stage to 400 individuals m^{-3} at the final stage. The curve exhibited that individual meat weight decreased as the seeding density increased. Based on meat weight, oyster growth did not show clear difference between 12 individuals m^{-3} (0.74gC) and 32 individuals m^{-3} (0.73 gC). However, oyster growth decreased clearly in more than 40 individuals m^{-3} seeding densities and was very stagnant in 300 individuals m^{-3}. It means that the food field in aquaculture ecosystem altered in intensive seeding densities, and ultimately individual oyster growth rate will be reduced. Therefore, we have to consider the carrying capacity for sustainable and economical utilization of oyster culture farms.

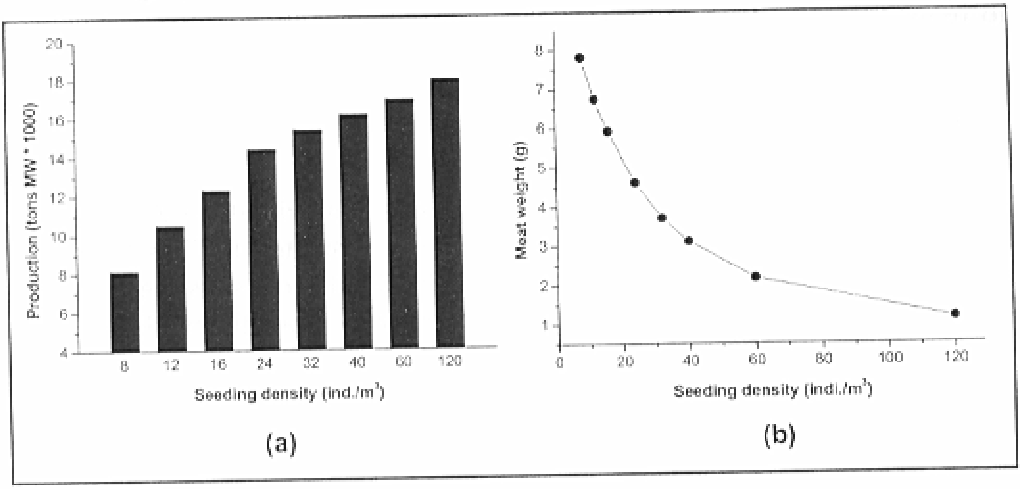
Carrying capacity

The carrying capacity of suspension feeding bivalves in coastal and estuarine ecosystem has implication for their culture as well as the structure and function of their ecosystem and is a fundamental concept in shellfish aquaculture for the sustainable production (Raillard and Ménesguen, 1994).

The carrying capacity for suspension feeders depends on the availability of space (substrate) and food. Available space is determined by hydro and geodynamic factors while food availability depends on primary production and transport processes (Smaal et al., 1998). According to a review by Heip et al. (1995), in many estuarine and coastal systems, physical factors constrain bivalve populations at the local scale while primary production limits them at the ecosystem level. Therefore, generic carrying capacity modeling should include both local- and ecosystem scale approaches (Smaal et al., 1998). Modified EUTROPII model can satisfy this requirement since it includes hydrodynamic, material cycle, and oyster growth processes in marine ecosystem.

According to Smaal et al. (1998), the carrying capacity of an ecosystem for natural populations can be defined as the standing stock of a population that can

Figure 7. Simulated Annual Production (a) and the Individual Growth (b) of Oysters at the End of Culture Period According to Seeding Density.



be supported by the ecosystem. Applying the concept of carrying capacity of an ecosystem in the context of exploitation of natural resources requires new definition on carrying capacity. Carrying capacity of an ecosystem for shellfish culture is the standing stock of the exploited population with annual maximum production as the marketable size (Carver and Mallet, 1990).

Figure 7a shows the simulated annual production of oyster at the end of culture period as a function of oyster density using modified EUTROPII model. Oyster production in Goseong Bay increased gradually from 3,000 M/T in eight individuals m^{-3} to 15,300 M/T in 32 m^{-3} expressed as meat weight. Production is not clear at densities of 40 m^{-3}, 60 m^{-3} and 120 m^{-3}. The individual wet meat weights at the harvest period as a function of oyster density are shown in Figure 7b. The result indicated that at higher culture densities, oyster growth was significantly affected, which means that intensive oyster culture has negative effects on individual oyster growth.

In Korea, oyster culture period is about nine months for meat wet weight to reach up to six grams individual^{-1}, which is considered as the marketable size. From the curve, 16 individuals m^{-3} were considered reasonable seeding density to obtain the desired marketable size. Based on these results, the optimum carrying capacity of Goseong Bay was estimated at 12,300 M/T with the assumption that oyster is cultured in the whole water volume. Since the available water surface area for oyster culture in Goseong Bay is only 13 percent, the carrying capacity of the bay could be re-estimated at 1,500 M/T meat weight. The present oyster production and seeding density are 1,300 M/T and 32 individuals m^{-3}, respectively. The present oyster production has almost reached the simulated carrying capacity with double seeding density. From these results, we could assume that total oyster production will not change significantly although seedling densities are reduced to 16 individuals m^{-3}.

Conclusion

The modified numerical ecosystem model was applied in the oyster farms of Goseong Bay. The numerical model experiment simulated the oyster growth, chlorophyll *a*, DIN, etc. in the bay very well. The simulated results suggested that 16 individuals m^{-3} were reasonable seeding density to obtain the marketable size (6 g meat weight) at the end of the culture period. The estimated carrying capacity of Goseung Bay was 1,500 M/T meat weight. This result was very close to the present oyster production.

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Environmental Carrying Capacity of an Oyster Culture in a Bay: A Case Study in Japan

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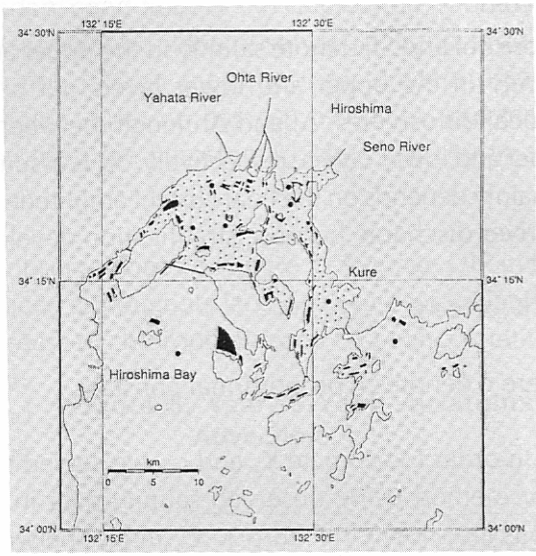
Abstract

Oyster culture has been prosperous in Hiroshima Bay, Japan but the productivity has gone down after 1990. The year-to-year variation in the lower trophic level ecosystem from 1984 to 1998 in the oyster culture particularly in August when the condition of the marine environment in Hiroshima Bay becomes worst was investigated using a numerical ecosystem model. The results of the numerical experiments in the northern part of Hiroshima Bay without oyster culture showed that the concentrations of chlorophyll-a, dissolved organic phosphorus (DOP), and detritus increased in the upper layer while dissolved oxygen (DO) concentration decreased in the lower layer. This means that oyster culture plays an important role in preserving the marine environment of Hiroshima Bay. The product of oyster culture was highest when the concentration of chlorophyll-a in the upper layer was 7.0 μg/l and the total phosphorus (TP) load from the Ohta River, which is the main river that drains into Hiroshima Bay, was 0.5 ton/day. It is necessary to keep the TP load from the Ohta River at 0.5 ton/day for sustainable oyster culture in Hiroshima Bay.

Introduction

Hiroshima Bay (Fig.1) is very famous for its prosperous oyster culture, which started about 300 years ago. However, oyster harvest has declined after 1990. In order to determine the possible causes of decline in oyster harvest, the year-to-year variation of lower trophic level ecosystem in Hiroshima Bay between 1984 to 1998 was investigated particularly in August, when the marine environment was at its worst condition due to high primary production in the upper layer sometimes triggering the occurrence of red and the development of oxygen deficiency in the bottom layer. In general, since oysters are suspension feeders, high productivity results in high oyster harvest. However, the high productivity in Hiroshima Bay in recent years has resulted in low oyster harvest. Some scientists claimed that the deterioration of the marine environment in the lower layer, which is a result of high productivity in the upper layer, induced the mortality in cultured oyster, thus resulting in low oyster harvest. In this study, we would want to understand the optimum environmental conditions in Hiroshima Bay that would result in high oyster harvest.

Figure 1. Observation Stations and Oyster Culture Field in Hiroshima Bay.



Legend:
Full Circles - Observation stations
Black Areas - Culture field
Dotted Areas - Area for the box model

Physical Model

In the northern part of Hiroshima Bay, an estuarine circulation is developed by the large river discharge from the Ohta River. A box model with upper and lower layers (Fig. 2), where the interface corresponds to the boundary between the euphotic and aphotic layers, was applied. The horizontal velocities were estimated in the upper and lower layers, U_u and U_l, and upwelling speed from the lower layer to the upper layer, W, by the continuity equation:

$$U_u A_u - R - AW = U_l A_l, \qquad (1)$$

where A_u and A_l denote the cross-sectional area of upper and lower layers, R river discharge and A the cross-sectional area of interface. The Ministry of Construction, Japan, observed the river discharge, R of the Ohta River every day. The horizontal velocity at the lower layer, U_l depends on river discharge, R from the past study of Yamamoto et al. (2000).

$$U_1 = 0.1 \times (4.85 \log_{10} R - 7.89) \qquad (2)$$

The conservation equations of salt are as follows:

$$- U_u S_{ui} A_u + W S_{li} A + K_h \frac{S_{uo} - S_{ui}}{\Delta l} + K_v \frac{S_{li} - S_{ui}}{\Delta h} = 0 \qquad (3)$$

$$U_l S_{lo} A_l - W S_{li} A + K_h \frac{S_{lo} - S_{li}}{\Delta l} - K_v \frac{S_{li} - S_{ui}}{\Delta h} = 0 \qquad (4)$$

where Sui and Sli denote salinity in the upper and lower layers of the box, S_{uo} and S_{lo} salinity in the upper and lower layers out of the box, K_h and K_v horizontal and vertical diffusivities, Δl and Δh length between the box and out of the box and the upper and lower layers, respectively. K_h is assumed to be constant every year because it mainly depends on the tidal current amplitude but K_v is assumed to depend inversely on river discharge because stratification develops when river discharge is large.

$$K_v = c / R \qquad (5)$$

where c denotes a proportional constant.

In order to estimate K_h and c , we solved equations 1 to 5 by the least-squared error method using observed salinity data in 22 years, from 1977 to 1998. The results are shown in Figure 3. U_u ranges between 0.01 and 1.0 cm/sec, U_l between 0.01 and 0.5 cm/sec, W between 0.1 and 4 x 10-4 cm/sec, K_v between 0.02 and 0.5 cm2/sec and K_h is 3.9 x 105 cm2/sec. Kv is small when U_u and U_l are large.

Year-to-year variation of average residence time of fresh water from Ohta River, T_f, is also calculated by the following equation:

$$T_f = (V_u + V_l)(S_o - S_l) / S_o R \qquad (6)$$

where V_u and V_l denote the volume of upper and lower layers. Average T_f is 32 days from Figure 3d.

Ecosystem Model

The ecosystem model used in this study is shown in Figure 4. Phosphorus cycling is considered here because the limiting nutrient in the northern part of Hiroshima Bay is phosphate (Lee et al., 1995). The governing equations on the biochemical processes of the ecosystem model were based on Kawamiya et al. (1995) and Yanagi et al. (1997) and the details including the parameters used were shown in Mitsushio et al. (2002). The average observed water temperature at six stations (shown by full circles in

Figure 2. Box Model Used in the Northern Part of Hiroshima Bay.

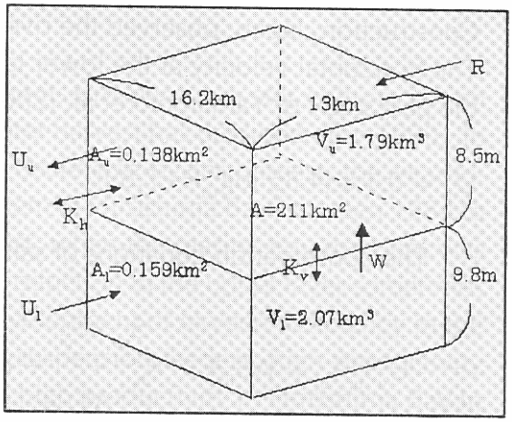
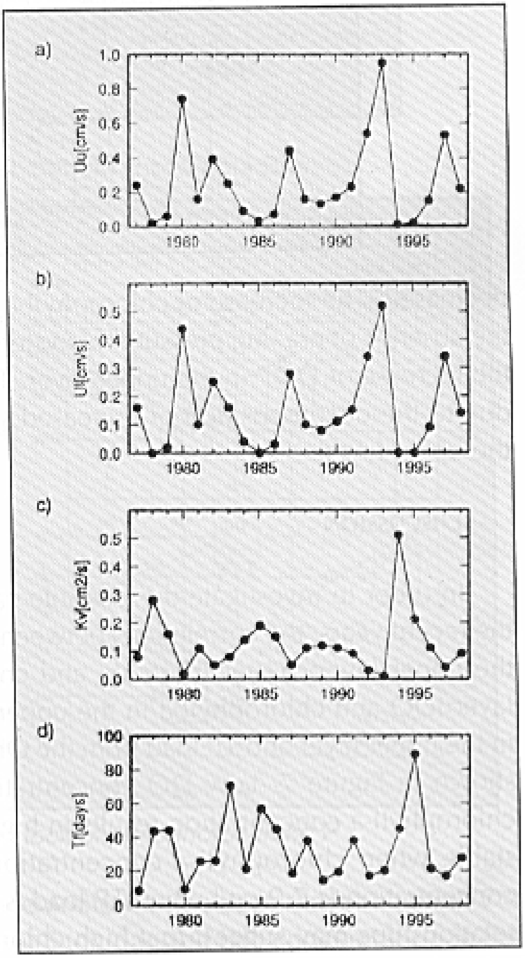


Fig.1) in the upper and lower layers of the box in August from 1984 to 1998 were used for the calculation, and average observed dissolved inorganic phosphorus (DIP), DOP, chlorophyll-a concentrations at two stations out of the box (shown by full circles in Fig.1) were used for the boundary conditions of the calculation. Moreover, the TP load from the Ohta River was estimated from the observed TP concentration and river discharge. Short wave radiation data at the Hiroshima Meteorological Observatory were used for the calculation. Data on harvest, mortality, and mean size of cultured oysters were obtained from the Hiroshima Fishermen Union and used for the calculation.

Calculated results are shown with observed ones in Figure 5. Average DIP in the upper and lower layers is well reproduced although its large year-to-year variability is not reproduced. The trend in the increase of chlorophyll-a and DO concentrations in the upper layer during the recent years are not reproduced but chlorophyll-a and DO concentrations in the lower layer are well reproduced.

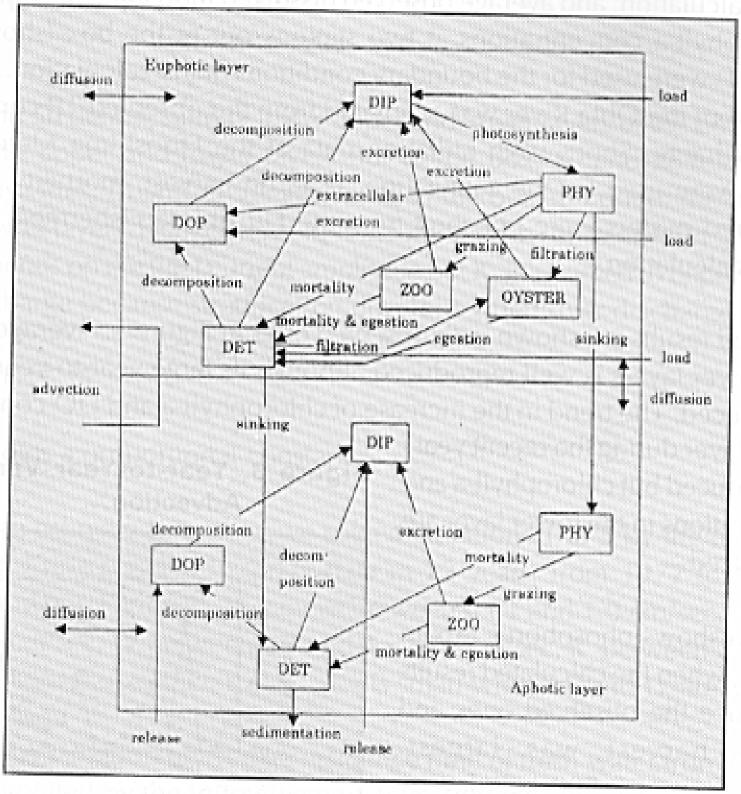
Figure 6a shows phosphorus flux in August 1987, when the calculated results well reproduce the observed ones and the oyster harvest was largest. Assimilation flux of 2,500 kgP/day by phytoplankton is about five times of the DIP load from the Ohta River and grazing flux of 1,450 kg/day by the oysters is about 58 per cent of primary production. The ratio of DIP sources to the euphotic layer from river: from the lower layer: from DOP decomposition: from POP decomposition: from zooplankton: from oyster is 1.0 : 2.6 : 1.7 : 0.6 : 0.3 : 0.2. DIP flux from the lower layer plays the most important role as DIP source for the euphotic layer. This is due to the development of an estuarine circulation in the northern part of Hiroshima Bay. Figure 6b shows the phosphorus flux in the case of no oyster culture under the same other conditions on August 1987 is shown in Figure 6b. Chlorophyll-a, DOP, detritus, and DO concentrations in the upper layer increased but DO concentration in the lower layer

Figure 3. Year-to-Year Variations in Advection.



Upper layer (a), lower layer (b), vertical diffusivity (c), and average residence time of freshwater (d).

Figure 4. Diagram of the Numerical Ecosystem Model.



decreased. The increase of chlorophyll-a and detritus in the upper layer is due to the absence of grazing pressure by oysters and the increase of detritus resulting in the increase of DOP in the upper layer. The decrease of DO in the lower layer is due to the decomposition of increased sinking phytoplankton and detritus from the upper layer.

Discussion

In order to investigate the possible causes of the decline in oyster harvest in the recent years, the correlations between oyster harvest and DO concentration in the upper layer; oyster mortality and chlorophyll-a concentration in the upper layer; DO and chlorophyll-a in the upper layer; and chlorophyll-a concentration in the upper layer and TP load from the Ohta River were examined. The results are shown in Figure 7. Low DO concentration results in high oyster harvest; high chlorophyll-a concentration results in high oyster mortality; DO concentration is stable when chlorophyll-a concentration is under 7.0 μg/l and chlorophyll-a concentration is 7.0 μg/l when TP load is 0.5 ton/day as shown in Figure 7. Such relationships may suggest that high chlorophyll-a and DO concentrations in the upper layer generate much detritus which sinks to the lower layer resulting in low DO concentration. This condition may have induced high oyster mortality resulting in low oyster harvest in Hiroshima Bay.

Figure 5. Comparison of Calculated and Observed DIP, Chl-a, and DO in the Upper and Lower Layers of the Box.

