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IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP
Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP)

Monitoring the ecological effects of coastal aquaculture wastes

GESAMP REPORTS AND STUDIES No. 57

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MONITORING THE ECOLOGICAL EFFECTS OF COASTAL AQUACULTURE WASTES

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS

Rome, 1996

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Courtesy of Dr M.C.M. Beveridge, Institute of Aquaculture, University of Stirling, Stirling, United Kingdom

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PREPARATION OF THIS STUDY

This study has been prepared on the basis of the work of the GESAMP Working Group on Environmental Impacts of Coastal Aquaculture.

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EXECUTIVE SUMMARY

Soluble inorganic and particulate waste from coastal aquaculture farms can result in organic enrichment of the local aquatic environment. To prevent unacceptable changes to the environment, an environmental management framework should be established as a means of regulating development and evaluating potential impacts before permission to develop is granted. However, any environmental assessment and monitoring effort should be related to the scale of perceived impact of a given aquaculture operation. An Environmental Impact Assessment should be undertaken to predict significant potential impacts, and monitoring carried out (once production has begun) to detect and evaluate the scale of impact. Monitoring is therefore part of the regulatory process which ensures that ecological change associated with aquaculture waste is kept within pre-determined, acceptable levels. Monitoring programmes may provide the information base for decisions to allow for further expansion or development should measured levels prove that observed ecological change is below unacceptable limits. Accordingly, information from monitoring can be essential for deciding whether or not to allow the expansion of existing aquaculture operations.

Successful monitoring will depend on a baseline survey being carried out in conjunction with an Environmental Impact Assessment. The purpose of the baseline survey is to obtain data which can assist in designing an appropriate monitoring programme, and to provide reference data against which changes caused by farm waste can be measured. To optimise resources, the level of monitoring (number of variables and frequency of monitoring) should be related to the size of the operation and the sensitivity of the receiving water body. Additional elements of monitoring programmes which need to be given careful consideration include: selection of reference stations; standardisation of sampling and analytical procedures; analysis and interpretation of data. Given that a particular monitoring programme should be matched to the size, type and location of a coastal aquaculture installation, it is not appropriate to recommend standard monitoring programmes. However, a range of variables commonly used in monitoring are discussed together with an evaluation of their value in interpreting changes resulting from release of waste from farms. To illustrate how particular monitoring programmes might be developed, five example scenarios are presented.

CONTENTS

ABSTRACT

GESAMP (IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection. Monitoring the ecological effects of coastal aquaculture wastes. *Reports and Studies, GESAMP.* No. 57. Rome, FAO. 1996. 38 p.

Unacceptable ecological change associated with wastes from coastal aquaculture farms can be minimised by good management practices. Any environmental assessment and monitoring effort should be related to the scale of perceived impact of a given aquaculture operation. An environmental management framework should include an environmental impact assessment (involving the use of predictive models) to quantify significant potential impacts and design a monitoring programme. Flexibility of monitoring undertaken for regulatory purposes is necessary, so that monitoring effort is related to the scale of development and sensitivity of the receiving water body. The choice of which variables to monitor must be based on the nature of the impact and the interpretative value of particular variables. Additional elements of monitoring programmes include: selection of reference stations; standardisation of sampling and analytical procedures; analysis and interpretation of data; feed back mechanisms. Five hypothetical scenarios are presented to show how monitoring programmes might be designed.

Key words: Coastal aquatic pollution, aquaculture waste, environmental impact assessment, monitoring

1. INTRODUCTION

This report discusses scientific aspects of the monitoring required to assess and manage the ecological effects of coastal aquaculture wastes from the perspective of environmental protection. Other ecological issues relating to coastal aquaculture are discussed by Rosenthal *et al.* (1988), Gowen *et al.* (1990) and Weston (1991). In addition to ecological issues, Chua *et al.* (1989), Chua, (1992 and 1993), GESAMP (1991a), Barg (1992) and Pullin *et al.* (1993) discuss the social and economic consequences of coastal aquaculture development. The scope of this report is restricted to particulate and soluble waste and does not include consideration of the chemicals used in aquaculture (Alderman *et al.*, 1994; Honculada Primavera *et al.*, 1993; Baticados and Paclibare, 1992; OIE, 1992; Smith *et al.*, 1994; Weston, 1996).