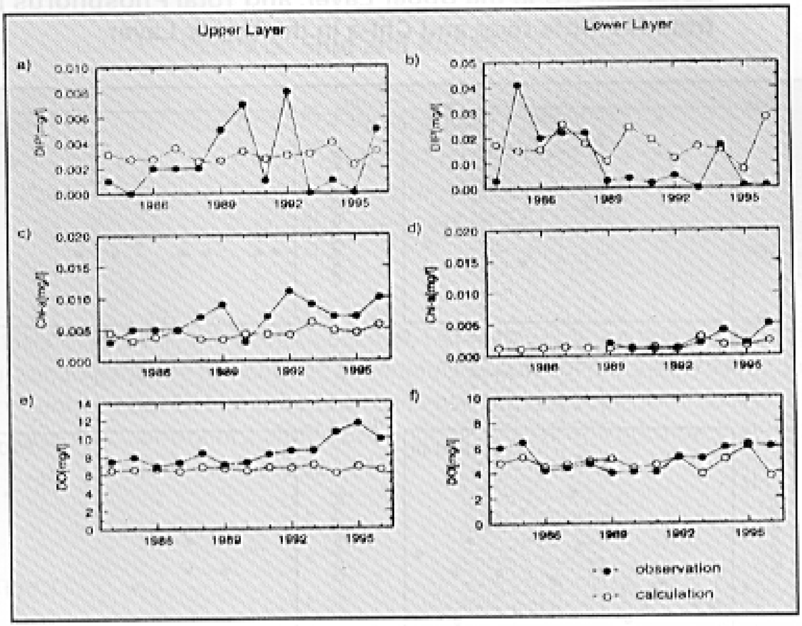
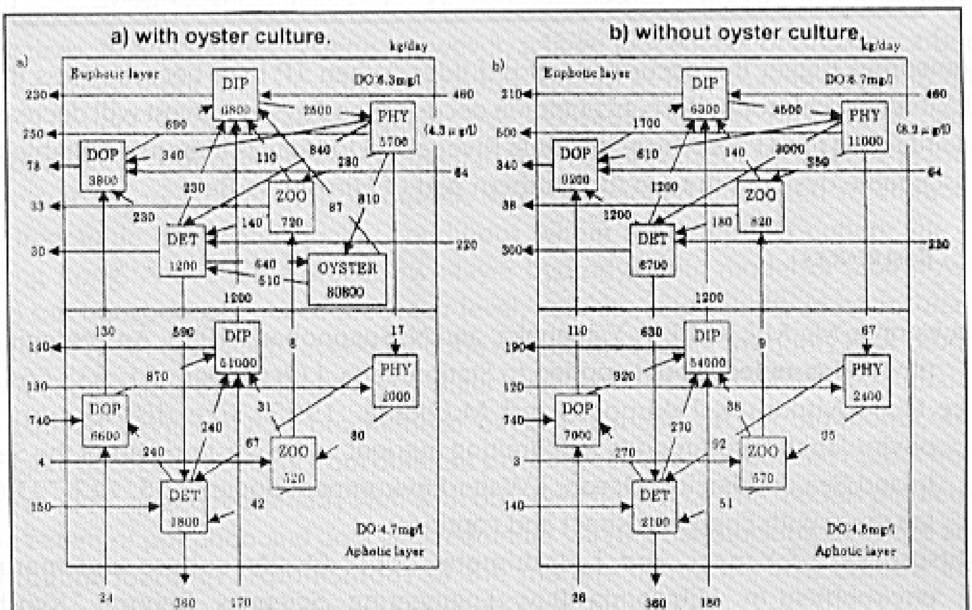


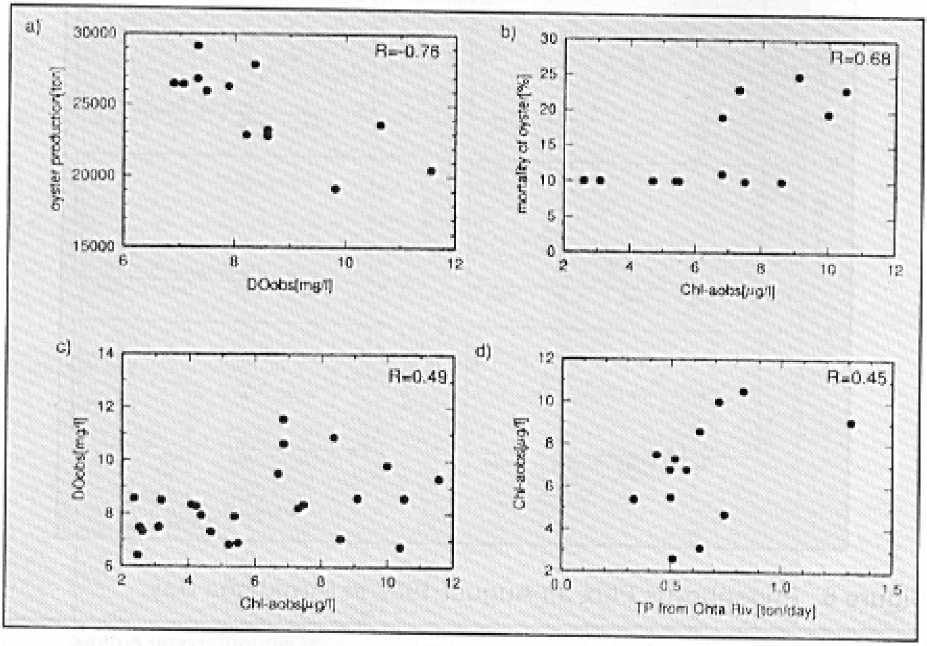
Figure 6. Phosphorus Flux on August 1987 in Hiroshima Bay.



Numbers in the box show the standing stock of phosphorus in kg P.

It is apparent that chlorophyll-a concentration in the upper layer is about 7.0 μg/l when TP load from the Ohta River is 0.5 ton/day and DO concentration in the upper layer becomes about 8.0 mg/l from Figure 7. Such condition results in low oyster mortality and high oyster harvest in Hiroshima Bay. When TP load exceeds 0.5 ton/day, chlorophyll-a concentration and oyster mortality increase and

Figure 7. Correlation Between DO in the Upper Layer and Oyster Harvest, Chl-*a* in the Upper Layer and Oyster Mortality, Chl-*a* in the Upper Layer and DO in the Upper Layer, and Total Phosphorus Load from the Ohta river and Chl-*a* in the Upper Layer.



sometimes trigger the occurrence of red tides. When TP load becomes less than 0.5 ton/day, chlorophyll-a concentration decreases and oyster harvest will decrease. Maintaining the TP load from the Ohta River at 0.5 ton/day is therefore suitable for sustainable oyster culture in the northern part of Hiroshima Bay.

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Estimating the Environmental Carrying Capacity for Sustainable Marine Fish Culture: A Modeling Approach

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Abstract

The environmental impact of marine fish farming depends on species cultured, culture method, stocking density, feed type, hydrography of the site, and husbandry practices. In all cultured systems, however, a large percentage of organic carbon and nutrient input into a marine fish culture system as feed may be lost into the environment through feed wastage, fish excretion, faeces production, and respiration. To estimate the environmental carrying capacity for sustainable marine fish culture, two deterministic models were developed and used to simulate hydrographic and water quality conditions within a marine fish culture site in Hong Kong. A two-dimensional, two-layer hydrodynamic model of tidal flows, and salt transport calculated the water level, velocity, and salinity in each square grid cell of 50 m in each layer within the culture area. Data from this flow model were input into a three-dimensional water quality model to simulate water quality [dissolved oxygen (DO), biochemical oxygen demand (BOD), ammonia, and organic nitrogen] resulting from different stocking densities. Simulated output from the models agreed reasonably well with observed field data. The models predicted the extent of pollution and area affected under varying fish stocking densities and pollutant loadings. Applications of the models in marine fish culture and environmental management are discussed.

Introduction

Marine fish culture has grown dramatically in recent years, and further growth is expected in the coming decade owing to the increase in demand of fish as a major food source in the world (New and Casvas, 1995). The rapid growth of the industry has already led to growing concerns over environmental impacts and conflicts with other coastal activities in Europe, North America, Australia, and Asia (Hammond, 1987; Waldichuck, 1987a, b; Morton, 1989; Miki et al., 1992). In particular, many studies clearly demonstrated the pollution effects of marine fish culture on water quality and sediments due to the release of organic and inorganic wastes (Hansen et al., 1990; Holmer and Kristensen, 1992; Handy and Poxton, 1993; Wu et al., 1994), therapeutic chemicals or antifoulants (Thrower and Short, 1991; Samuelsen et al., 1992), and physical obstruction and modification of water movement and sedimentation (Inoue, 1972; IDRC/SEAFDEC, 1979). Indeed, environmental concerns have resulted in tighter control measures and even moratorium on new marine fish culture developments in many countries/ areas such as New Zealand, Denmark, Norway, Canada, and Hong Kong (Duff, 1987; Wu, 1988, Morton, 1989; BC Ministry of Environment, 1990).

GESAMP (1991) introduced management guidelines to mitigate the adverse impacts caused by intensive fish farming practices. One of the proposed strategies is aimed at a sound utilization of the environmental capacity of the coastal waters, while producing aquatic products on a commercial basis. Wu (1995) emphasized, among other environmental control and improvement measures, the importance of maintaining stock density and pollution loadings below environmental capacity, in order to sustain future development of marine fish farming activities. The carrying capacity of the water depends on tidal flushing, current, and assimilative capacity of the water body to pollutants. A simple example is that of dissolved oxygen. Given that oxygen consumption of culture species ranges from 83 to >400 g O_2/t/h (Wu, 1990; McLean et al., 1993), and assuming that dissolved oxygen in seawater is 7 mg O_2/L, at least 17-57 m^3/h of fresh seawater would therefore be required to compensate for the oxygen consumption of 1 t of culture fish; let alone additional oxygen will be required to meet demand exerted by wastes from the farming activities. The above simple calculation demonstrates that the maximum carrying capacity (in terms of oxygen) of water with a flushing rate of 17-57 m^3 is less than 1 t of fish stock. Using the same approach and water quality modeling techniques, one can estimate the maximum permissible stock that may be held in a certain area to keep water/sediment quality within defined levels.

In this paper, we report the results of two simulation models which were used to simulate the effects of marine fish farming on ambient water quality in Hong Kong and demonstrate their applications to estimate environmental carrying capacity for fish culture operations.

The Study Site

The fish culture site, Three Fathoms Cove, is a semi-enclosed, shallow (ca. 5 m depth) embayment located inside Tolo Harbour at the northeast part of Hong Kong

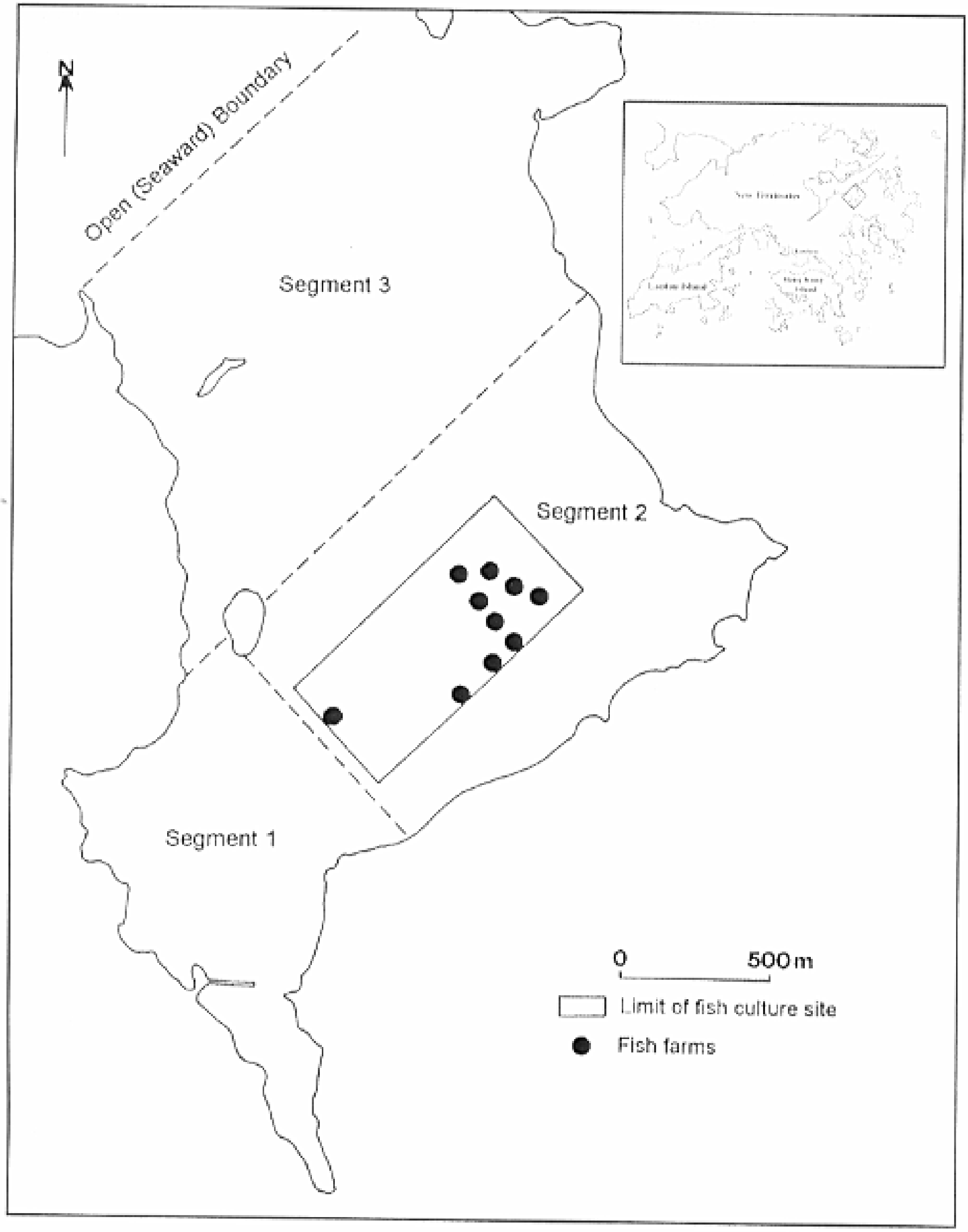
(Fig. 1). The site has a total sea area of 336 ha, with poor tidal flushing and a residual current speed of 0.01-0.02 m/s. Marine fish culture activities are confined to 34.2 ha within the inner part of the Cove, with rafts occupying 4.9 ha. Only 75 percent (3.7 ha) of the raft area is occupied by cages whereas the remaining 25 percent (1.2 ha) is reserved for holding or transferring fish during clean-up of fouling organisms on the culture cages. Each fish cage is approximately 3 m x 3 m x 3 m. The average fish stock density is 4.43 kg/m^3, with the culture species comprising 30 percent grouper, 45 percent sea bream, 15 percent snapper, and 10 percent other fish species.

Modeling

Two deterministic models, viz., a hydrodynamic model and a water quality model, were applied to the study site. The first model was a 2-D, 2-layer hydrodynamic model of tidal flow, and salt transport which calculated the water level, velocity, and salinity in each grid cell of 50 m^2 in each layer within the culture zone approximately 30 s time step during a tidal cycle. The model solved the equations describing conservation of mass and momentum and salt movement in two vertical layers of the water column and in two horizontal dimensions, where the input of the model (boundary conditions) consisted of the tidally driven water surface level along the model's boundary within Tolo Harbour (Fig. 1). The bathymetry of the study site was surveyed and salinity data were obtained from the routine monitoring program of the Environmental Protection Department (EPD), Hong Kong Special Administrative Region Government. Results from this flow model provided hydrodynamic data for input into the second 3-D (segmented and layered), tide-averaged water quality model, which was run to simulate water quality due to specific pollutant loadings from the marine fish culture operations. The model applied mass balance equations to quantify the relationships between the following twelve water quality parameters: slow dissolved carbonaceous BOD (biochemical oxygen demand), fast dissolved carbonaceous BOD, slow organic nitrogen, fast organic nitrogen, ammoniacal nitrogen, nitrate nitrogen, dissolved oxygen, salinity, temperature, suspended solids (inert particulates), fast particulate BOD, and slow particulate BOD. Fish biomass and the resulting pollutants (organic waste and nutrients) generated from various activities (e.g., food wastage, fish faeces and excreta) were quantified and input into the model. Based on the input organic and nutrient loadings from a given stocking density, the model calculated the resulting levels of ammonium nitrogen, total organic nitrogen, dissolved oxygen, and BOD in the receiving water. Details of the model equations, boundary conditions, parameter values, model time steps etc. can be referred to Wu et al. (1999).

Figure 1 also shows the schematization of the water quality model applied to the study site. Each model segment was divided into two elements vertically by a

Figure 1. Location of the Mariculture Site, Three Fathoms Cove, Hong Kong, and Schematization of the Water Quality Model.



horizontal interface. Model segment 2 represented the fish culture site. At the seaward boundary in segment 3, seasonal variations of water quality parameters for the model were obtained from field observations. In the model, the oxidation rate of organic matter (BOD and organic nitrogen) was assumed to comprise a slow and a fast component, the former being oxidized at a rate of one-fifth of the fast component (Ove Arup et al., 1989). To account for seasonal variations in water temperature, a sinusoidally varying temperature was prescribed in each element of the model, ranging from 18 °C in the winter to 29 °C in the summer.

Table 1. **Basic Data Assumed for the Water Quality Model.**

Parameter	Value	Data Source
Total raft area	49160 m^2	field observations
Total cage area	36870 m^2	field observations
Stocking density	4.43 kg/m^3	field observations
Total stock	490 t	field observations
Feed supplied	14.7 t/d (3% total stock in winter) 29.4 t/d (6% total stock in summer)	field observations field observations
Particulate feed lost	30% of feed supplied	Chu et al., 1995
BOD of particulates	82 g/kg feed	experiments
Soluble feed lost as		
BOD	38.2 g/kg feed/d	experiments
Ammoniacal nitrogen	0.5 g/kg feed/d	experiments
Organic nitrogen	1.1 g/kg feed/d	experiments
Oxidized nitrogen	0.1 g/kg feed/d	experiments
Suspended solids	18.2 g/kg feed/d	experiments
Faecal production		
Suspended solids	0.19 g/kg fish/d	experiments
Equivalent particulate BOD	0.22 g/kg fish/d	experiments
Ammonia excretion		
Ammoniacal nitrogen	0.32 g/kg fish/d	experiments
Urea production as		
Organic nitrogen	0.06 g/kg fish/d	experiments
Oxygen respiration	2.9-8.6 g O_2/kg fish/d	experiments; Lee et al., 1991

Note: Experiments refer to studies carried out by the authors.

Table 1 lists the basic data input for the water quality model, including major pollutants, viz., unconsumed food (trash fish), fish faeces and excretion products, generated by fish farming activities in the study area. For simulated nutrient inputs due to fish culture, the increase in organic nitrogen is attributable to a combination of the ammonium excreted by fish and the organic nitrogen leached from the trash fish (used as feed). In the absence of other data, the production of faeces was assumed constant (Ove Arup et al., 1989). Table 2 estimates the pollutant loadings generated from the fish culture activities for the wet (summer) and dry (winter) conditions, with doubling of food release (feed loads) to the fish stock in the summer. Predictions from the water quality model were further validated with field data collected in two surveys at the study site from November 1989 to January 1990. Details of these field studies were reported in Wu et al. (1994).

Table 2. Estimated Loadings of Marine Fish Culture Activities in Three Fathoms Cove, Hong Kong.

Source	Ultimate BOD (dissolved)	Ultimate BOD (particulate)	Organic Nitrogen	Ammoniacal Nitrogen	Nitrate	Suspended Solids
Summer Load (kg/d)						
Fish faeces		81.9	29.9	156.4		
Human waste	7.7	7.7	1.1	1.9		14.9
Food release	1122.0		32.4	14.7	2.9	535.5
Fish respiration	2245.6					
Food wastage		1059.2				2206.7
Nutrient release from sea bed			264.8			
Winter Load (kg/d)						
Fish faeces		81.9	29.9	156.4		
Human waste	7.7	7.7	1.1	1.9		14.9
Food release	561.0		16.2	7.3	1.4	267.7
Fish respiration	1032.9					
Food wastage		529.6				1103.3
Nutrient release from sea bed			132.4			

Results

Hydrodynamic Model Output

Water velocities generated from the model showed a magnitude of <0.04 m/s for both neap and spring tides in the dry season and increased to 0.08 m/s for neap tide and 0.11 m/s for spring tide in the wet season. Residual flows of <0.01 m/s largely showed an anti-clockwise movement out of the culture site near the northern shore in the dry season (Fig. 2). In the wet season, however, residual flows showed a clockwise direction out of the culture site near the western shore for the surface waters, with velocities up to 0.03 m/s, whereas the bottom waters mostly moved landward, with velocities <0.01 m/s.

Water Quality Model Output

Figures 3 to 5 show representative results of impact of mariculture activities on seasonal water quality from segment 2 of the water quality model output. In the surface waters, depletion of dissolved oxygen (DO) was minimal (within 0.5 mg/L); whereas in the bottom waters, the fish culture activities caused a drop of DO of about 1 mg/L from the base conditions (Fig. 3). An increase of organic nitrogen between 0.015 mg/L and 0.020 mg/L was also predicted in the surface waters and 0.025 mg/L in the bottom waters (Fig. 4). For ammonium nitrogen, the simulated concentrations in the surface waters varied between 0.015 mg/L in the summer

Figure 2. Simulated Spring Tide Residual Flows in the Mariculture Site.