Production of fish and crustacea generates particulate organic waste (faecal material and uneaten food) and soluble, inorganic excretory waste. Of the different types of coastal aquaculture, intensive production (which relies entirely on provision of external feed inputs) has the greatest potential to generate waste. It has been estimated for example, that production of one tonne of Atlantic salmon in Scandinavian countries generates ≈ 80 kg of soluble nitrogen (ammonium), ≈ 7.5 kg soluble phosphorus and ≈ 1300 kg particulate carbon (see Ackefors and Enell, 1994, and references cited therein). Certain types of intensive shrimp cultivation can generate ≈ 1500 kg total nitrogen and 400 kg total phosphorus per hectare of pond per year (Phillips, 1995; see also Briggs and Funge-Smith, 1994). Bivalve culture generates faecal and pseudofaecal material (biodeposits) and large scale culture can generate considerable quantities of organic particulate material. Grentz *et al.* (1991) have estimated that approximately 600 kg of particulate biodeposits (faeces and pseudofaeces) are generated for each tonne of production (measured as wet weight).

The waste from coastal aquaculture operations has the potential to enrich aquatic ecosystems, particularly when farms are located in semi-enclosed coastal basins which have restricted exchange with more open coastal waters. Particulate waste settles to the sea-bed and has been shown to bring about changes in the community structure of the benthic macrofauna (Brown *et al.*, 1987; Ritz *et al.*, 1990; Weston, 1990) as well as physical and chemical changes in the sediment, (Brown *et al.*, 1987; Dahlback and Gunnarson, 1981) which, in extreme cases, can result in the sediment becoming completely anoxic (Brown *et al.*, 1987) and oxygen depletion of bottom water (Tsutsumi and Kikuchi, 1983). Soluble excretory waste (ammonium and phosphorus) can result in nutrient enrichment (Gowen and Ezzi, 1992) which in a few cases has been reported to have resulted in eutrophication (enhanced phytoplankton production) in coastal waters (Persson, 1991).

To satisfactorily manage the scale of enrichment and ensure that ecological change does not exceed pre-determined and acceptable levels, a management framework should be adopted prior to development. Such a framework should include the establishment of Environmental Quality Objectives (EQOs) and Standards (EQSs) and must include scope for an Environmental Impact Assessment (EIA) and a monitoring programme. The latter is undertaken to detect ecological change associated with waste production when the farm has commenced production. For the purposes of this report, a working definition of this type of monitoring is given as:

> *"the regular collection, generally under regulatory mandate, of biological, chemical or physical data from pre-determined locations such that ecological changes attributable to aquaculture wastes can be quantified and evaluated."*

It is recognised however, that monitoring may be undertaken for other reasons as outlined in Table I. Section 2 of this report discusses monitoring in the context of an environmental management framework and Section 3 reviews models which have been used to predict the effects of coastal aquaculture waste. The basic principles which should be considered during the design of monitoring programmes are discussed in Section 4 and the variables most commonly measured in monitoring programmes together with an assessment of their interpretative value are presented in Section 5. Given the broad range of coastal aquaculture practices and the need to structure a monitoring programme in relation to the anticipated ecological effect(s), it is not possible to recommend the details of individual programmes. In Section 6 therefore, hypothetical scenarios have been used to show how monitoring programmes can be designed to detect changes in the natural environment.

Table I

The purposes for which monitoring at coastal aquaculture operations may be undertaken.

(i)	Regulation (a) compliance with the terms of the licence (b) protection of the natural environment (ecological protection) (c) safeguarding water quality for aquaculture (e.g. monitoring dissolved oxygen in ambient waters)
(ii)	Farm Management (a) optimising husbandry practice (b) control of water quality (e.g. dissolved oxygen, out gassing from sediment to safeguard farm stock) (c) limiting interference from other aquaculture operations
(iii)	Public Health (a) protection of product quality (e.g. bacterial, chemical or natural toxin contamination) (b) control of disease transmission
(iv)	Research (a) identification of unexpected impacts (b) validation of models (c) monitoring methodology