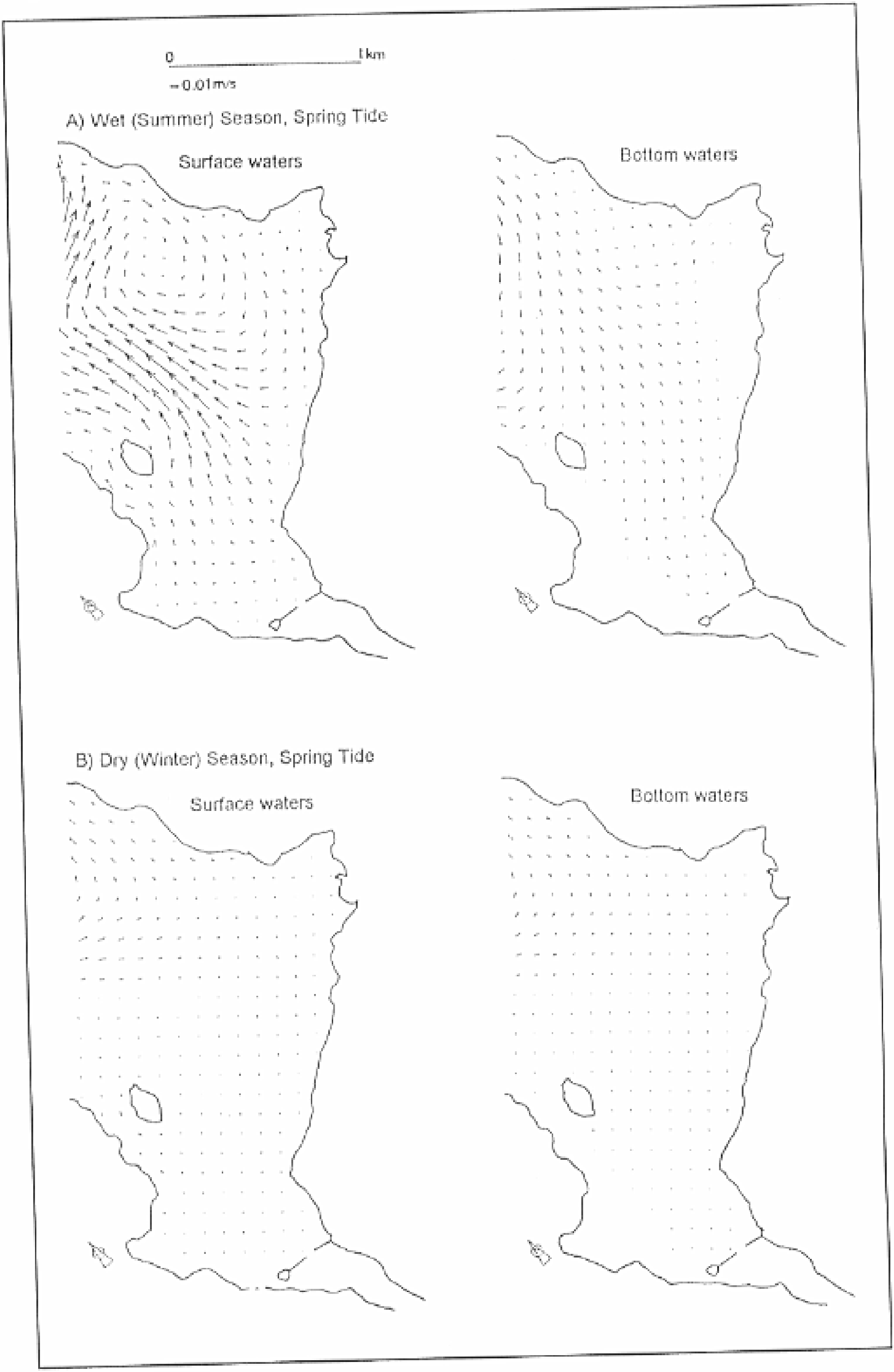


Figure 3. Effects of Fish Farming on Seasonal DO Variations in Model Segment 2 (Mariculture Site).

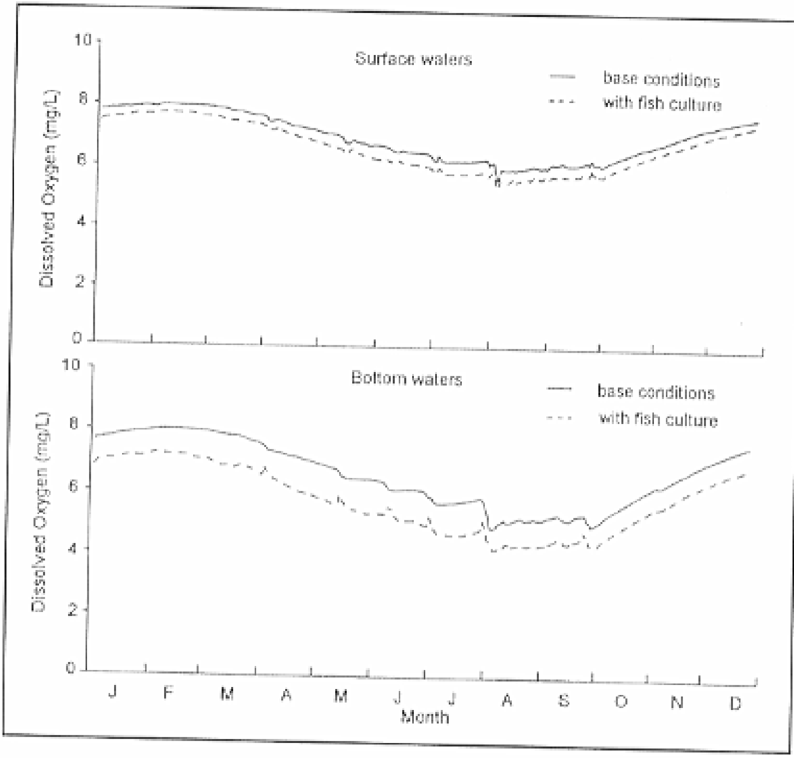


Figure 4. Effects of Fish Farming on Seasonal Organic Nitrogen Variations in Model Segment 2 (Mariculture Site).

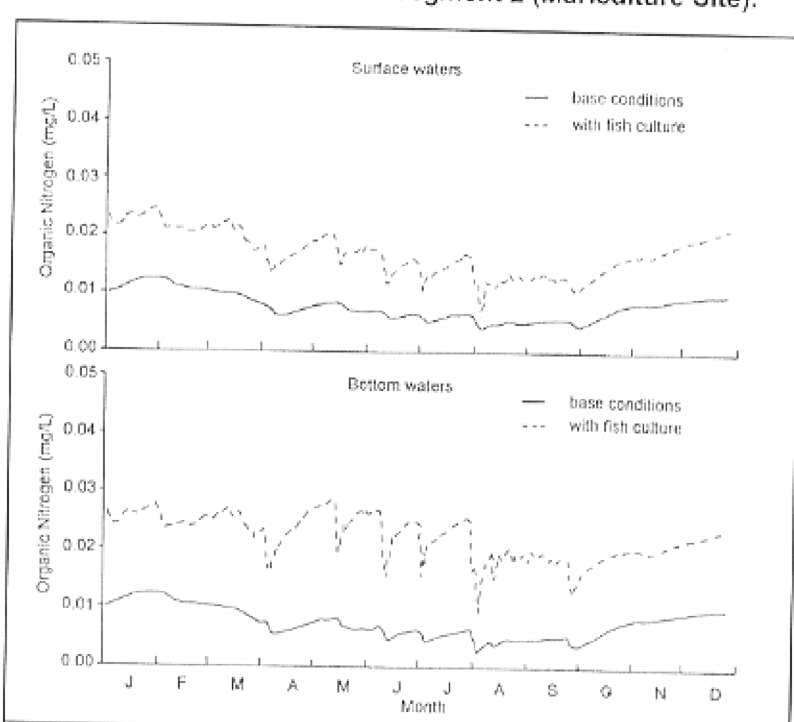
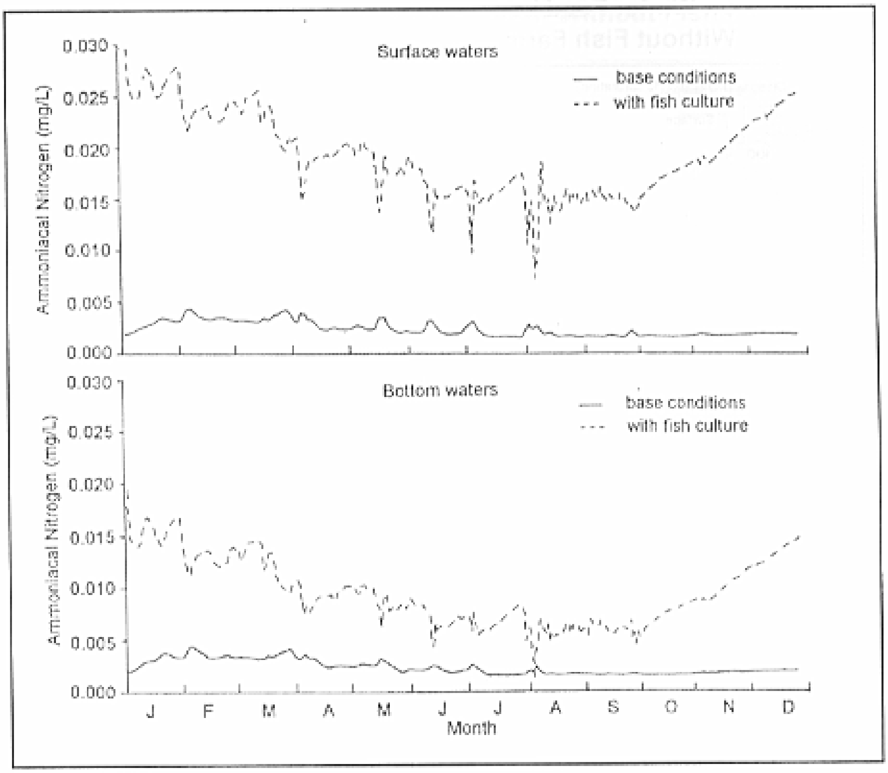


Figure 5. Effects of Fish Farming on Seasonal Ammoniacal Nitrogen Variations in Model Segment 2 (Mariculture Site).



and 0.030 mg/L in the winter compared with a base concentration of 0.003 mg/L; whereas in the bottom waters, increase in concentration of between 0.005 mg/L and 0.020 mg/L was predicted (Fig. 5). The higher predicted level of ammonium nitrogen near the surface could be due to ammonia excreted from the fish together with soluble feed lost.

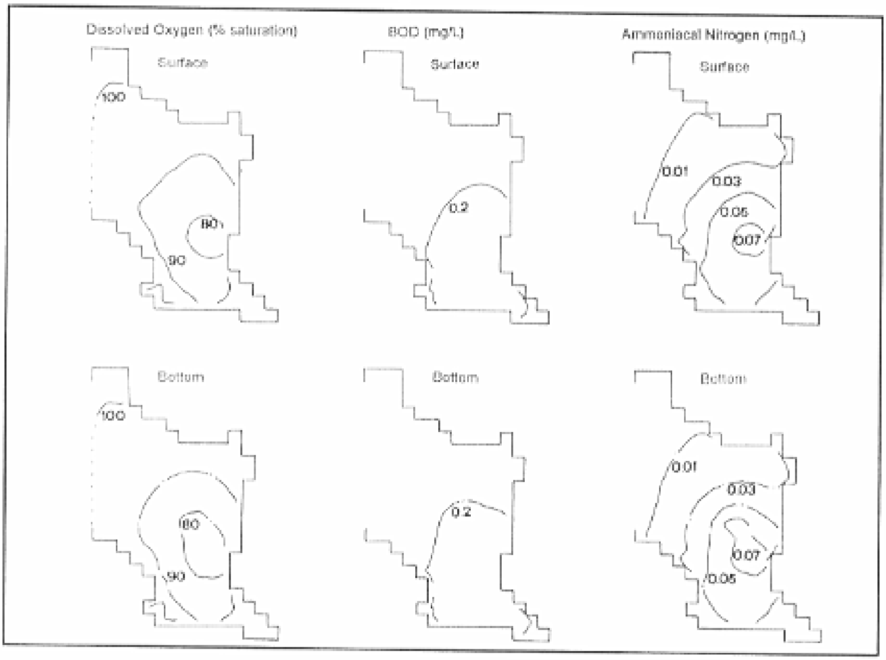
Estimating Carrying Capacity Using Water Quality Model

The effect of changes in feed loads to ambient water quality was modeled as an example to the applications of the model in estimating carrying capacity of the study site. Isolines were generated from grid cells from the 3 segments and compared with that obtained without fish farming. A comparison of Figures 6 and 7 demonstrates the effects of doubling the feed loads (see Table 2 for details on estimated pollutant loadings) on ambient water quality, revealing an increase in the areal extent of lower DO level and higher BOD and ammoniacal nitrogen concentrations in the water column.

Discussion

The results of the simulation are in close agreement with those from a field study of the same site reported upon earlier (Wu et al., 1994). In addition to revealing

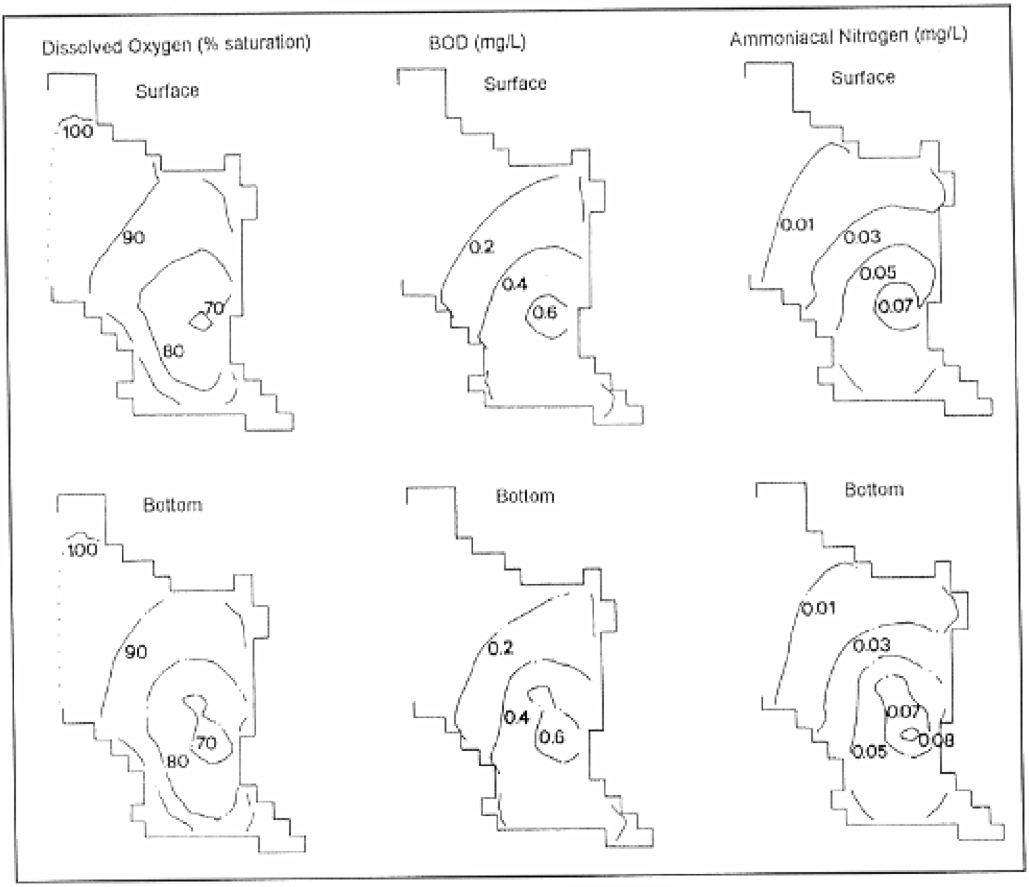
Figure 6. Simulated Neap Tide Spatial Variations of DO, BOD and Ammoniacal Nitrogen in Water at the Mariculture Site, Based on Winter Feed Load to the Fish Stock, as Compared to that Without Fish Farming.



changes in water quality around the mariculture site due to fish farming activities, the models demonstrate the effects of variation of feed loads on ambient water quality in the water column. The results thus illustrate the sensitivity of the water quality model to changes in input parameters. By using this set of hydrodynamic and water quality models, one can predict the upper level of fish stock (and hence the pollutant loadings) that can be maintained at a culture site without violating acceptable water quality objectives defined by the authority. Hence, by comparing the output of water quality data under different scenarios of stocking density, the models can serve as an effective tool to derive scientifically sound management decisions on the maximum fish stock permissible at a particular fish culture site so that acceptable water quality objectives can be met for the sustainable development of the industry. In Scotland, a suite of simple box models has been developed to provide a basis for assessing the impact of marine fish farming and regulating farming activities in sea loches (Gillibrand and Turrell, 1995).

Apart from assessing the impact caused by fish farming, the models may also be used to test the effects of changing feed types and culture species on water quality under the same growth stage and food ration regime. They can also serve as a useful planning tool for assessing the suitability of a proposed marine fish culture site prior to its operation, e.g., for environmental impact assessment. Traditionally, site selection in the marine fish farming industry has been based on such characteristics as proximity to land and degree of shelter. This has proven to

Figure 7. **Simulated Neap Tide Spatial Variations of DO, BOD and Ammoniacal Nitrogen in Water at the Mariculture Site, Based on Summer Feed Load to the Fish Stock, as Compared to that Without Fish Farming.**



be inadequate owing to the generally slow flushing conditions in sheltered areas that may cause deterioration of water quality (O'Connor et al., 1992). Deterministic mathematical models, such as the ones presented, may provide an effective tool in assisting fish farm managers to locate suitable culture sites in coastal areas and to ensure that mariculture production does not exceed the carrying capacity of the environment (Falconer and Hartnett, 1993; Beveridge, 1996).

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Estuarine and Coastal Water Quality Modeling: Concept and a Case Study in Korea

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Abstract

Water quality conditions and management in estuarine and coastal waters have received increased attention in recent years and water quality modeling has been used extensively as a scientific and managerial tool. This paper discusses the concept of numerical modeling of estuarine and coastal water quality, particularly eutrophication and hypoxia, and presents a case study in Korea. Water quality models, as they stand now, are site-specific, problem-dependent and data-dependent. Most mechanistic water quality models are based on mass-balance equations, consisting of physical transport and biogeochemical processes. Various aspects in the numerical modeling of both processes are discussed. The differences in the time and spatial scales of both processes are emphasized. Because of the large uncertainties in the simulation of biogeochemical processes, accurate simulation of physical transport processes, particularly the residual circulation and turbulent mixing, is critical for a reliable water quality modeling. A case study for Kwang-Yang Bay, a semi-enclosed bay system in southern Korea, is presented. With its results, some of the issues emphasized in the concept part have been revisited and the importance of a comprehensive data set for reasonable water quality modeling is discussed.

Introduction

Water quality conditions and management in estuarine and coastal waters have received increased attention in recent years as human activities in these areas increase. Since water quality problems such as eutrophication and hypoxia generally result from a combination of many processes including physical transport, biogeochemical transformations, and anthropogenic inputs, it is difficult to assess the relative importance of each process. To this end, a mathematical model based on physical and biogeochemical principles is useful to understand the prototype behavior and to provide consistent and rational tools for water quality management.

A water quality model package for estuarine and coastal waters, ideally speaking, may consist of four components: watershed basin model, hydrodynamic model, water quality model, and sediment process model (Fig. 1). Using meteorological inputs, land use, soil type, and other geophysical characteristics, a watershed basin model simulates flow rates and nutrient loads from point and nonpoint sources. Using this information along with surface forcing and open boundary conditions, a hydrodynamic model simulates water movements giving surface elevation, current velocity, and salinity (sometimes temperature). A water quality model, using the physical transport information from the hydrodynamic model, simulates the spatial and temporal distributions of biochemical pollutants. A sediment process model, upon receiving particulate pollutants settling from the water column, simulates the diagenetic processes in sediments and returns the fluxes of organic and inorganic materials back to the water column.

This paper discusses the concept of the numerical modeling of water quality, particularly eutrophication and hypoxia, in estuarine and coastal waters. Various aspects in water quality modeling related to hydrodynamic and water quality models are discussed. Watershed basin model is not explicitly discussed and only the importance of sediment process model is briefly mentioned. A case study for an enclosed bay system in Korea (Kwang-Yang Bay) is presented. With its results, some of the issues discussed in the "concept" part have been emphasized, particularly the importance of a comprehensive data set for a reasonable water quality modeling.