2. MONITORING THE EFFECTS OF AQUACULTURE WASTES IN THE CONTEXT OF A MANAGEMENT FRAMEWORK FOR COASTAL DEVELOPMENT

It is important to view monitoring as a component of a wider management framework for managing and protecting the natural environment, an example of which is shown in Figure 1. With respect to the impact of aquaculture wastes, the key elements of the framework in Figure 1 are:

i) defining goals or use(s) for a water body
ii) establishment of Environmental Quality Objectives (EQO)
iii) establishment of Environmental Quality Standards (EQS)
iv) undertaking Environmental Impact Assessment (EIA)
v) monitoring
vi) review and evaluation (auditing)

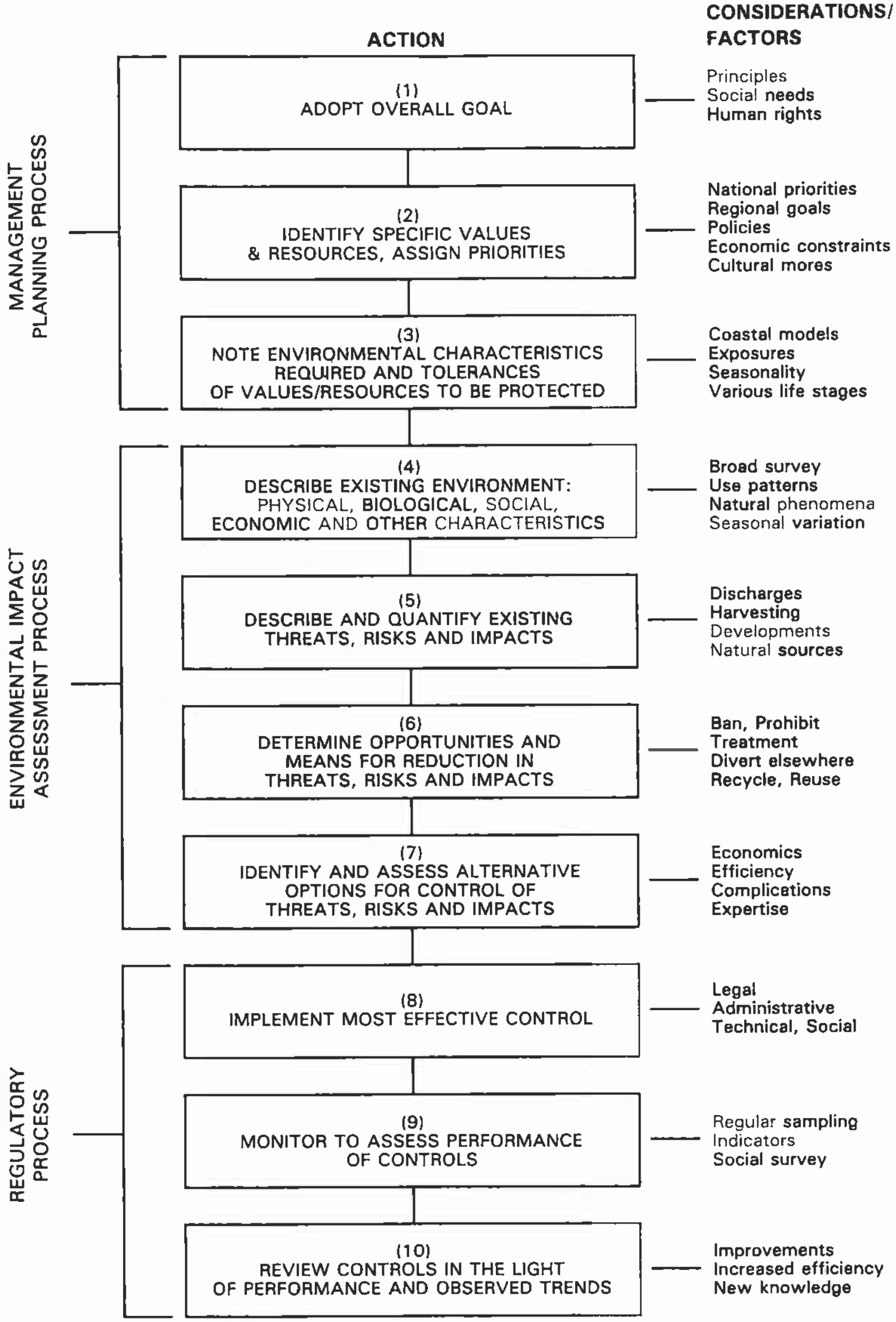


Figure 1. An example of a comprehensive management framework for protection of the marine and other environments. (From GESAMP, 1991b)

Management objectives (based on intended uses) for specific coastal waters might be to maintain the natural environment, or alternatively, to develop activities such as tourism/recreation, aquaculture, fisheries, logging, gravel extraction, etc. Such uses are not necessarily mutually exclusive. The management process should provide for the equitable use of the aquatic resource and safeguard the natural environment by setting Environmental Quality Objectives. These objectives define the conditions to protect a particular use. Environmental Quality Standards are levels of particular variables associated with that use which may be imposed to ensure that the objectives are not compromised. An example of the EQO/EQS approach related to the protection of the natural environment is given in Table II.

Table II

An example of the way in which environmental quality objectives and standards can be developed in relation to a defined use of a water body.
In this example the defined use is protection of the general ecosystem.
(Modified from MPMMG, 1992)

Environmental Quality Objective	Criteria and standard or approach to standards	
	Criterion	Standard
Maintenance of environmental quality so as to protect aquatic life and dependent non-aquatic organisms, such that the ecosystem is typical of coastal waters with those physical characteristics and latitude (i.e. biotic characteristics)	Faunal benthic composition	Either (i) faunal composition not to be altered by a set quantifiable change in a suitable biotic index. (Further research is required to define this standard), **OR** (ii) outside of a defined impact zone, faunal composition shall not be significantly different from a control site.
Beyond the immediate farm area, the chemical quality of the receiving environment will be indistinguishable from that of the adjacent marine or brackish water environment	Eh (redox potential) Sediment carbon content	Outside an agreed zone where impact is accepted as inevitable, Eh and sediment carbon content shall not be significantly different from that of selected control sites
	Dissolved oxygen level in water column	Should not fall below 7 mg/l (except in cases where deoxygenation is due to other causes)
	Nutrients	Further work required to define the mechanisms linking nutrients with algal growth

There are a number of definitions of what EIA is and some debate concerning what EIA involves or should involve (see for example, Ahmad and Sammy, 1985; Carpenter and Maragos, 1989; Jernelöv and Marinov, 1990; UN/ESCAP, 1991; Sorensen and West, 1992; Barg, 1992).

It is generally accepted however, that EIA should encompass the evaluation of social, economic and ecological impact of a proposed development as well as the identification of impact mitigation measures and alternative development options (GESAMP, 1991a; Pullin *et al.*, 1993). For the purposes of this report, consideration is only given to that part of the EIA which relates to the evaluation of ecological change associated with aquaculture wastes.

From an ecological perspective, the purpose of an EIA must be to assess or predict the ecological consequences of the planned facility, thereby providing the regulatory authority with the scientific basis to determine the acceptability of the perceived impacts relative to pre-determined standards. EIAs have frequently consisted of collections of largely descriptive data, with little *a priori* consideration of the specific changes to be expected from the proposed development. At best, if comparative data from other operations in similar locations are available, such an approach may provide a semi-quantitative evaluation of potential impact. Often, however, the data obtained serve little more than a basis for a case history, giving little insight into cause and effect in relation to eventual impacts. It is suggested that specific, testable hypotheses be formulated at an early stage in the EIA process. Numerical models have the potential to provide quantitative predictions (Section 3) and are therefore a useful tool in quantifying impacts resulting from aquaculture waste. A combination of field investigation and modelling is recommended since this will introduce scientific rigour into the evaluation process.

Thus, the output from models might be used to develop hypotheses regarding the extent of ecological change which are tested using data collected during the monitoring programme. The use of models may also be cost-effective by reducing the amount of field investigation required, and may help to illustrate impacts in ways which are more readily interpreted. Assessments incorporating modelling and hypothesis testing also facilitate feedback processes which allow results of assessments and monitoring to be used more effectively (see Item 4.8 "Feedback").