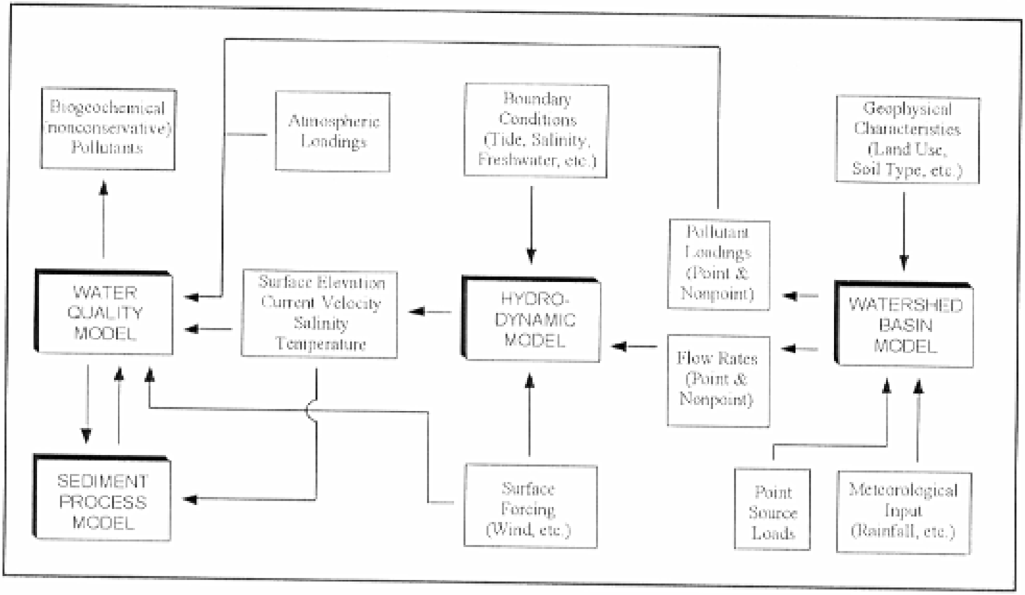
Concept of Water Quality Modeling

Most mechanistic water quality models calculate the spatial and temporal distributions of water quality state variables by solving mass-balance equations. Its general form may be expressed as:

$$\begin{aligned}\frac{\partial(V\cdot C)}{\partial t} &= [\text{physical transport processes}] + [\text{biogeochemical processes}] \\ &= [\text{advective transport} + \text{diffusive transport}] + [S_I + S_E]\end{aligned} \qquad (1)$$

where V = volume of any given portion of a water body; C = average concentration of a dissolved or suspended substance in the given volume; t =

Figure 1. An Ideal Water Quality Model Package for Estuarine and Coastal Waters.



time; S_I = internal sources and sinks; S_E = external sources and sinks. Equation 1 states that the time rate change of mass within the given volume is determined by physical transport and biogeochemical processes.

Physical transport processes in Equation 1, consisting of advective and diffusive transports, indicate mass transport by water movement. Physical transport processes take the same mathematical forms for all water quality state variables: i.e., physical transport processes do not depend on the nature of the substances as long as they do not affect the water movement. Their time scales can be intratidal (minutes to an hour) or intertidal (days) and the spatial scales can be zero-, one-, two- or three-dimensional depending on the characteristics of the system. Biogeochemical processes in Equation 1 consist of internal and external sources and sinks, the former being primarily due to the kinetic processes. Most of the kinetic processes in water quality models are mathematically represented by the empirical formulations based on observations, although some (e.g., Monod-type expression) may have theoretical backgrounds. Different water quality state variables are affected by different biogeochemical processes. The fastest kinetic process in eutrophication models usually is the growth of algae, which has the rate coefficient of about 2 day^{-1}, and thus the biogeochemical processes have time scales on the order of hours. Biogeochemical processes have no spatial scales except the settling of particulate matters. Materials introduced into a coastal system are transported in space by water movement. While being transported, their concentrations are modified by biogeochemical processes. The objective of a water quality model is to simulate these processes all together.

Modeling of physical transport processes

In estuarine and coastal waters, we usually obtain the information of physical transport processes from hydrodynamic models (Fig. 1). Depending on the characteristics of the system and problems of our concern, we may employ a complex three-dimensional intratidal model consisting of continuity, momentum, salt-balance, and heat-balance equations (Park et al., 1995) or a simple model such as an one-dimensional intertidal tidal prism model based on tidal flushing (Kuo and Park, 1995). Hence, modeling of physical transport processes for water quality modeling still tends to be site-specific and problem-dependent.

Water quality modeling has evolved in terms of modeling physical transport and biogeochemical processes. So far, advances in the modeling of physical transport processes far exceed those of biogeochemical processes. It is mainly because the mathematical expressions describing biogeochemical processes, compared to those for physical transport processes, are more coarse approximations of the properties of the system being modeled, and thus subject to less accuracy. Also, the field data that can be obtained from the current sampling techniques have finer spatial and temporal resolutions, and are of higher quality for the hydrodynamic parameters than the water quality ones. It makes the calibration and verification of hydrodynamic models much easier and more reliable than that of water quality models. Hence, because of the larger uncertainties in the modeling of biogeochemical processes in Equation 1, accurate simulation of physical transport processes is crucial for a reliable water quality modeling.

It should be noted that it is the residual circulation, not oscillating tidal current, that dominates the large-scale, long-term transport of pollutants in tidal systems (Dortch et al., 1992). In modeling the physical transport processes for water quality modeling, accurate simulation of residual circulation is critical but is not easy compared to that of tidal current (Park and Kuo, 1994). Another important aspect in hydrodynamic modeling for water quality conditions in estuarine and coastal waters is the vertical mixing. Because of the large vertical gradients exhibited by most of the important water quality variables such as algae and dissolved oxygen, and of the presence of vertical stratification that inhibits vertical mixing, accurate simulation of the vertical eddy diffusion coefficients is essential for a reliable water quality modeling, but is not easy at all (Blumberg, 1986).

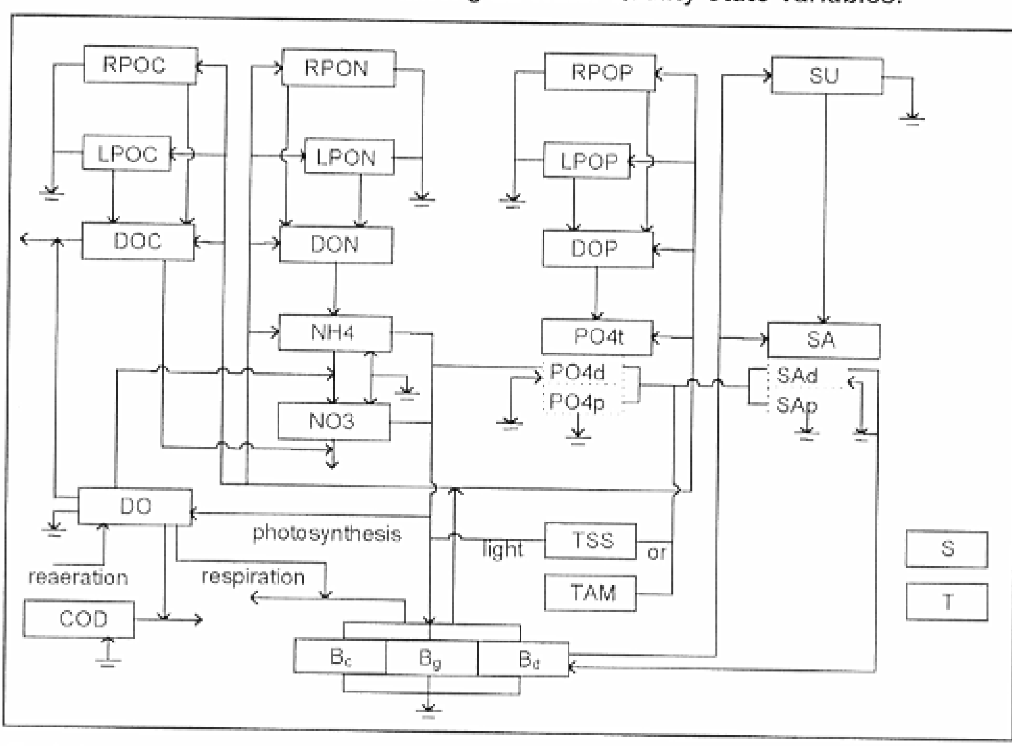
Accurate simulation of the spatial and temporal distributions of conservative substances such as salinity is a good evidence of demonstrating the model performance of overall physical transport processes including residual circulation and vertical mixing, and thus is essential in water quality modeling. Modeling of physical transport processes (i.e., hydrodynamic modeling) can be used as a framework for many applications including sediment transport models, ecosystem models, larval transport models and water quality models.

Modeling of biogeochemical processes

Modeling of biogeochemical processes has evolved in terms of water quality state variables and kinetic processes and formulations. Considerations regarding water quality state variables and kinetic processes in a model, mathematical representations (kinetic formulations), and estimation of kinetic coefficients in kinetic formulations largely depend on the characteristics of the system, problems of our concern, and data availability. Hence, modeling of biogeochemical processes is site-specific, problem-dependent, and data-dependent.

As our understanding of biogeochemical processes advances, a water quality model needs to be improved in terms of water quality state variables and kinetic formulations. Eight-state variable models have been extensively used to address eutrophication problems (Park et al., 1996). Recent eutrophication models, on the other hand, include increased number of water quality state variables and refined kinetic formulations. The eutrophication models in Cerco and Cole (1993) and Park et al. (1995), for example, include 22 state variables (Fig. 2) and highly nonlinear complex kinetic formulations. The more complex a kinetic formulation, the more kinetic coefficients it contains but the less variability each coefficient is subject to. A case study presented below employed the model in Park et al. (1995).

Figure 2. Kinetic Interactions among 22 Water Quality State Variables.



Each box represents a state variable and the arrows represent kinetic interactions among state variables.

The role of sediments is crucial for long-term variations in the water column water quality conditions and becomes more and more important as the water column water quality conditions improve (DiToro, 2001). DiToro and Fitzpatrick (1993) developed a sediment process model, which receives the depositional fluxes of particulate organic matter from a water column water quality model. It then simulates the diagenetic processes in the sediments, and returns sediment oxygen demand and benthic fluxes of inorganic materials back to the water column. Although an immense volume of data would be required for its application, the sediment process model has been successfully applied to several systems including the Chesapeake Bay (DiToro, 2001). The water quality models in Cerco and Cole (1993), Kuo and Park (1995), and Park et al. (1995) incorporate the sediment process model developed by DiToro and Fitzpatrick (1993) to simulate the long-term variations in the water quality conditions in the water column.

In the water quality models (Fig. 2) mentioned above, autotrophic algae is the highest trophic level that is explicitly simulated. The predation of algae by heterotrophic organisms such as zooplankton constitutes the pathway of energy flows through the food chain. A water quality model, to explicitly simulate the energy flows, needs to include all the trophic levels in food chain. It, however, is not yet feasible because our current understanding of the prototype processes, especially of the biogeochemical processes, is far from complete, and much of the complex interactions taking place in the food chain are not fully understood. A brief discussion of modeling of higher trophic levels such as zooplankton, submerged aquatic vegetation, heterotrophic bacteria, and fish is given in Park (1996).

Water quality modeling

Water quality modeling for estuarine and coastal waters requires information on physical transport processes, which is obtained by applying a hydrodynamic model appropriate for the system and problems of our concern. Water quality model needs to represent the state variables and kinetic formulations appropriate for the system and problems of our concern. Water quality modeling requires a comprehensive set of data for both physical transport and biogeochemical processes, including data for forcings, model calibration, and verification. Hence, water quality modeling is site-specific, problem-dependent and data-dependent.

Water quality modeling requires data for prescription of forcing functions, model calibration, and verification. Forcings for physical transport processes include open boundary conditions (e.g., surface elevation, salinity, and temperature), freshwater discharges, and wind stress. Examination of the model credibility for residual circulation, an important aspect in the modeling of physical transport for water quality variables, requires time-series data for surface elevation and current velocity.

Examination of the model credibility for vertical mixing, another important aspect in the modeling of physical transport, requires data for spatial distributions of conservative substances (usually salinity in estuarine and coastal waters). Since external loads of pollutants are the driving forces for the water quality model, data for external loads are critical for a reasonable water quality modeling. External loads include point and nonpoint source loads, atmospheric loads, benthic exchange rates, and fluxes through open boundary. Loading data for total materials, for example, total organic carbon, total nitrogen, and total phosphorus for eutrophication models, are required. Data for solar radiation, energy source for algal photosynthesis, are required. Also required for the modeling of biogeochemical processes are data for estimation of kinetic coefficients and data for model calibration and verification to examine the model credibility. It should be noted that simultaneous measurements of all data required are desirable.

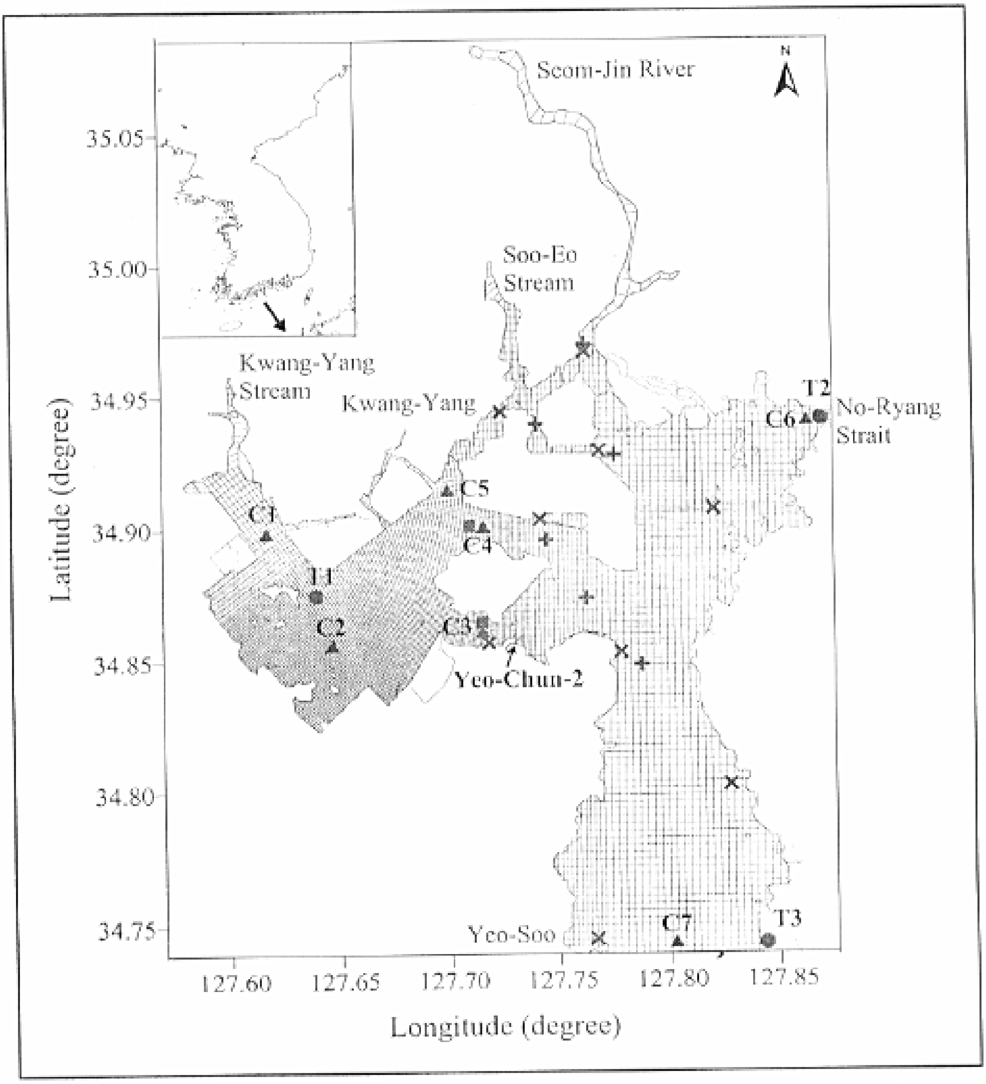
To understand their meanings and to properly solve them numerically, time and spatial scales of the processes in the mass-balance equation (Equation 1) are important. The time scales are more controlled by physical transport processes in intratidal models and by biogeochemical processes in intertidal models. The spatial scales are mainly controlled by physical transport processes. These differences in the time and spatial scales have several implications in water quality modeling (Park 1996). For example, Park et al. (1998) used the differences in the scales to develop a new solution scheme of Equation 1, which has better accuracy and computational efficiency than the traditional explicit or implicit schemes.

A water quality model, once reasonably calibrated and verified, can be a powerful tool for many applications. A water quality model may be used as a tool for process studies (e.g., testing of hypotheses), for quantitative analysis of large data set (e.g., interpretation of monitoring data set), and for water quality management [e.g., total maximum daily loads (TMDL) development]. It may also guide the direction of future research (e.g., identification of missing processes) and help design field surveys.

A Case Study in Korea

Kwang-Yang Bay, a semi-enclosed bay system in southern Korea, is connected to the south to the coastal sea (South Sea) and to the east to Jin-Joo Bay through narrow No-Ryang Strait (Fig. 3). Tidal ranges at the Bay mouth (station T3 in Fig. 3) are about 3.0 and 1.0 m during periods of spring and neap tides, respectively. The largest freshwater input is from Seom-Jin River with a median discharge rate of 42 $m^3 s^{-1}$, and Kwang-Yang and Soo-Eo streams are the next important freshwater inputs. Because of recent active development in Kwang-Yang Bay, the establishment of modeling framework as a management tool has been suggested.

Figure 1. Grid Map of Kwang-Yang Bay in Korea with the Insert Showing the Korean Peninsula.



× = 9 stations for tidal harmonic constants, + = 6 stations for tidal current harmonic constants, ● = 3 stations for time-series surface elevation, ■ = 2 stations for time-series current velocity, and ▲ = 7 stations for water quality.

Model description

A three-dimensional hydrodynamic-eutrophication model (HEM-3D) has been applied to Kwang-Yang Bay. HEM-3D, which has been listed as a tool for TMDL development (US EPA, 1997), consists of hydrodynamic and eutrophication models. Detailed description of HEM-3D can be found in Hamrick (1992) for the hydrodynamic model and in Park et al. (1995) for the water quality model. A brief description is given below.

The hydrodynamic model, often referred to as environmental fluid dynamics code (EFDC), is a real-time model based on continuity, momentum, salt-balance, and heat-balance equations (Hamrick, 1992). The model simulates density, topographically induced circulation, tidal and wind-driven flows, and spatial and temporal distributions of salinity and temperature. This model can handle features such as the wetting and drying of shallow areas and Lagrangian particle tracking. The information on physical transport processes, both advective and diffusive, simulated by the model is used to account for the transport of passive substances including non-conservative water quality state variables. The water quality model is based on mass-balance equations for 22 water quality state variables in the water column (Table 1 and Fig. 2) including suspended algae (three groups), dissolved oxygen, and cycles of carbon, phosphorus, nitrogen, and silica. The kinetic processes included in the model use highly nonlinear complex kinetic formulations, which are mostly from the Chesapeake Bay three-dimensional water quality model (Cerco and Cole, 1993). The model also incorporates the sediment process model developed by DiToro and Fitzpatrick (1993), that was not activated in the present study due to the relatively short modeling period (76 days from May 18 to July 31 in 2001).

Model application

The Kwang-Yang Bay area, located between latitudes 34°44'-35°05'N and longitudes 127°34'-127°54'E has been selected as the modeling domain (Fig. 3). An orthogonal curvilinear grid was used to resolve the complex shoreline, and highly varying bottom topography for the inner portion of the Bay, and narrow rivers while a Cartesian grid was used for the rest of the domain. A varying-size grid of 70-300 m was used and five sigma-layers were considered vertically. The number of total water cells at the surface layer is 9,002, including 994 intertidal cells.

Table 1. Twenty-two Water Column Water Quality State Variables in HEM-3D.

Salinity (S)	Temperature (T)
Three groups of algae (B_c, B_d and B_g)	
Refractory particulate organic carbon (RPOC)	Dissolved organic carbon (DOC)
Labile particulate organic carbon (LPOC)	
Refractory particulate organic phosphorus (RPOP)	Dissolved organic phosphorus (DOP)
Labile particulate organic phosphorus (LPOP)	Total phosphate (PO4t)
Refractory particulate organic nitrogen (RPON)	Dissolved organic nitrogen (DON)
Labile particulate organic nitrogen (LPON)	Ammonium nitrogen (NH4)
Nitrate (nitrate+nitrite nitrogen (NO3)	
Particulate biogenic silica (SU)	Available silica (SA)
Dissolved oxygen (DO)	Chemical oxygen demand (COD)
Total suspended solids (TSS) or total active metal (TAM)	

Detailed description of the model application is not presented in this paper. Instead, some of the model results appropriate to show the importance of accurate simulation of physical transport processes and of a comprehensive data set for biogeochemical modeling are presented.