Data and model predictions from the EIA are needed to design efficient monitoring programmes. A proposed plan for integrating modelling, monitoring and management of benthic enrichment associated with intensive fish farming in Norway is outlined in Figure 2.

The example in Figure 2 illustrates one way of using predictive models to determine the level of exploitation of a particular site and the scale of monitoring which should be undertaken. As part of an EIA, a dispersion and loading model is used to predict the input of organic carbon waste to the sea-bed. The model prediction is compared to an EQS and the degree of exploitation (maximum size of farm) decided. On the basis of this, one of three levels of monitoring is imposed. The results of the monitoring programme are used to assess the EQS, evaluate the model predictions and ensure that the degree of exploitation does not compromise the EQS.

In most cases, an EIA has not been undertaken prior to development of an aquaculture operation (see for example, Burbridge *et al.*, 1993). In the absence of a management framework with pre-determined standards and an EIA to define the scope of the monitoring programme, many monitoring programmes have failed to fulfil their function. On occasion, monitoring has been imposed as a result of public pressure because of the perceived ecological damage caused by aquaculture wastes. The result has often been: the measurement of a wide range of ecological variables, many of which are inappropriate; the collection of data which are difficult to interpret; failure to analyse and interpret data and implement feedback mechanisms to modify farm practice/production and the monitoring programme itself. The converse is also true. When socio-economic and political pressures have superseded ecological concerns, only limited monitoring has been undertaken and the resultant ecological change has sometimes had adverse consequences for the sustainable operation of farms. Unfortunately, in many countries there are budgetary, manpower and organisational constraints which often restrict the implementation of monitoring programmes.