Field program

A field program was conducted in May-July of 2001 to collect data for model application, of which only the measurements reported in this paper are described below. Time-series data for surface elevation were obtained using tide gauges at three stations (T1 to T3 in Fig. 3). Measurements at two boundary stations (T2 and T3) covered the entire modeling period, while measurement at station T1 lasted 36 days from May 16. Time-series data for the vertical profiles of current velocity were obtained from May 15 to June 14 using acoustic Doppler current profiler (ADCP) at two stations (C3 and C4 in Fig. 3). Four surveys in May 15-18, May 23-26, June 20-22, and July 29-31 were conducted to obtain the vertical profiles of salinity and temperature using conductivity, temperature, and depth profiler (CTD) at seven stations (C1 to C7 in Fig. 3), with hourly CTD casting over 13-hour period at each station. The reported values for daily freshwater discharge rates from Seom-Jin River were compiled and the daily discharge rates from Kwang-Yang and Soo-Eo streams were estimated based on the ratios of their drainage basin areas to that of Seom-Jin River. Daily wind data were compiled from the meteorological station at Yeo-Soo Airport.

In the four surveys for salinity and temperature, water quality parameters were measured from surface and bottom waters at the same seven stations, with two to three measurements taken over 6-hour period at each station for each parameter. Parameters measured are chlorophyll-a, particulate organic carbon, dissolved organic carbon, total particulate phosphorus, phosphate, total particulate nitrogen, ammonia, nitrate, nitrite, and dissolved silica. For the loads from rivers and streams, concentrations of the water quality parameters mentioned above were measured in the four surveys from surface waters at nine river stations. For the six small streams (not shown in Fig. 3) excluding Seom-Jin River and Kwang-Yang and Soo-Eo streams, freshwater discharge rates were also measured in the first and last surveys to estimate their loads. For 10 point source facilities, the reported values for discharge rates and the concentrations of five-day biochemical oxygen demand (BOD5), ammonia, nitrate, nitrite, and phosphate were compiled. Daily solar radiation data were compiled from the meteorological station at Yeo-Soo Airport.

Physical transport processes

The hydrodynamic model was calibrated with respect to bottom roughness height by simulating mean tide characteristics. The open boundary conditions were specified using the harmonic constants of five major constituents (M2, S2, N2, K1 and O1) in Tide Tables. The harmonic constants at nine stations (×'s in Fig. 3) were compared with the model results calculated with the bottom roughness height of 0.3 cm. Table 2 lists the absolute relative errors and mean errors averaged over nine tidal stations for the amplitude and phase of tidal constituents. Table 3 lists the errors averaged over six stations (+'s in Fig. 3) for the amplitude and phase of tidal current constituents. The results show that the present model application is capable of reproducing tidal dynamics for the amplitude and propagation (phase) of tidal waves and tidal currents throughout the modeling domain.

To verify the hydrodynamic model, a model run was conducted for a period of 76 days from 18 May to 31 July 2001. Open boundary conditions for surface elevation were specified with the observed time-series data at the stations T2 and T3, and all other boundary conditions were specified with the data as well. The model-data comparison for surface elevation at station T1 is shown in Figure 4 for both instantaneous and residual (a cut-off period of 48 h) components. The model calculated instantaneous and residual components of current velocity are compared with the data at the station C3 in Figure 5. The model is capable of reproducing

Table 2. Mean Tide Calibration Results for Tide.

Tidal Constituents		M_2	S_2	K_1	O_1	N_2
Amplitude	ARE (%)	1.6	1.9	2.4	3.7	2.7
	ME (cm)	-0.6	-0.6	-0.0	-0.5	-0.5
Phase	ME (deg.)	-0.1	0.7	0.7	0.5	-0.7

Absolute Relative Error (ARE) and Mean Error (ME) averaged over nine tidal stations.

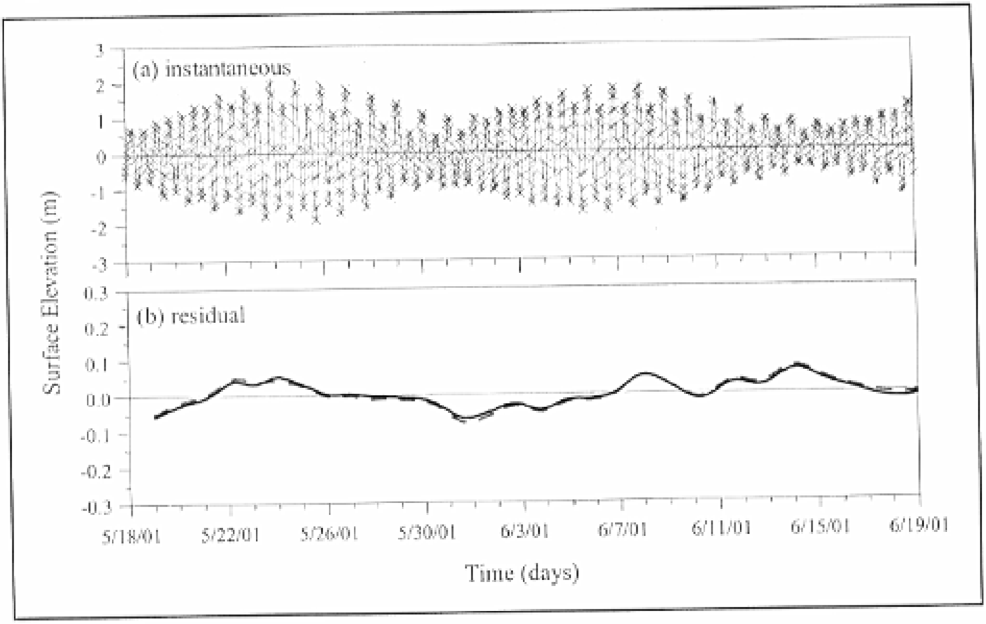
Table 2. Mean Tide Calibration Results for Tidal Current.

Tidal Current Constituents[1]			M_2	S_2	K_1	O_1
U-component	Amplitude	ARE (%)	32.5	11.9	27.0	34.7
		ME (cm sec^{-1})	1.1	-0.3	-0.7	-0.5
	Phase	ME (deg.)	5.5	10.5	-7.2	6.5
V-component	Amplitude	ARE (%)	17.4	8.8	19.6	63.2
		ME (cm sec^{-1})	3.8	1.0	0.1	0.1
	Phase	ME (deg.)	6.6	8.1	12.2	-0.6

Absolute Relative Error (ARE) and Mean Error (ME) averaged over six tidal stations.

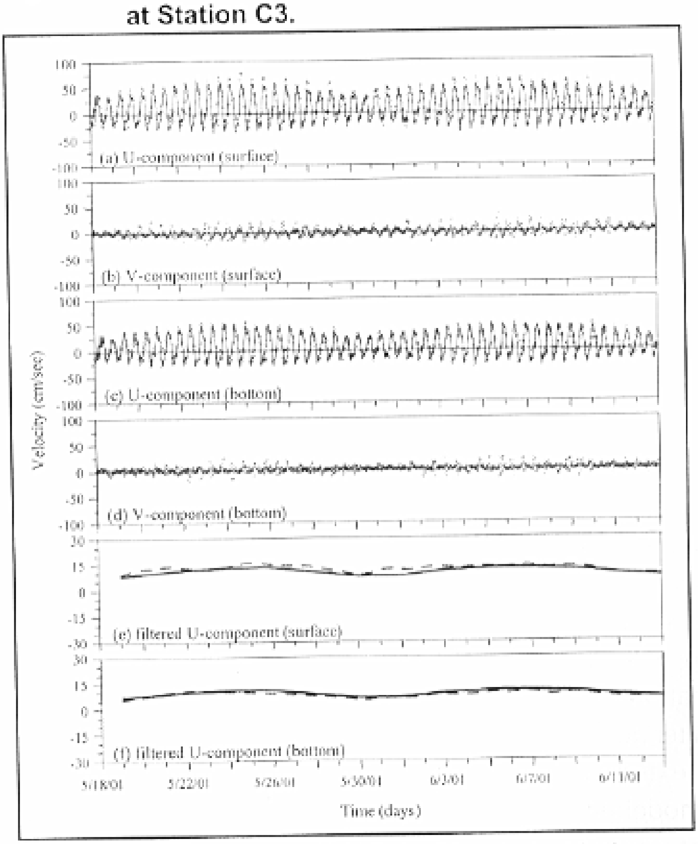
[1]Since the time-series data were only 15 days long, the N_2 component was not evaluated

Figure 4. Instantaneous and Residual Surface Elevation at Station T1.



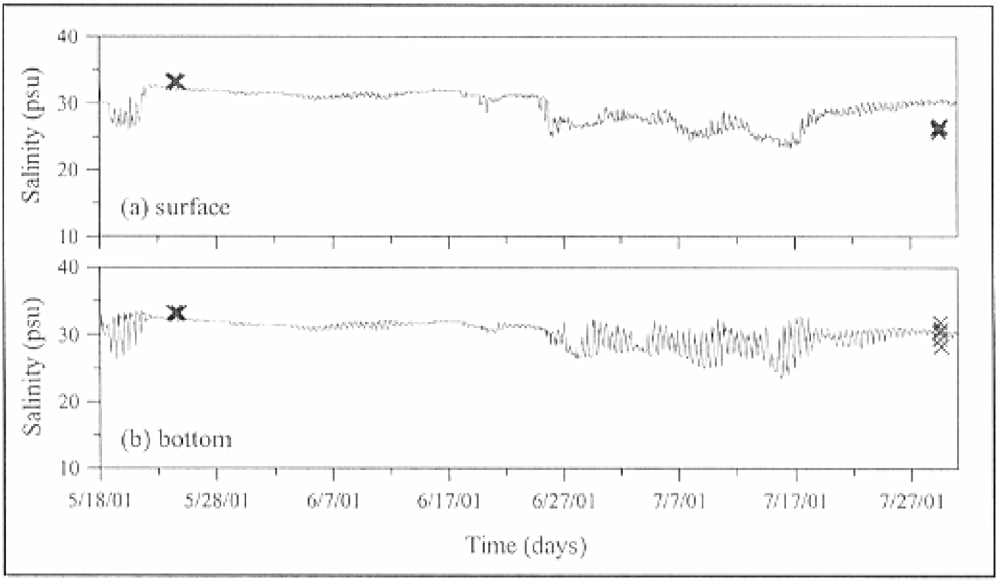
Model Results: (solid line); Field Data: (X or dashed line)

Figure 5. Instantaneous and Residual Current Velocity at Station C3.



Model Results: (solid line); Field Data: (X or dashed line)

Figure 6. Model-Data Comparison for Salinity at Station C3.



surface elevation and current velocity for instantaneous and residual components. The model also gives a reasonable reproduction of the observed salinity at station C3 (Figure 6). The present model application gives information on physical transport processes in a good agreement with the observations, which can be used for water quality modeling.

Biogeochemical processes

Data from the field program were used to estimate the external loads from 10 point source facilities and nine rivers and streams. For point source loads, however, only flow rates and BOD_5 were available for almost all facilities. For all facilities, effluent concentrations of organic carbon, nitrogen and phosphorus were not available. BOD_5 concentrations were converted to total organic carbon for use in the model using the empirical relationship developed for New York City municipal plants (HydroQual, 1991). Concentrations of organic nitrogen and organic phosphorus were estimated based on the default concentrations depending on the levels of treatment of each facility (Cerco and Cole, 1993). The loads from rivers and streams were estimated from a few measurements only (4 measurements for concentrations and 2 for discharge rates). Furthermore, since no data were available, nonpoint source loads along the shoreline were not considered. These uncertainties in the estimation of external loads limit the applicability of the present model results for water quality state variables. A comprehensive data set including detailed data for all external loads is essential for a reliable water quality modeling. Watershed modeling approach may be necessary to reasonably estimate the river and nonpoint source loads. Table 4 summarizes the loads estimated in the present

Table 4. Summary of Point Source and River Loads.

Loads		TOC (kg day^{-1})	TN (kg day^{-1})	TP (kg day^{-1})
Point Source Loads	Yeo-Chun-2	2,082	1,930	450
	Other 9 Facilities	1,572	1,852	393
	Total (10 facilities)	3,654	3,782	843
River Loads	Seom-Jin River	62,622	62,712	1,549
	Other 8 Rivers	5,933	4,068	100
	Total (9 rivers)	68,555	66,780	1,649
Total		72,209	70,562	2,492

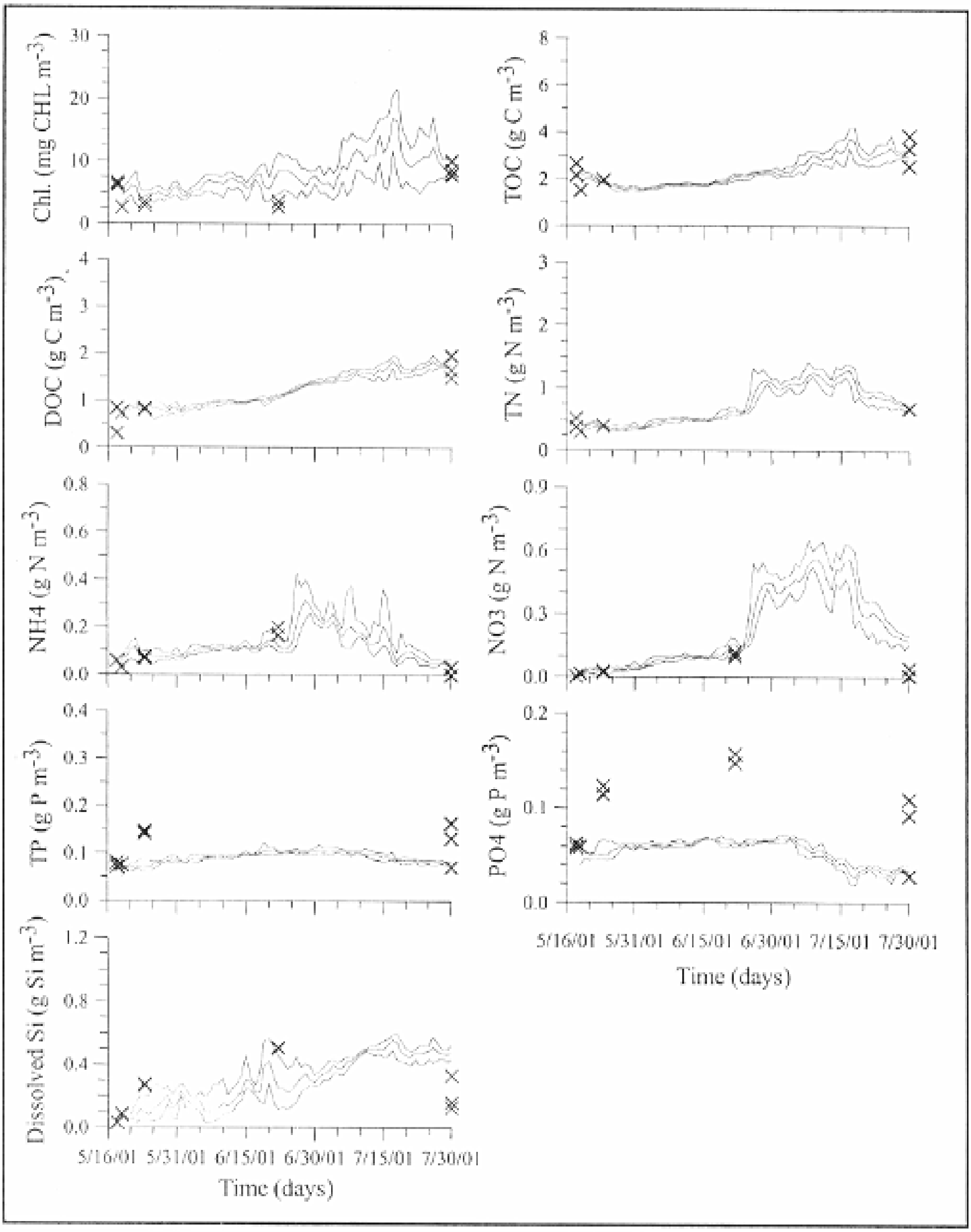
model application. Kwang-Yang Bay is dominated by river loads, which comprise 95 percent of total loads for carbon and nitrogen and 66 percent for phosphorus. Of river loads, Seom-Jin River is responsible for 91-94 percent of total river loads. Of point source loads, Yeo-Chun-2 facility (Fig. 3) is the largest one.

Figure 7 compares the model results with the field data for the surface water of the station C3 for chlorophyll-a, total organic carbon (TOC), dissolved organic carbon (DOC), total nitrogen (TN), ammonia nitrogen (NH4), nitrate + nitrite nitrogen (NO3), total phosphorus (TP), dissolved phosphate (PO4), and dissolved silica. The model gives a reasonably good reproduction of the data except phosphorus. The model underestimates phosphorus, particularly dissolved phosphate, which probably is attributable to the uncertainties in the estimation of the loads.

Summary and Conclusion

The concept of water quality modeling for estuarine and coastal waters is discussed and a case study using HEM-3D (Fig. 2) for Kwang-Yang Bay in Korea (Fig. 3) is presented to emphasize some of the issues discussed in the concept part. Most mechanistic water quality models are based on mass-balance equations consisting of physical transport and biogeochemical processes. The time scales in intratidal models are more controlled by physical transport processes, but by biogeochemical processes in intertidal models. The spatial scales are mainly controlled by physical transport processes. These differences in the time and spatial scales of both processes are important to understand their meanings, to properly solve them numerically, and to design field programs. So far, our understanding of biogeochemical processes is not as advanced as that of physical transport processes. Because of the larger uncertainties in the modeling of biogeochemical processes, accurate simulation of physical transport processes, particularly the residual circulation and turbulent mixing, is critical for a reliable water quality modeling (Figs. 4 to 6).

Figure 7. Daily Maximum, Mean and Minimum Model Results Compared with Field Data for the Surface Water of Station C3.



Model Results: (solid line); Field Data: (X)

Inasmuch as the numerical models of hydrodynamic and water quality conditions are dependent on our understanding of the prototype processes, the importance of data cannot be over-emphasized. A comprehensive data set including simultaneous measurements of hydrodynamic and water quality parameters and external loads are essential for a reliable water quality modeling. The water quality model results in Figure 7 could be substantially improved, particularly for phosphorus, if more data were available for external loads, model calibration, and verification.

A general water quality model applicable to any system and to any problem is not available yet. Characteristics of the system to be modeled, problems of our concern, and data availability affect the modeling of physical transport in terms of the spatial (zero-, one-, two- or three-dimensional) and temporal (intratidal or intertidal) resolutions, and the modeling of biogeochemical processes in terms of state variables and kinetic formulations. Hence, water quality modeling, as it stands now, is site-specific, problem-dependent and data-dependent. Advances in our understanding of physical transport processes would eventually result in a general hydrodynamic model applicable to any system. Advances in biogeochemical processes would result in the refinement of kinetic formulations and in the explicit incorporation of the components in the high trophic levels of food chain into the numerical model.

Acknowledgements

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Total Pollution Load Management in Lake Shihwa: Method and Application

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Abstract

Water quality is a major issue in Lake Shihwa. The lake's water quality has improved significantly [from 17.4ppm to 4.3ppm in chemical oxygen demand (COD)] due to the dilution of polluted lake water with cleaner coastal waters and the work of major wastewater treatment facilities. However, it falls short of the required level (grade II in seawater quality criteria) to support industrial water usage, water-related outdoor activities, and growth of some marine organisms.

The Ministry of Maritime Affairs and Fisheries (MOMAF) recently established a comprehensive environmental management plan to improve the lake's water quality. To support the designated water uses, the plan aims to decontaminate the water to a COD of 2ppm. However, assessment using water quality models has indicated that COD load reduction based on economically viable technologies and current emission control measures were insufficient for reaching this target. Therefore, this study strongly recommends the introduction of total pollution load management system (TPLMS) and that the comprehensive plan includes TPLMS implementation strategies.

Since TPLMS has been implemented in the major river systems in Korea, introduction of TPLMS for Lake Shihwa is less technically complicated. A prototype of coastal environment management system (CEMS) has already been developed to provide information on point sources and their distributions, analytical procedures to calculate quantitatively the targeted pollutant's loads, and several water quality and ecosystem models to predict water qualities of streams and coastal water bodies.

However, basic information on both external and internal non-point pollution sources and modeling tools (watershed models, in-stream water quality models, and coastal water quality models) to quantitatively assess the cause (pollution

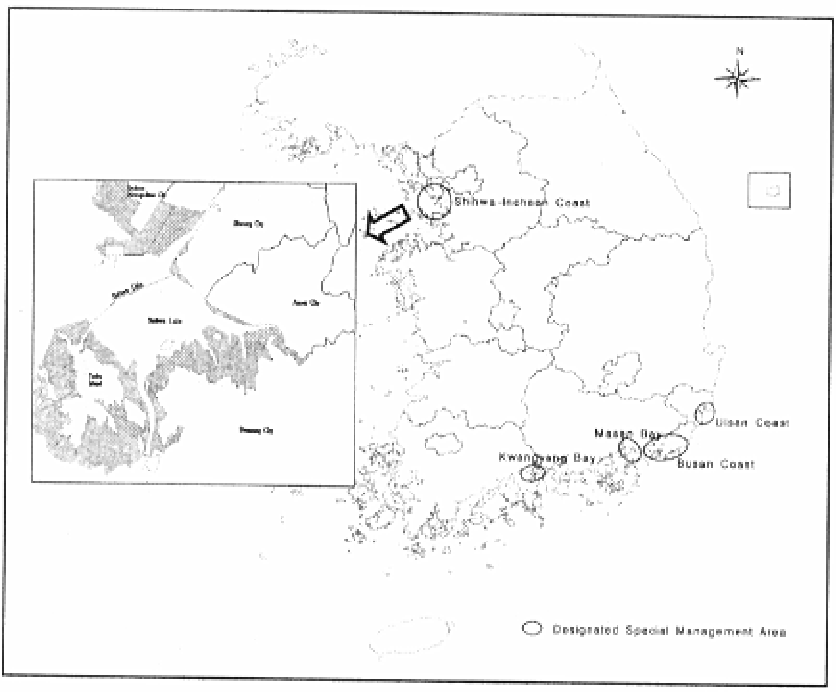
source)-response (water quality) relationship are inadequate for estimating the lake's assimilative capacity and the effect of load allocation among the various pollution sources. Therefore, more research efforts and technical developments focused on these subjects are necessary to fully support successful implementation of TPLMS in Lake Shihwa.

Introduction

Lake Shihwa had been designed as a freshwater reservoir (effective water volumes about 180 million ton) for irrigation and industrial water uses to the total 110km2 reclaimed land created by the Shihwa Reclamation Project (Fig. 1). However, severe water quality problems associated with untreated sewer and wastewater inputs from the watersheds and limited physical mixing prompted the Korean Government to open the lake's water gates to dilute the polluted water with the relatively clean coastal waters.

Although a significant improvement in water quality has been attained as a result of seawater dilution (Table 1), water quality is still far below the required level to support a wide range of designated water uses. In addition, considering the anticipated development pressure from the undeveloped southern parts of reclaimed land (about 98 km^2), one can hardly expect a remarkable water quality improvement in the near future (Lee, 2001). Previous studies predicted that the

Figure 1. Study Area and Designated 'Special Management Area (SMA)' in Korea.



lake's water quality can never be better than 2ppm (COD) even after the implementation of currently feasible all point source control measures (KOWAKO, 1998; Choi, 2001). This means that the pollution control measures based on traditional end-of-the-pipe approach and concentration-based effluent regulations are not enough to significantly improve the water quality in Lake Shihwa.

In view of the above, MOMAF has planned to implement TPLMS which is a water quality management system based on the idea that total pollution loads from watersheds and internal sources should be controlled within the assimilative capacity of the receiving water. This system has been implemented in Japan and the United States as 'area wide total pollution control system' and 'total maximum daily loads (TMDLs)', respectively. Recently, the Ministry of Environment (MOE) of the Korean government also implemented a kind of TMDL, the TPLMS to improve the water quality of major river systems (MOE, 2000; 2001).

In this paper, the general framework of TPLMS is briefly introduced and the technical approaches applied in Lake Shihwa are presented. The discussion is particularly focused on pollution load estimation, which is a crucial part in quantitatively estimating the assimilative capacity and determining the cause (pollution source) and effect (water quality) in Lake Shihwa.

Framework of the TPLMS

MOMAF has identified five heavily polluted coastal bays, which include Incheon-Shihwa, Busan, Kwangyang, Ulsan, and Masan, as 'Special Management Areas (SMAs) (Fig. 1) and developed a strategic environmental management plan to improve the water quality of the SMAs (MOMAF, 1999).

A site-specific comprehensive environmental management action plan for Lake Shihwa was developed based on the strategic plan. The plan basically adopted a watershed management approach to effectively control land-based pollutants and included very comprehensive water quality control measures such as construction of sewage treatment plants, management of non-point pollution sources, active

Table 1. Annual Water Quality Trend in Lake Shihwa.

Year	PH	DO	BOD	COD	TN	TP
1995	8.1	9.9	7.6	9.5	4.78	0.16
1996	8.4	9.5	8.9	14.2	6.05	0.25
1997	8.9	9.6	11.5	17.4	5.11	0.34
1998	8.3	11.6	7.1	8.0	5.64	0.10
1999	8.3	8.9	4.5	5.2	1.86	0.08
2000	8.3	11.1	3.5	4.3	1.88	0.07

participation of stakeholders, coordination and cooperation among relevant agencies, land use planning based on the assimilative capacity of coastal waters, and capacity building at the national and local levels (MOMAF, 2001).

Although the action plan includes comprehensive water quality control measures and approaches, implementation results seemed to be not so optimistic because those measures are basically dependent on traditional end-of-the-pipe approach and concentration-based effluent regulation. Indeed, assessment using water quality models indicated that load reduction based on economically viable technologies and current emission control measures are not sufficient to improve the lake's water quality up to the desired level (MOMAF, 2001). Thus, prompt implementation of TPLMS was strongly suggested because it is considered a powerful tool to reduce pollutant loads down to targeted level as well as to encourage sustainable land use planning within the water's assimilative capacity.

General figures of suggested TPLMS are summarized in Table 2. In Lake Shihwa, conventional and toxic pollutants (particularly mercury) were recognized as major water quality problems. However, major management target was confined to the organic materials (indicated by chemical oxygen demand, COD) in the fist phase of TPLMS implementation because of technical difficulties to quantitatively estimate

Table 2. Proposed TPLMS Framework in Lake Shihwa.

Subject	Contents
Target	-Seawater quality criteria Grade II (COD=2ppm) for main part of the lake -Seawater quality criteria Grade III (COD=4ppm) for inner part of the lake
Area	-Designated 'Shihwa Special Management Area' (lake=56.5km^2, coastal watershed=351.74km^2)
Control Pollutant	-Primary management target: chemical oxygen demand (COD) -Include total phosphate (TP) and total nitrogen (TN) if necessary
Pollution types	-All pollution sources(point & non-point sources)
Implementation	-MOMAF: Development Strategic TPLMS plan, provide guidance and technical support for TPLMS action plan development, approval TPLMS implementation plan. -Cities and Counties: Development of TPLMS action plan and implementation
Planning period	-TPLMS action plan: 5-year planning period -Submit annual TPLMS progress report
Inspection and monitoring	-Water quality monitoring: stream water quality (cities, counties, or MOE), lake water quality (MOMAF) -Development of total pollution load account list and submission of annual progress report (cities and counties) -Inspection of individual wastewater dischargers (MOE)
Legal scheme	-Marine Pollution Prevention Act or proposed special act for TPLMS -Agreement among stakeholders

pollutant load and lack of basic information to identify detailed transport passages of toxic pollutants.

TPLMS management target at 2ppm in terms of COD was suggested for main part of Lake Shihwa because it is the required level to support swimming and other water-related outdoor activities including growth of some less sensitive marine organisms in national seawater quality criteria. A less strict management target was set for the inner part of the lake (3ppm in COD) to reflect the rather worse local management conditions such as limited water mixing and direct influences of land-based activities. It is notable that the TPLMS targets were not based on the lake's assimilative capacity but on the general seawater quality management goal of the country. This is partly due to the lack of knowledge on the assimilative capacity and the responses to the pollution loads from the coastal watersheds.

Pollution Load Estimation

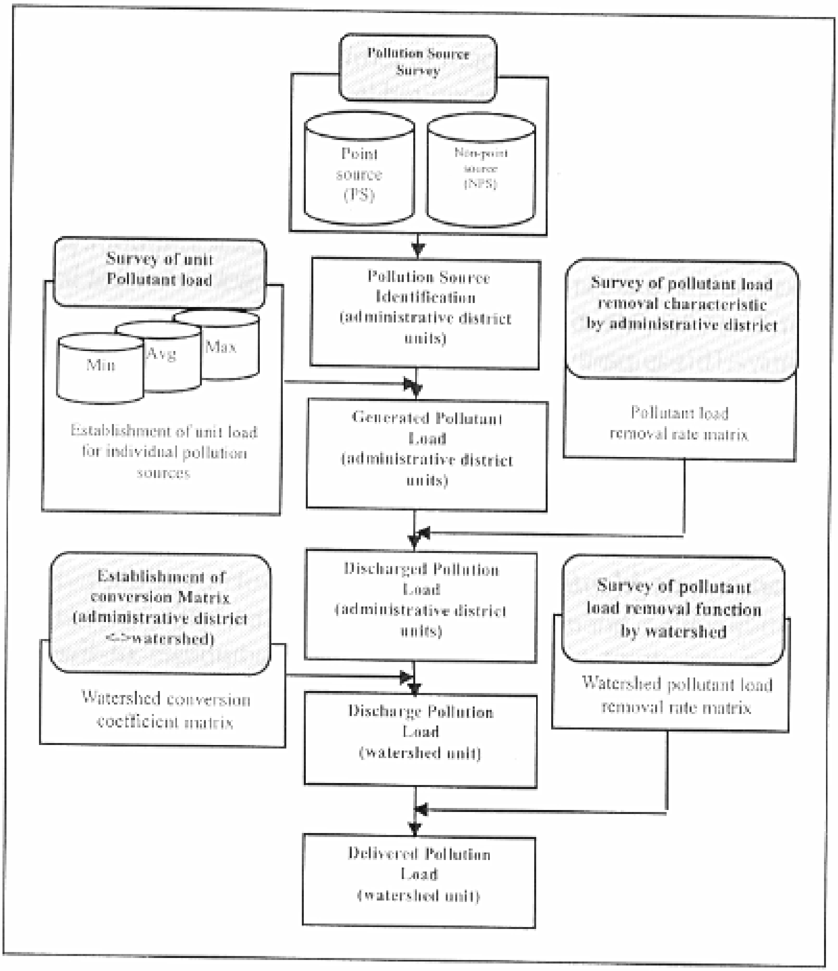
Pollution source identification

Information on point sources (e.g., population, number of livestock, and industrial wastewater dischargers, etc.) and non-point sources (e.g., land-use, etc.) were collected from the statistics and documents available in the coastal administrative districts of Ansan City, Siheung City, and Hwasung City. Distributions of individual pollution sources were identified at least in Ri or Dong scales (the smallest administrative district units in Korea) to increase the geographical resolution. Since the administrative district boundaries are not matched with watershed boundaries, the collected data were converted to watershed units to assess pollution sources on watershed basis (Fig. 2). In addition, all collected and analyzed pollution source information were stored in the geographic information system (GIS) based CEMS to effectively support the decision-making process pertaining to the lake's environmental issues (Yoon, 2001).

Generated pollution load

Three different pollution loads were defined and termed respectively as generated pollution load, discharged pollution load, and delivered pollution load for the detail load estimation (Fig. 2). A conventional unit-load approach was used for the estimation of generated pollution load. Generated load is given by the times of a pollution source unit to corresponding unit-load, which reflects the pollution load at source level before any treatment. It was strongly recommended to establish the site-specific unit-load for TPLMS development. Due to time and resource constraints, the standard unit-load given in the Technical Guide for TPLMS Development (MOE, 2000) was used for the site-specific unit-load development.

Figure 2. General Flowchart of Pollution Load Estimation (MOE, 2001).



The standard unit-loads are established according to the general pollution source classification schemes such as residential modes for the population, species for the livestock, standard industrial classifications for industrial wastewater dischargers, and land use purposes for non-point sources. For example, standard unit-loads for the population were set in four sub-categories: residents in the city, residents in the rural area, workers in the city, and workers in the rural area to reflect the different amount of sewage generation and pollutant concentration (Table 3). Unit-loads for other pollution sources were also adapted from the TPLMS development guideline.

Discharged pollution load

The generated load is significantly decreased in many ways before it is finally discharged into public waters. Thus, identification of detailed treatment passages and efficiency information on the applied treatment methods are required to

Table 3. **Unit-Loads for Population.**

Area	Mode	BOD (g/cap.day)	TN (g/cap.day)	TP(g/cap.day)
City Area	Resident	50	10.5	1.2
	Workers	26	8.0	0.7
Rural Area	Resident	49	13.2	1.5
	Workers	26	8.0	0.7

calculate the discharged load from the generated load. It was found that the generated loads from the population in the lake's watersheds took at least eight different treatment passages until they were finally discharged into the streams or Lake Shihwa (Fig. 3).

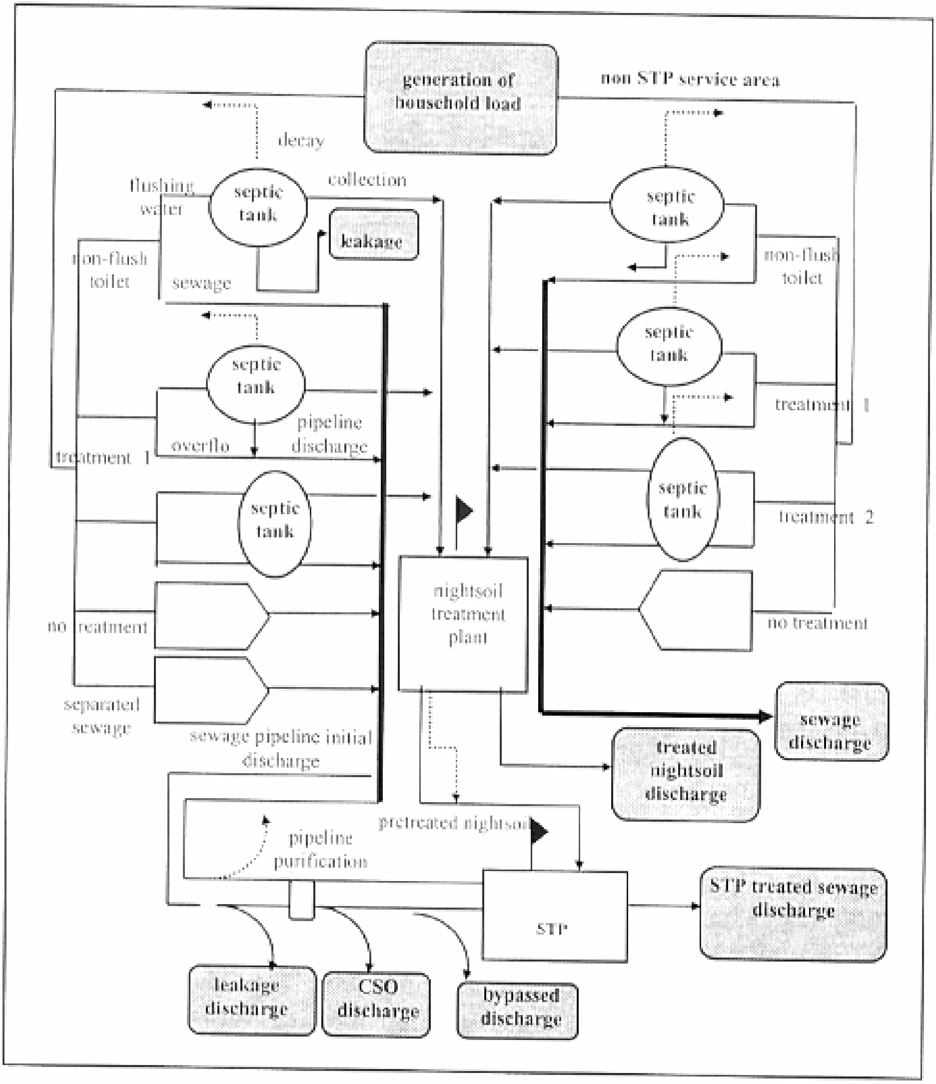
Since the removal rates of different types of septic tanks or other treatment facilities were hardly estimated, the installed septic tanks were grouped into two categories based on the legally required minimum removal rates according to treatment capacity (treatments 1 and 2 in Fig. 3). It was assumed that lower efficient septic tanks have averages 50 ~ 65 percent of removal rate for biochemical oxygen demand (BOD) and nominal removal rates (< 10 percent) for other pollutants, but higher efficient septic tanks have 90 ~ 95 percent removal rates for BOD. Presentation of the detailed calculation procedure of discharge load is beyond the scope of this paper. A similar applicable procedure can be found elsewhere (MOE, 2000; 2001).

Delivered pollution load

Except the discharged load from the sewage treatment plants (Shihwa STP and Ansan STP), which discharge effluent directly into the coastal waters through submerged ports, only part of discharged loads were actually delivered to Lake Shihwa through eight streams and CSO discharge ducts. Considering that the natural purification process in streams are highly variable and site-specific in nature, direct measurements of stream flow and pollutant concentration under different flow conditions (typical normal and high flow conditions) were made for estimations of delivered load and delivery ratio (=measured delivered load/discharged load in the upstream watershed). Annual delivered load, which was used for the lake's water quality input data, was estimated from extrapolation of measured delivered load.

In coastal water quality management, assimilative capacity is usually indicated by the targeted pollutant load. Thus, it could be defined in terms of generated load, discharged load, or delivered load according to the management purpose. In previous studies undertaken in Lake Shihwa, assimilative capacity was indicated by the delivered load because it was generally used as the model input to predict the lake's water quality (KOWAKO, 1998; Choi, 2001).

Figure 3. Treatment Passages of Household Sewage and Night-Soil (MOE, 2001).



Discussion and Conclusion

Although many decent load estimation models were available, the traditional unit-load method was used because it is simple and practically applicable in lake Shihwa. In addition, this method is consistent with the standard load estimation method adapted in TPLMS, which was already implemented for the water quality improvement in lakes, rivers, and streams in Korea (Lee, 2000; MOE, 2000; 2001).

It should be noted that this approach requires more efforts than any other analytical load estimation methods to obtain information on the transport passages and efficiency of treatment facility for the discharged load estimation. From the technical point of view, the discharged load estimation method may be limited because transport passages are not always possible to be clearly identified and the removal rates of some treatment facilities are not well known. Despite these limitations, it has advantages from the management perspective. Firstly, water pollution problems not only at pollution source level can be clearly identified and assessed but also at subsequent treatment levels in tracking the transport passage. Secondly, it is relatively easy to establish targeted pollution control measures adequate for the identified levels in TPLMS planning stage and to quantitatively assess the effects of the implemented pollution measures in evaluation stage later.

Standard unit-load for non-point sources were derived from the number of actual load measurements for the representative land-use types such as paddy field, crop field, forest, dairy farm, golf field, residential area, etc. The unit-load method for non-point load estimation was taken because it was practically applicable given the financial constraint, although it did not account for the site-specific characteristics and pollutant behaviors on individual rainfall events. A watershed-scale modeling techniques such as AGNPS, STORM, SWMM, and HSPF for non-point load estimation (USEPA, 1997) are being applied to overcome the problems.

Measurement of reliable delivery ratio is very important because it reflects the direct relationship between the discharged load that is readily controllable entity in water quality management and the delivered load that actually affects the lake's water quality. However, the delivery ratios estimated in this study and previous studies (KOWAKO, 1998; Choi, 2001) seemed to be not entirely representative because measurements were made in short periods. Considering the extreme seasonal changes of stream flows in the lake's watersheds, a long-term flow and water quality measurements are needed to obtain more reliable delivered load and delivery ratio.

It is reasonable to use the delivered load as a model input to predict water quality. The lake's assimilative capacity, however, needs to be given in terms of discharged load at least in management perspective simply because it is a readily manageable load. In this sense, conversion of delivered load to discharged load is necessary to effectively define the assimilative capacity. This is the reason we paid so much attention in load estimation procedures, to obtain the reliable delivery ratio.

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The Use of Ecological Carrying Capacity for Assessment of Entrainment Impact of Cooling Water Intake of the Hin Krut Coal-Fired Power Plant on the Coast of Thailand

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Abstract

The project entails the establishment of a 2 x 700 MW coal-fired power plant at Prachuabkirikhan Province, Southern Thailand. In its operational phase, at a plant efficiency of 84 percent, 33,000 m^3/hr or 242,827,000 m^3/year of seawater will be drawn from the sea in front of the project site for cooling purposes. It is assumed that all aquatic animal larvae will be killed in the cooling system. The entrainment impact has first been assessed by using the Thai Department of Fisheries formula, which was previously used in assessing the entrainment impact of two other power plants along the coast of Thailand. The result reveals uncertainty due to the survival rate of animal larvae. The authors therefore utilized the carrying capacity concept by using data on fishery production of every type of fishing gear from both commercial and artisanal fisheries. The fishing area is 11,000 km^2 with an average depth of 15 m. Fish recruitment is assumed to be 50 percent. The term ecological carrying capacity is arbitrarily adopted as the capacity that a unit volume of water can carry a certain biomass of aquatic animals. The results show that the carrying capacity of the seawater in the area is about 0.009258 kg/m^3. Accordingly, the annual loss by cooling water intake of this coal-fired power plant is about 2,248,185 kg/yr. The developer has proposed mitigation measures by installing artificial reefs covering an area of 50 km^2 and a sea ranching program for compensation of the mentioned loss.