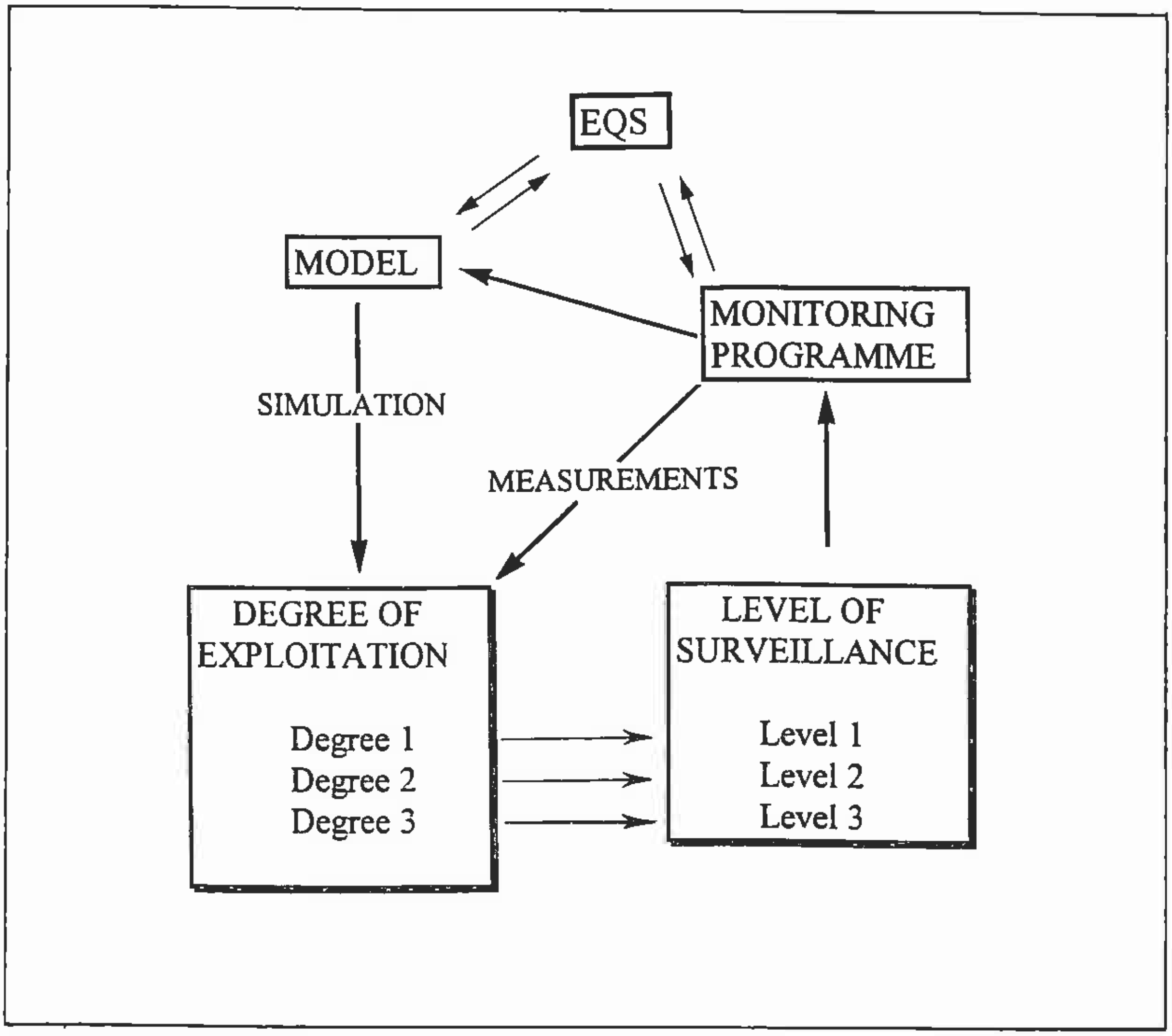


Figure 2. A schematic illustration showing the relationship amongst modelling, monitoring and farm size (degree of exploitation). (From Ervik and Kupka Hansen, 1994)

3. THE USE OF MODELS IN ENVIRONMENTAL IMPACT ASSESSMENT

The use of EIA in the management of coastal aquaculture development requires the application of ecological knowledge. In this context, resource managers should be seen as practitioners who use the best available knowledge, and are able to justify their decisions to the community influenced by those decisions. Models can be an important tool in management both for predicting impacts and as an aid in the design of monitoring programmes. With respect to wastes from aquaculture operations, there are three processes to model: the quantities of material generated; dispersion after release or discharge; biological consequences.

Models may be empirical or mechanistic. The former are based on a statistical relationship between variables derived by observation, and do not necessarily require any understanding of the underlying principles. Mechanistic models describe the relationship between cause and effect with the expectation that all variables have significance within the natural system. Complex ecosystem models are typically mechanistic and have generally been developed in relation to large-scale or multiple developments. With increasing levels of model sophistication, the amount of data required to parameterise and validate the model increases and this is likely to involve high cost. Where complex ecosystem models have been used to manage water bodies, it is clearly appropriate to include aquaculture as an additional source of dissolved and particulate waste. In most instances, however, given the low risk of large-scale ecological change from aquaculture wastes, there is little justification for the use of complex ecosystem models except in areas where there is likely to be large-scale development. For these reasons, it is suggested that complex ecosystem models are more appropriate for research rather than use as management tools (see also GESAMP, 1991c).

There is debate over the advantages and disadvantages of different types of model (see for example, Ahlgren *et al.*, 1988) and regulatory authorities are therefore faced with a choice of which type of model to use. The choice must be based on the objectives of the regulatory authority in terms of managing coastal ecosystems. These objectives need to be clearly defined, and the limitations of any model which may be used must be recognised. For example, simple models may suffice for the general management of a number of semi-enclosed coastal water bodies (e.g. coves) of similar topography and hydrodynamics or of larger, more exposed coastal areas but should not be expected to yield predictions about transient extremes such as periods of low oxygen concentration or algal blooms. A sequential approach should therefore be adopted. That is, to use simple models as a first stage and if the results of the investigation indicate that such models are inadequate, the second stage would be to evaluate the conceptual framework and, if necessary, increase the complexity of the model. Simple models may therefore provide an efficient means of screening for potential ecological impacts from aquaculture and other developments in coastal environments.

The use of models does not preclude the need for field observations. Data are required to initialise the model and validate predictions. The latter must be regarded as an integral part of using a model as a predictive tool. All model predictions will have an associated error, the acceptability of which will usually depend on the perceived risk. In the case of a development where the expectation is for minimal ecological change, the use of a model may be little more than a means of providing a formal evaluation of the potential impact. As the size of the farm or scale of development increases, the potential for ecological change increases, and the consequences of error in model predictions become more significant. It is essential to avoid predicting no impact when one will in fact occur ("Type II error"; see Carpenter and Maragos, 1989; Morrisey, 1993; Suter, 1993). In such cases, it is usual to proceed in a precautionary manner. Resource managers will typically choose to regulate on the basis of unfavourable initial conditions of low probability (commonly termed "worst case") rather than the most probable initial conditions (i.e. use neap tides rather than average tidal range; use maximum summer temperature rather than mean summer temperature).

Most existing models used to quantify the ecological effects of aquaculture wastes have been developed and tested in temperate waters. The rapid development of tropical coastal aquaculture (particularly shrimp farming) requires the urgent adaptation and validation of existing models. A first step in this direction would be to test the applicability of simple empirical models to tropical systems to save time and resources in ecological assessment.

3.1 Eutrophication

Eutrophication, the biological consequence of nutrient enrichment, may be manifest as changes in the biomass and community structure of phytoplankton or macrophytes (algae and higher plants). To date, attempts to model the impact of soluble nutrient waste from coastal aquaculture has dealt solely with the response of phytoplankton in coastal embayments, in which the summer growth of phytoplankton is assumed to be controlled by nutrient (dissolved inorganic nitrogen) availability. The main components associated with developing such models include:

defining the basin or water body influenced by waste inputs; estimating dilution rate; calculating the level of nutrient enrichment; relating nutrient enrichment to the response by phytoplankton.

Working in coastal waters of the non-tidal Baltic Sea, Hakanson and Wallin (1991) used a multiple regression technique to derive an empirical relationship amongst sensitivity of coastal waters (dependent on the rate of exchange of water between the coast and more open sea), nutrient concentration, phytoplankton biomass and abiotic variables such as Secchi depth. Thus, depending on the sensitivity of the coastal region, a greater effect in terms of increased phytoplankton biomass or reduced Secchi depth is predicted. Some caution should, however be adopted in the use of Secchi depth as a measure of phytoplankton biomass. Secchi depth can be influenced by suspended particulate material (riverine or resuspended from the bottom sediment) in the water, and this is particularly likely in high energy coastal environments. For this reason, chlorophyll represents a much less ambiguous measure of phytoplankton biomass.

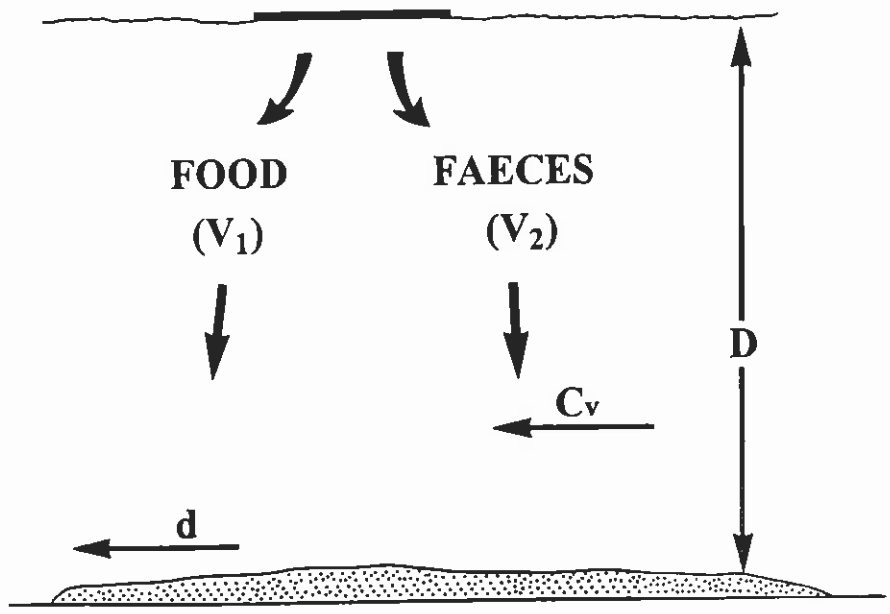
Gowen *et al.* (1992) investigated the relationship between phytoplankton chlorophyll and dissolved inorganic nitrate (plus nitrite) in Scottish west coast sea-lochs. Using linear regression analysis these authors defined the slope of the regression as the yield (q) of chlorophyll from nitrate in units of mg chlorophyll (mmol nitrate)$^{-1}$. The yield represents the ecosystem response to anthropogenic nutrients and is the net increase in biomass which might be expected taking account of losses due to factors such as zooplankton grazing and dilution.