Project Description

The project entails the establishment of a 2 x 700 MW coal-fired power plant at Amphoe Bangsapan, Prachuabkirikhan Province on the coast of southern Thailand. It is being developed by the private sector, the Union Power Development Co., Ltd., an independent power producer according to the policy of the Thai Government through the Electrical Generating Authority of Thailand. The coal to be used is bituminous which is imported from Indonesia, Australia, and South Africa.

In the cooling process, the closed circuit system is applied. The seawater will be drawn from the sea in front of the project site at a rate of 33,000 m^3/hr or if the plant efficiency of 84 percent is taken into consideration, would be 242,827,200 m^3/yr, with 1 percent loss during the process before discharging. The intake building is about 600 m offshore at about 4 m from MSL. At the intake point, it will be equipped with a traveling band screen of 9.5 mm mesh size and bar screen so that the intake velocity can be limited to 0.09 m/sec. This can prevent juvenile animals from being entrapped. Since sodium hypochlorite, an anti-fouling chemical, is used along the intake pipe, it is assumed that all organisms taken in with the cooling water will die out in the process.

Impact Definition and Assessment

As mentioned in the brief project description, one of the many impacts of the project is the entrainment of animal larvae, which is unavoidable, despite some of the engineering measures employed to prevent it. Two methods were employed in this assessment. One is the Thai Department of Fisheries (DOF) method and second, the ecological carrying capacity method. The authors used the concept of carrying capacity and the Office of Environmental Planning and Policy of Thailand accepted the method.

The Thai DOF developed the traditional method in 1983 (Division of Marine Fisheries, 1983; Inland Fisheries Institute, 1983). It is carried out by counting the animal larvae present per unit water volume. The present survey was conducted by sampling animal larvae in the sea and river in front of the project site for eight months so that covers both southwest and northeast monsoon seasons.

The occurrence of animal larvae was determined in a cubic meter of water. Results show that each cubic meter of water contains one fish larva, four fish eggs, 58 shrimp-crab larvae, and 45 mollusk larvae (Table 1).

The two organizations of the DOF have used the formula below in calculating the loss of aquatic animals through entrainment of two power plants:

Table 1. Summary of Economic Loss.

Animals found	Number/m³	Volume of intake water (242,827,200 m³)	Survival rate of animal larvae (0.1%)	Weight of animals (kg)	Economic loss Thai Baht
	N	I x T	x S	x M	x Pr
Fish larvae	1	242,827,200	242,827	24,827	206,400
Fish egg	4	971,308,900	97,130	9,713	82,500
Shrimp-crab larvae	58	14,083,966,000	14,083,966	282,679	14,083,900
Mollusc larvae	45	10,927,215,000	10,927,215	218,544	1,398,700
Economic loss					15,771,500
Real loss					31,543,000

Given:
- *Volume of water intake is 33,000 m³/hr or 343827,000 m³/yr at 84% efficiency*
- *Fish size 10 pcs/kg, shrimp-crab 50 pcs/kg, mollusk 50 pcs/kg*
- *Fish price (based on DOF's Economic Office, 1999)*
- *Survival rate of animals is 0.1%, except fish eggs 0.01%*
- *Fish recruitment 50%*

$$EL = N \times I \times T \times S \times M \times Pr$$

where
EL - Economic loss
N - Average number of animal larvae
I - Intake water volume (m3/hr)
T - Number of intake hours
S - Natural survival rate of aquatic animals
M - Animal weight entering into fishing condition
Pr - Market prices of aquatic animals

In such a calculation, however, the DOF has used two different survival values of 5 percent and 1 percent for the first and second power plants, respectively, which seemed to be very high.

In the present project, the survival rate of 0.1percent was used which is modified from the study of Wright (1978) used in the survival rate of flounders. The calculation of economic loss of the project according to the DOF formula is shown in Table 1.

However, using this calculation the authors are not satisfied with the assumption regarding the survival rate of the animal larvae, which was deemed to be too arbitrary. If it is too low, survival will be underestimated or vice versa. The concept of ecological carrying capacity was therefore utilized so that the real loss of animals in each cubic meter of seawater can be more precisely estimated. The concept is based on the capacity that a unit volume of seawater can carry a certain weight of marine life.

In general, the stock assessment or standing crop is required to calculate the carrying capacity. However, in such a wide area of open sea, the stock assessment of every species of animals is impossible to obtain.

The production from Fishing Zone 3 by various types of fishing gear is shown in Table 2.

The total fish production from all fishing gears is therefore 694,378 MT or 694,378,000 kg. Together with the production from small-scale coastal fisheries accounting about 10 percent of commercial production, the total production should be about 763,815,800 kg. This figure is accordingly assumed to be 50 percent of the total biomass. The total biomass of aquatic animals in the fishing zone is thus as much as 1,527,631,600 kg or 1,528 million MT.

Considering a fishing area of 11,000 km^2 with an average depth of 15 m, the total volume of fishing water is equal to 165,000 million m^3. Taken against a total animal production of 1,528 million MT, each cubic meter of seawater in the area can carry animals weighing as much as 0.009258 kg. This has been used for the basic calculation of total loss *in situ* of aquatic animal value per year, which is as much as 2,248,185 kg/yr.

Mitigation Measures

Despite numerous preventive measures provided, such as limiting the intake current to 0.09 m/second, by installing a fish screen under the water intake building and 9.5 mm band screen, loss is still unavoidable and requires compensation measures. Alternative measures to compensate the *in situ* resource loss have been

Table 2. Fish Production in 1995 from Fishing Zone 3 Using Various Types of Gear.

Fishing gear	Fish production (MT)		
	Pelagic fish	Demersal fish	Others
Otter board trawler	8,077	129,283	289,206
Twin trawler	3,487	32,326	6,932
Beam trawler	--	33	494
Purse seine	154,629	18,347	--
Anchovy purse seine	47,830	8	--
Mackerel drift net	1,424	8	--
Mackerel gill net	1,986	--	--
Push net	--	202	106
Total	217,433	180,207	296,738

Source: Department of Fisheries, 1999

proposed. The project developer has adopted two compensation measures such as installation of artificial reefs and establishment of sea ranching program.

The effects of an artificial reef have been intensively reviewed by De Silva (1988) who concluded that it still requires further research. However, for the Thai situation, Supongpan (1995a & b) and Sinanuwong and Singtothong (1997) recommended the installation of artificial reefs along the coastal area of Petchburi Province, 200 km north of the project site. The reefs are made of cement blocks measuring 1.5 x 1.5 x 1.5 m. Five hundred pieces were deployed covering an area of 50 km^2. Their study reveals that after three years of monitoring artisanal fishing production in the area, an increment of four times (given actual production figures) from the first year has been observed. Moreover, some species that had disappeared from the area quite a long time ago, such as the white pomfret, have returned.

The sea ranching program on the other hand, which involves propagation, nursing and releasing of aquatic animal into the coastal area, is a new innovation in the tropics. However, Menasveta (pers. comm.) has indicated satisfactory results of a sea ranching program along the coast of Sichang Marine Station of Chulalongkorn University, Thailand. Economic calculation suggests that the returns from both programs can compensate for the loss.

Discussion

Although the use of ecological carrying capacity in this study for impact assessment has proven to be satisfactory, there are still at least three shortcomings. The first is due to the uncertainty of the *in situ* resources value due to the used assumption. Further research is still required to estimate reliable standing crop value in a particular water area, so that a precise result can be further obtained. The second is due to uncertainty in artificial reef installation whether it really increases productivity or just concentrates the aquatic animals. The third is that there is very little research in sea ranching in the tropics.

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Tourism Carrying Capacity: Assessment and Application

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Abstract

The concept of carrying capacity, which is basic in resources management, has moved into recreation and tourism since the 1960s. Research on the concept in recreation has shifted from finding the numbers of users to establishing appropriate levels of use and developing new management frameworks. Although the concept remains appealing in tourism research it has not been widely applied due to the diffuse nature of tourism itself, different reaction of the local population, different perception of the tourists themselves, and its unpopularity with the industry as it implies control. Likewise, good examples of carrying capacity are limited in coastal tourism, although a generalized model is available. For diving sites, the carrying capacity can also be estimated, based partially on the divers' experience. The concept of tourism carrying capacity is too premature to be ignored as research problems associated with it have yet to be investigated properly. Future research to identify maximum appropriate numbers of visitors and tourists is suggested.

Introduction

Carrying capacity is a fundamental concept in natural resource and environmental management to define the maximum level of use an area can sustain as set by natural limits. For example, in wilderness management, this describes the number of animals of a particular species that can use the range on a sustained basis, given available food, shelter and water (Hendee et al., 1990). Much of the earlier work on carrying capacity thus has a strong biological focus (Stewart, 1993).

Research on carrying capacity in recreation and tourism evolved rather differently (Butler, 1996). In the early 1960s, the concept moved across to recreation to determine the visitor capacity as wilderness areas became degraded under escalating levels of visitor use (Newsome et al., 2002). From its ecological basis, the concept expanded to include a social component, thus increasing the research on the social aspects (Butler, 1996).

The 1970s saw little interest in the concept in tourism as attention was on the assessment of the effects of tourism, reflecting the growing concerns of developmental impacts on the environment. But significant progress was made in recreation as the research moved away from a search for specific numbers or limits to a recognition of alternative capacity levels, some based on the visitors' expectations. Overall, tourism remained slow in incorporating such research from recreation. From the 1990s, renewed interest on tourism carrying capacity is related to work on sustainable development (Butler, 1996).

This paper gives an assessment of tourism carrying capacity, including definitions, types, and operational difficulties, and an overview of its application, focusing on coastal and reef tourism.

Definitions and Types

The following commentary provides some idea of the range of definitions on tourism carrying capacity. It starts with the basic and moves up the scale, to reflect the increasing components and factors to be considered in the concept.

(1) The most basic form of the concept refers to the maximum number of tourists:
"Tourism capacity can be simplistically defined as the maximum number of tourists that can be contained in a certain destination area." (O'Reilly, 1986).

(2) The concept can also be applied to some other maximum levels of tourist development:
"'Tourist carrying capacity'...applies not only to the maximum number of tourists - or tourist accommodation - which seem desirable at a given time, but also to the maximum rates of growth above which the growth process itself would be unduly disruptive." (De Kadt, 1979)

(3) Besides the number of tourists, their satisfaction has to be considered:
"The carrying capacity may be defined as the maximum number of visitors that can be accommodated without causing excessive environmental deterioration and without leading to a decline in visitor satisfaction." (Hovinen, 1982)

(4) Tourists' satisfaction is also linked to the level of unacceptable change or degradation of the physical environment:
"Carrying capacity is the maximum number of people who can use a site without an unacceptable alteration in the physical environment and without an unacceptable decline in the quality of the experience gained by the visitors." (Mathieson and Wall, 1982).
"Carrying capacity can be defined as the level of use beyond which impacts exceed acceptable levels specified by evaluative standards." (Shelby and Heberlein, 1986)

(5) Other aspects of degradation or use exceeding acceptable levels are included to complete the picture:
"Tourism carrying capacity is defined as the physical, biological, social, and psychological capacity of the park environment to support tourist activity without diminishing environmental quality or visitor satisfaction." (Lindsay, 1986)
"Carrying capacity can be defined as the number of visitors that an area can accommodate before negative impacts occur, either to the physical environment, the psychological attitude of the tourists, or the social acceptance level of the hosts." (Martin and Uysal, 1990).

Researchers have also attempted to provide more comprehensive definitions (Pigram, 1983; Shelby and Heberlein, 1984). Overall, the concept of tourism carrying capacity carries several implications: a certain number of tourists that a destination area can accommodate; the notion of a level beyond which conditions are unacceptable, e.g., degradation of the physical environment; and the impacts not only on the physical environment but also on the psychological attitude of tourists and social acceptance level of the host population. The concept has surfaced many times in relation to achieving sustainable tourism.

As the concept of carrying capacity moves into recreation and tourism, it has broadened into various components or types of carrying capacity (Hovinen, 1982). Three to five major categories of capacities can be identified to include the destination or area visited, the tourists themselves, and the local or host population (Ceballos-Lascuráin, 1996; Gee and Fayos-Solá, 1997; Johnson and Thomas, 1996; Watson and Kopachevsky, 1996). The following categorization of tourism carrying capacity is not meant to be rigid but helps in identifying and analysing the unacceptable level associated with tourism development:

- biophysical, ecological, environmental (level beyond which the area is degraded or compromised);
- psychological, perceptual (level beyond which the tourists are no longer comfortable in the area);

- sociocultural, behavioural (level beyond which the local population do not want the tourists);
- economic (level beyond which tourism is more of an economic liability), and
- physical-facility (level beyond which tourist facilities are 'saturated'); managerial (level beyond which tourist management is no longer effective).

The various capacities are related or commensurate with various tourist impacts in the area and the interaction between tourists and the host population (Butler, 1996; Watson and Kopachevsky, 1996). These carrying capacities do not *all* have to reach their limits or exceeded for problems to emerge. For example, too many tourists in five-star hotels overtax the water resources resulting in problems with sewage facilities, in turn casing a shortage of potable water and beach pollution (Butler 1996). Cooper et al. (1993) have emphasized that the level of tourist presence creates the impacts rather than tourist numbers.

The various types of carrying capacities form a basic step in tourism and recreation planning to determine the upper limits of development and visitor use. Various criteria are used to determine the capacity levels, many of which are minimum or basic standards specified for facilities and uses established by planning authorities (Baud-Bovy and Lawson, 1998; Inskeep, 1991).

Overviews and Assessment

Butler (1996) in his survey of the carrying capacity in tourism over the past four decades, noted that research has progressed from ignoring the topic, to a search for specific numbers, and to management approaches based on social and experimental expectations. He lamented that tourism researchers paid specific attention to the issues only in the last half decade and, in general, were not familiar with the extensive literature already in recreation publications.

According to Murphy (1985), the early studies on tourism carrying capacity focused on the physical and biological dimensions of the environment. Generally, they followed a three-step assessment approach: (1) recreation activities and needs of tourists; (2) physical and biological capacity of the site; and (3) combination of both steps to establish the level of compatibility in various zones. There were few examples of actual application in tourism management and planning. The landmark study was for the Irish Tourist Board (Stewart, 1993), a demonstration project to devise the methodology and ongoing planning process. Other studies include determining the physical limits of beaches and water availability in Yugoslavia (Hall, 1974) and the identification of capacity thresholds based on analysis of tourism impacts in Spey Valley, Scotland (Getz, 1982).

Basically, carrying capacity is conceptually appealing but difficult to operationalize. Murphy (1985) stated that "the notion behind the carrying capacity concept is simple, its application is complex, due to the difficulty of measuring change and establishing causal relationships." Getz (1983) has also stated that the reluctance or inability to apply the concept in tourism strategies can be attributed to its conceptual and methodological complexity.

Various attempts have been made to establish the methodology of tourism carrying capacity. One general approach is to express it as a function of various factors as seen in the equation by Glasson et al. (1995):

TCC = f (Ecol, Phys, Econ) (TC, RA, Pol)

Tourism Carrying Capacity (TCC) is expressed as a function of ecological systems (Ecol); physical infrastructure (Phys); economic characteristics of tourist investment and expenditure (Econ); tourists' characteristics in social-cultural and behavioral terms (TC); residents' acceptance or tolerance of tourism activity (RA); and political capability and authority to take effective management decisions (Pol).

Ceballos-Lascuráin (1996) cited a Spanish source on the methodology for estimating the protected area capacity, which involved six steps. Three levels of carrying capacity were identified and applied to examples: (1) physical carrying capacity (PCC), the maximum number of visitors that can physically fit into a defined space over a particular time; (2) real carrying capacity (RCC), the maximum permissible number of visits to a site, once the corrective site factors (biophysical, environmental, ecological, social and management) have been applied to the PCC; and (3) effective or permissible carrying capacity (ECC), the maximum number of visitors that a site can sustain, given the management capacity (MC) available. ECC is obtained by a comparison of RCC with MC of a corresponding protected area administration.

It is obvious that numerous factors and complex interrelationships need to be considered in determining the carrying capacity. Mathieson and Wall (1982) stated that capacity levels are influenced by two groups of factors: (1) tourist characteristics (socio-economic characteristics, level of use, length of stay, types, and levels of tourist satisfaction); and (2) characteristics of the destination area and its population (natural environmental features and processes, economic structure and development, social structure and organization, political organization, and level of tourist development).

In a further analysis, Lindberg et al. (1997) stated that the carrying capacity concept has three limitations. Their definitions often provide little guidance for implementation. The concept is considered as scientific and objective and not a management notion, and it focuses on use levels or numbers of visitors while the management objectives relate to conditions. In their evaluation of tourism in natural areas, Newsome et al. (2002) gave five reasons for the failure of the concept to generate practical visitor use limits. Thus, the implementation of the tourism carrying capacity in nature areas to highly developed destinations faces a wide range of conceptual and methodological difficulties.

An obstacle to the application of the tourism carrying capacity comes from the tourism industry itself. One reason is that the concept implies a limit to growth and is therefore quite unpopular with the industry, which sees possible measures to limit growth. Tourism represents free enterprise and the less control on the private sector, the better. The concept of capacity implies control, and to the private sector, this is a potential loss of profits (Butler 1996). Getz (1983) is right in stating that economic reasons are likely to prevail in arguments against carrying capacity. Also, a clear responsibility for the quality of resources of many destination areas is absent. There is virtually no regulation beyond the normal planning controls and no clear and effective method of the enforcement of limits (Butler, 1996). In contrast, environmentalists want tourism carrying capacity as it can be a possible measure of health of the environment before degradation sets in.

In their assessment, Glasson et al. (1995) concluded that the carrying capacity concept "is difficult to define in any absolute sense, and as such has met with mixed results when expressed as a management tool." "Carrying capacity is a relative management concept, or framework...is not precise, as it requires both judgment and scientific theory." In his assessment, Butler (2000) concluded: "Thus to even determine, let alone apply, carrying capacity limits in most tourism destinations is next to impossible because of the mix of tourists, the nature of the industry, the mix of local opinions and desires, and the lack of overall control of tourism."

The usual or traditional approaches to carrying capacity management have little success due to various reasons: the unrealistic expectation of a magic number; untenable assumptions such as direct relationship between tourism and impacts; inappropriate value judgments; and insufficient legal support. This has led to alternative approaches with the shift from establishing use limits to issues of identifying environmental, social, and economic conditions desired by the community, and the creation of growth management strategies for managing tourism's carrying capacity challenges (Williams and Gill, 1998).

In recreation management strategies, the focus has shifted from "How many is too many?" to "What are the desired conditions?" Various alternative frameworks have been developed to include more appropriate concepts and processes. These include limits of acceptable change (LAC), which attempts to solve some problems of identifying maximum use levels and is the most widely used framework. Others include visitor impact management (VIM); recreation opportunity spectrum (ROS) and its variation; tourism opportunity spectrum (TOS); ecotourism opportunity spectrum (ECOS), which incorporates and modifies ideas from ROS and TOS to address ecotourism; visitor activity management process (VAMP); visitor impact management process (VIMP); visitor experience resource protection (VERP); and tourism optimization management model (TOMM) (Boyd and Butler, 1996; Hendee, Stankey and Lucas, 1990; Newsome et al., 2002). A table on the choice of an appropriate recreation/tourism framework can be found in Newsome et al. (2002).

Coastal Tourism

There are few good examples of carrying capacity in coastal/marine development control (Clark, 1996). According to Stewart (1993) many past studies focused on impacts and management needed to prevent or minimize damage arising from impacts, e.g., Heritage Coast, United Kingdom, and Heron Island (Great Barrier Reef).

Basic to understanding the carrying capacity in the coastal environment is the need to know the impacts and the sensitivity of coasts to recreation and tourism. Edwards (1987) provides a summary table on the sensitivity of and ecological damage to the habitats by various recreational and tourist activities. Sometimes, the intensity of the impacts can be reflected by specific standards. For example, Sowman (1990) provided space standards for major coastal recreation activities in which general standard, low density, and high density values were given for various beach and shore activities, boating activities, and mixed boating activities.

A tourism specialist and a coastal geomorphologist (Pearce and Kirk, 1986) have proposed a generalized model for relating tourism impacts in four different zones of the coast with various types of carrying capacity. Moving in a seaward direction, the four zones with their corresponding tourism use and type of carrying capacity are as follows: (1) hinterland (accommodation and service sector) with physical carrying capacity; (2) dunes (transit zone) with environmental carrying capacity; (3) beach (recreational activity zone) with social carrying capacity; and (4) sea (recreational activity zone) with environmental carrying capacity.

But carrying capacity for coastal tourism has not been widely applied comprehensively, reflecting a similar situation in tourism: the complexities of the coastal environment and the multifaceted character of tourism; continued unplanned tourist expansion along the coast; and difficulty in specifying limits or thresholds which can be applied. In some instances, guidelines are possible, e.g., water quality standards, but implementation is difficult because of shortage of money (Pearce and Kirk, 1986).

Even for environmental carrying capacity, it is not easy to determine, as illustrated in a study from the Maldives (Brown et al., 1997). The ecosystem stress of the Maldives was assessed by three carrying capacity indicators: solid waste disposal, water quality, and tourist perceptions based on data gathered from questionnaire surveys of tourist and tourist resorts, and interviews with officials and resort operators. Lindberg and McCool (1998) commented that the set of "concern, indicator, and standard" reflects a particular desired condition and is subjective, depending on the priorities and objectives of diverse stakeholders, and thus cannot form the basis of carrying capacity. They argued that the carrying capacity framework may work well in situations such as wildlife management where a consensus exists for objectives and extensive data are available on use-impact relationships. With the complexity of tourism-development situations, there is rarely consensus or adequate data and they recommended various management-by-objectives approaches, e.g., LAC, VIM. In their response, Brown et al. (1998) recognized the problematic and controversial nature of carrying capacity. They stated that defining a safe minimum standard for levels of tourist activity is extremely difficult for a number of reasons: lack of appropriate and accurate data on ecological and other processes and poor implementation of regulations.

For coastal tourism, all types of capacity thresholds have to be considered (physical, environmental, economic, and sociological). De Ruyck et al. (1997) and Saveriades (2000), in their studies on the social carrying capacity for sandy beaches and coastal resorts, respectively, were aware of the need for other types of carrying capacity.

Diving Tourism and Coral Reefs

The coral reef carrying capacity for underwater tourism depends on factors that include the size and shape of the reef, composition of coral communities, recreational activities, and the level of experience of snorkelers and divers (Salm, 1986). It has been estimated for various locations based on the extent to which coral reefs can support diving without serious degradation and supported by divers' observation or experience, e.g., Bonaire, Red Sea.

Dixon et al. (1993) estimated that dive sites in Bonaire, Netherlands Antilles, could support 4,000-6,000 dives/year without causing serious degradation. Based on this, a maximum limit of 200,000 dives/year (or 20,000 divers if each makes 10 dives) is recommended for Bonaire's carrying capacity for diving tourism.

The coral reefs in Red Sea diving sites are estimated to withstand 10,000-15,000 dives/year without serious degradation. The current number of sites can be doubled to 74. If each diver makes 10 dives, the reefs in Red Sea could potentially support 74,000-111,000/year. With 300,000 divers planned for year 2000, this would generate three million dives. If the dives were evenly spread, each site would have 40,000 dives/year (3 million divided by 74 sites) (Hawkins and Roberts, 1994).

In the Grand Cayman island, the goal was to determine the cumulative number of users (divers and snorkelers) that a reef can tolerate without becoming significantly degraded. Two site types were examined, high intensity mooring sites (+6000 dives/mooring/year) and low intensity mooring sites (<1000 dives/mooring/year). Three samples each were collected at 15m, 55m, and 200m from a mooring eye at each site. The results indicated that diving pressure should not exceed 2,500-3,000 divers per mooring/year to maintain a coral cover and high coral diversity (DOE, 1997).

Two widely used frameworks to estimate recreation/tourism thresholds have been applied to coral reefs. Shafer and Inglis (2000) applied the LAC to the Great Barrier Reef World Heritage Area. The procedure requires that standards for indicators are related to the health of coral and fish, the amount of infrastructure, and/or numbers of users be developed and eventually monitored to determine if changes are acceptable.

The LAC has also been applied to Saba Marine Park, Saba, Netherlands Antilles. This provides some standards (defined as minimally acceptable conditions, not necessarily those that are desirable) of acceptable change for water quality, sedimentation, damaged corals, fish stock, number of dive boats, and group size. The standards should provide the park managers a clear indication of whether conditions are acceptable or they have been violated (Schultz et al., 1999).

Marion and Rogers (1994) applied the VIM to coral reefs of Virgin Islands National Park, to design strategies for influencing the amount or type of visitor use, location of visitor use, and visitor behavior. The VIM recommendations for the protection of coral reefs include the following: restrict high-impact uses, contain rather than disperse recreational use, encourage use of resistant environments, teach low-impact recreational practices, and enforce park rules and regulations.

Conclusion

The concept of tourism carrying capacity can be approached from several perspectives: biological and ecological, sociological, physical, behavioural, planning and design, and policy. The issue of environmental carrying capacity has featured more frequently in tourism development plans and is closely related to the concept of sustainable development through the idea that there are identifiable capacity limits which should not be breached in the presence of present and future generations. The range of options to measure environmental carrying capacity can include checklists, inventories, guiding criteria, environmental standards, perception studies, and consistency issues and values (Coccossis and Parpairis, 1996).

Overall, the comments on tourism carrying capacity by Anon (1986) remain valid, "[T]he determination of tourism carrying capacity is an essential prerequisite to environmentally sound tourism development. It is extremely difficult to determine and no single formula is applicable for the different types of tourism. Nor can a single formula by applied to the same type of tourism in different parts of the world because of different geographical, ecological and even political, economic, social and cultural conditions." The determination of carrying capacity itself is the responsibility of the regional or national authorities. The subsequent development of policies and plans and their implementation should always be undertaken in close collaboration with the 'industry', i.e. tour operators, airline companies, hotel owners and associations, etc., as well as the local inhabitants who are most likely to be affected...The concept of carrying capacity however is far less understood."

Despite the research problems, the lack of cooperation from the industry, and many other constraints, the tourism carrying capacity should not be ignored. In his assessment of the tourism carrying capacity, Butler (1996) has called for a return to identify maximum appropriate numbers of visitors and tourists. This is the challenge for tourism researchers.

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Development Planning and Environmental Capacity

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Abstract

The Environment Agency is the UK government regulator in England and Wales responsible for the protection and management of the water environment, with an overarching duty to further sustainable development. Increasing pressures on the environment from proposed development prompted the Agency to develop an environmental capacity method which links the science of environmental management with spatial land use planning and better informs the decision-making process for locational choices for development to minimise environmental impacts. The method is set within a framework of environmental objectives and is based on a sequential process using threshold assessments to identify a score for each relevant factor. Management options are identified to help guide development planning to minimise environmental impacts. The method is described and work in progress is reported on a joint project between the Environment Agency and the South East England Regional Assembly. This study is investigating the potential constraints that climate change and flood risk, water resources and water quality may pose on the development of the region.

Introduction

The responsibilities for planning and managing the water environment within England and Wales rest primarily with the Environment Agency. Other organisations also have duties relating to the water environment and these include the Department for Environment, Food and Rural Affairs (DEFRA), which is responsible for regulation of marine activities including shipping and fisheries; local authorities; the Highways Agency; and water companies. Regulation is provided through a complex variety of UK and European legislation which is shortly to be integrated through the implementation of a new European Water Framework Directive (EC, 2002). This aims to manage the water environment with a more holistic approach based on River Basin Management Plans. All the various organisations involved in water management have an overarching duty to further sustainable development as required by the UK Strategy for Sustainable Development (DETR, 1999). The complexities of the varying duties and responsibilities have led to the different organisations seeking partnership working arrangements for mutual and synergistic benefits. Nonetheless, there remains a plethora of different plans and strategies associated with the water environment and this presents challenges for strategic forward planning.

England, and in particular the south-east region, is under considerable development pressure. Development planning is controlled by government policies and guidance. The type and location of development is allocated in accordance with development plans prepared at regional, subregional, and local levels. These plans consider development constraints, such as ecological designations to protect natural resources and regeneration/development needs.

Local authorities are finding it increasingly difficult to reach decisions on proposals for development planning, and locational options for new development due to the constraints and pressures associated with environmental protection, the use of natural resources and the predicted effects of climate change. They need expert advice and information from others. The Environment Agency is a statutory consultee in the planning process and needs to ensure that its responsibilities and interests, such as sustainable water management, are taken into account. Accordingly, the science of environmental management and the geography of spatial planning need to be integrated. Therefore, the Agency instigated a national research and development (R&D) project to investigate the capacity of the environment to absorb land use change in the context of development planning and control.

Developing and Testing the Environmental Capacity Method

The research team comprised of environmental and planning practitioners and researchers from Enfusion and the University of the West of England (UWE). An initial feasibility study (Environment Agency, 2000) identified the most appropriate approach for the Agency to take with regard to environmental capacity (EC). The methods and experience of EC were reviewed and an approach was developed. This built upon earlier work by UWE (Barton et al., 1995) which followed the criteria-based approach as used in the strategic environmental assessment (SEA) of plans and programmes and aimed to help with locational choices for proposed development.

The approach proposed for the Environment Agency was a sequential process set within the framework of environmental objectives and appropriate environmental boundaries. Relevant environmental issues (criteria) are scoped and thresholds at which an environmental impact occurs are identified together with a score. The Spectrum Score ranges through five categories from red (absolute constraint to development) to blue (development encouraged because it will resolve an existing environmental problem). For the orange and yellow categories with predicted impacts on the environment, management options may be identified in order to offer choices for mitigating impacts and thus increasing environmental capacity.

It had been agreed that the method should be compatible with existing techniques in both spatial planning and environmental management. SEA methodologies have tended to develop into sustainability appraisal (SA) methods in the UK where an SA is now required of each development plan. The sequential approach is also compatible with traditional planning techniques of constraints mapping and recent government guidance regarding urban capacity and the re-use of brown field land in accordance with Planning Policy Guidance Note 3 on housing (DETR, 2000a).

The proposed approach to EC was tested through workshops and the method further refined through seminars and discussions. Pilot studies were organised in different parts of the country including both coastal and inland locations to reflect differences in environmental characteristics and sensitivities, as well as testing at different levels of administration and planning: regional, subregional (county), and local (district). The pilot studies were carried out during 2000-2001 and were designed to test the impacts of additional proposed housing which is the greatest development pressure currently in the UK. The EC method was used to test the relative risk of environmental impacts from land options available for housing allocations in order to assist with decision-making. An additional pilot study was carried out to investigate the application of the method to a single development site. This also presented the opportunity to consider the approach from the developer's viewpoint.

Initially the pilot studies were intended to develop the method for the Agency in consultation with local authority planners. However, such a positive interest was received from the local authorities, who wanted to be more involved, that several workshops were held with staff from both organisations. Whilst the approach to EC and threshold assessment was the same for the Agency and the local planning authorities, the details of the method developed slightly differently for each according to their different requirements.

The workshop participants included engineers, scientists, and planners working in water resources, flood risk management, contaminated land, pollution control, ecology and fisheries, recreation, and navigation from the Agency. Participants from the local authorities included staff working in planning, conservation, building regulation, and economic development/regeneration. The workshops tended to follow a similar pattern. During the morning sessions objectives, boundaries, and thresholds were identified and during the afternoon, the scoring system was applied to the potential development sites. Participants worked in small groups using professional experience and readily available information and the groups were designed to encourage cross-functional debate.

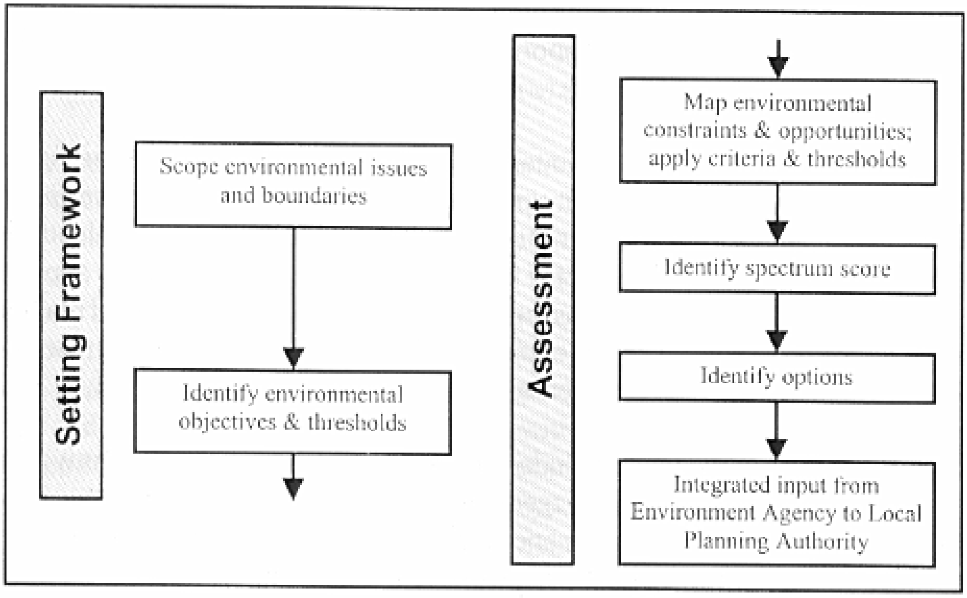
The Threshold Assessment Method

The original feasibility study considered the "carrying capacity" of catchments. However, this terminology was considered to be too associated with ecological capacity studies and the wider "environmental capacity" term was used during the pilot studies. The feedback from these pilot studies was that many of the participants at the workshops felt that EC was too absolute as a concept and preferred the name of "threshold assessment" for a method to be applied to spatial planning.

The role of the Environment Agency is illustrated in summary in Figure 1. The flow diagram shows the sequential process which produces an integrated input to the planning process for those environmental factors within the remit of the Agency. The local authority can work in the same way with economic and social factors in order to address the whole sustainable development agenda.

The first stage is to scope the relevant environmental issues and identify the appropriate boundaries for the particular study, which could be assessing different sites for proposed development or assessing the form of development for a chosen

Figure 1. The Environment Agency's Role.



site. Typical environmental issues include energy, waste, air quality, biodiversity, soil quality, flood risk, coastal erosion, groundwater vulnerability, river water flow/ quality and fisheries. A particular feature of this EC threshold approach is the identification of appropriate environmental boundaries. For example, water resources are suited to a regional level, whereas water quality may be more appropriate at the local level. Catchments and catchment management plans rarely coincide with administrative boundaries, whilst development/planning decisions reflect political boundaries. This approach offers a means of considering the characteristics of the receiving environment and its sensitivity to change (i.e., capacity to absorb). Both spatial and temporal boundaries are identified. Development planning tends to be defined within cycles of 5, 10, and 15 years, whereas environmental impacts and sustainability issues need to be considered over longer periods of time such as 50 years. The method thus demonstrates how different elements of the environment have different boundaries.

For each relevant environmental issue, objectives are identified. This recognises regional differences and environmental characteristics such that the process is dynamic and can reflect changing capacities. For example, one region may have sufficient habitat to support otters and the objective might be to maintain the otter populations; another region may still be trying to restore such river habitats and the objective might be to increase otter populations. Clearly, the capacity to accept land use change will be different.

For each issue, thresholds are identified at which there is a consequence or an impact on the environment which requires options to avoid or mitigate. The thresholds may include both qualitative and quantitative measurement. The Environment Agency has to work with certain absolute values as required by legislation, for example, chemical and biological standards for coastal, estuarine, and river waters. However, there are other criteria, such as aesthetic and heritage values of the environment, for which only qualitative measurement is available.

Identification of issues, boundaries, objectives, and thresholds sets the framework for the subsequent assessment of environmental capacity. From the pilot studies, it was found that this was most efficiently achieved, initially separately, by those professionals working in each environmental aspect and having experience of the geographical area under study. Subsequently, representatives from each environmental function work together to compile an integrated response for the environment. Thus, the approach does not weigh one criterion against another (which implies trade-offs) but rather identifies an acceptable bottom line for each criterion and the approach is about finding an integrated solution. Figure 2 is an example of a working matrix from flood risk experts.

The assessment against each criterion recognises five levels of "environmental capacity" for proposed development represented by colours as shown in Figure 3.

Figure 2. Example of Working Matrix.

Criterion:	Flood Risk Management	
Thresholds	•	Flood risk area; clear evidence of risk
	•	Flood risk area; options to orientate development
	•	Flood risk area; compensation possible
	•	No flood risk; no surface water discharge problems; existing flood problems resolved by new development
Management Options	Provision of compensatory floodplain storage Sustainable drainage systems River restoration and habitat creation	
Interactions	Water quality Biodiversity Amenity and recreation	

Figure 3. The Spectrum Approach.

Red	Absolute environmental constraints to development, e.g. national designations
Orange	Problematical and improbable because of known environmental issues, e.g. area of water shortage; mitigation or negotiation difficult and/or expensive
Yellow	Potential environmental issues; mitigation and/or negotiation possible, e.g. provision of Sustainable Drainage Systems
Green	No environmental constraints and development acceptable
Blue	Development actively encouraged as it would resolve an existing environmental problem

Earlier studies had worked with three colours but it was found that five were needed; any more would make the method unnecessarily complicated.

For assessing the relative capacity of alternative sites to accept proposed development, the score for each site is identified as shown in Figure 4.

Development requirements may be accommodated if there are enough scores of green and blue. If not, the approach identifies what further work will be needed to overcome problems. An orange score suggests that major reconsideration is needed, although high costs of mitigation may be acceptable if the socio-economic needs are justified. A yellow score indicates that environmental issues may be satisfactorily mitigated, for example, through the use of sustainable drainage systems to mitigate predicted exceedance of floodplain storage capacity. The approach also identifies where further studies may be needed, for example, where there was uncertainty or insufficient information in the assessment process. Such situations would score a yellow and require further investigation.

The approach thus provides a systematic method of informing local planning authorities and developers about the capacity of the environment to absorb land use change. Information about the social and economic aspects of proposed development may be compiled in the same way by others and added to the environmental information such that proposals may be tested through sustainability assessment procedure. Accordingly, environmental factors are not traded-off but rather integrated into the sustainable development agenda.

Summary Conclusions from the R&D Studies

The key conclusions from the research (Environment Agency, 2002) may be summarised as follows:

- the threshold assessment and spectrum approach offer a systematic and transparent process for assessing the capacity of the environment to accept change arising from development
- the process itself affects integration of environmental aspects for the Environment Agency and encourages partnership working with others
- the Environment Agency can better inform the land use planning system more proactively through more strategic and earlier influence
- the approach identifies the appropriate environmental boundaries which allow consideration of strategic and cumulative effects
- opportunities for both environmental enhancement and constraints for environmental protection are identified

Figure 4. Example of Spectrum Scores for Comparison of Alternative Sites for Proposed Development.

Criteria	Development potential/locational options						
	A	B	C	D	E	F	G
Transport	·	●	●	●	●	·	●
Energy	●				·		
Waste Management			●	●	·		·
Water Resources					·		
Flood Risk	●		·	●	●	·	·
Biodiversity	●	●		●		●	·
Air Quality	●			·		·	
Contaminated Land		·	●	●	●	·	·
Amenity and Recreation	●	·				●	·
Access							●
Fisheries				·		●	
Heritage		·		●		●	

- the method is compatible with other environmental planning tools including EIA, SEA, and Cumulative Effects Assessment
- the approach offers a simple, method that is, easy to understand and manage, and is able to identify areas where further detailed studies may be needed

The South East of England Regional Study

The South East region of England, bordering the North Sea and the English Channel, is one of the most densely populated areas in Europe. The region has significant economic importance and it is critical to the performance of the UK as a whole since it is the second largest regional economy after London. It also has one of the fastest growing and most economically active populations in the UK, with associated development pressures including significant housing and transport. The South East of England Regional Assembly (SEERA) is responsible for preparing regional planning guidance (DETR, 2000b) and it identified natural resources and climate change to be the major issues facing the region. The climate of the southeast is already changing and greater climate changes are predicted (DETR, 2002); wetter winters, drier summers, and more extreme weather with increased storminess. Around the region's coastline the sea level is rising, threatening important coastal habitats and increasing the risks of coastal flooding. The situation is compounded by geological tilting to the southeast.

Sustainable water management will be a key issue for future development of the region and SEERA with the Environment Agency instigated a joint project to investigate the potential constraints that flood risk, water resources, and water quality may pose upon development. Environmental capacity, as described previously in the threshold approach, is being used as one of the techniques to assess the relative constraints of development options, together with opportunities for mitigation and environmental enhancement, in order to develop planning policies and management options. Emerging results suggest that subregional policies will be required to recognise the differences in the receiving environments and needs of development.

Conclusions for Environmental Capacity

The studies have demonstrated that it is possible to integrate the science of environmental management with the geography of spatial land use planning. A method has been developed which assesses the capacity of the environment to accept land use change. The use of thresholds with a spectrum approach offers a means to integrate environmental factors within a sustainability framework which informs the decision-makers in the land use planning system. It goes further than traditional sieve and constraints mapping by identifying options for environmental mitigation and enhancement. The method is simple to use and understand, transparent, and systematic. It offers an initial overview and identifies the need for further studies, thus being efficient in resource demands. Ongoing studies demonstrate that the value of the environment can be integrated with economic vitality and social well-being for strategic planning in a coastal region with significant natural assets and under considerable development pressure.

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