As with empirical relationships in inland waters (Ahlgren *et al.*, 1988), Gowen *et al.* (1992) observed a range in the value of q (the median was 1.05 with 95% of the values between 0.45 and 4.21). This clearly presents a difficulty in deciding which value of q to choose, although these authors suggest that the median represents the average response for Scottish west coast sea-lochs. For management purposes, however, it may be more appropriate to use a loch-specific value of q. One of the problems associated with the use of q is in determining the probability of the yield being realised in a given water body. To address this question CSTT (1994) proposed the use of a simple algal growth equation to determine light controlled growth rate. This calculated growth rate can then be compared to losses caused by factors such as dilution and grazing and hence determine whether or not biomass will accumulate. This approach has been adopted as an initial screening tool to evaluate the need for secondary sewage treatment in U.K. coastal waters. The necessary steps in the procedure are shown in Scenario 2 of Section 6.

Pridmore and Rutherford (1992) developed a model to investigate the influence of nutrient enrichment (resulting from fish farming in a small coastal embayment in New Zealand) on phytoplankton. Rather than an empirical relationship, however, these workers used a phytoplankton growth model. The model successfully simulated phytoplankton biomass in the embayment only when growth was nitrogen limited. It seems likely, as suggested by Pridmore and Rutherford (1992), that when nitrogen was not limiting growth, other factors such as zooplankton grazing and sedimentation, which were not included in the model, controlled phytoplankton biomass. It is clear therefore, that it would be necessary to include more processes in the model. As suggested in the introduction to this section, however, the question remains as to whether this is a cost-effective method of management for such coastal embayments.

3.2 Sedimentation

Of all the ecological effects of coastal aquaculture, probably the most frequently reported and best characterised are effects on the benthic environment. Sedimentation models have been developed in order to predict the magnitude and spatial extent of particulate matter deposition. These models typically attempt to predict the trajectory of particulate wastes based on the hydrographic regime and settling velocities of feed and/or faecal matter (Figure 3). Current velocity and direction may be parameterised by multiple measurements collected at regular (e.g. hourly) intervals (Gowen *et al.*, 1989). Settling velocities of feed and faeces may be assigned mean values (Gowen *et al.*, 1989) or more realistically, should be treated as a probability distribution with a defined mean and standard deviation (Hagino, 1977).



Distance dispersed (d) = $D \cdot C_v / V(_{1 \text{ or } 2})$

Figure 3. A diagram illustrating the principle upon which simple, benthic deposition models are based. Where D is the depth of water beneath the cages, C_v the current velocity and V_1 and V_2 the settling velocities of uneaten food and faecal waste respectively. (From Gowen *et al.*, 1989)

Existing models vary in their degree of complexity and inherent assumptions (Gowen *et al.*, 1994), but model output is generally in the form of a contour diagram showing particulate matter deposition with reference to the location of the farm (Figure 4). Most existing sedimentation models are based on the trajectory of a settling particle and are designed for low-energy environments. As such, they do not address resuspension and any subsequent particle transport. In high-energy coastal environments there may be regular or episodic resuspension of sediments and some caution should be used in applying existing sedimentation models in these locations.

Existing sedimentation models make no predictions about the ecological consequences of a given loading. Linkages between predicted loading and biological response (e.g. changes in population densities, species number, a functional response of the community) are not yet established and are likely to be highly site-specific. Findlay and Watling (1994) have taken some initial steps in this direction by relating carbon loading with sea-bed oxygen consumption and CO_2 production, but much work remains to be done before sedimentation modelling can be used to quantify the loading that would maintain a given ecological state.

Cage culture of fish is readily amenable to sedimentation modelling and has been the subject of all such modelling to date. The fate of particulate wastes from suspended bivalve culture (e.g. mussels, oysters) could also be modelled using the same techniques. With respect to particulate waste from land based farms and hatcheries, existing models for sewer outfalls and pulp mill effluents, could be readily adapted for use.

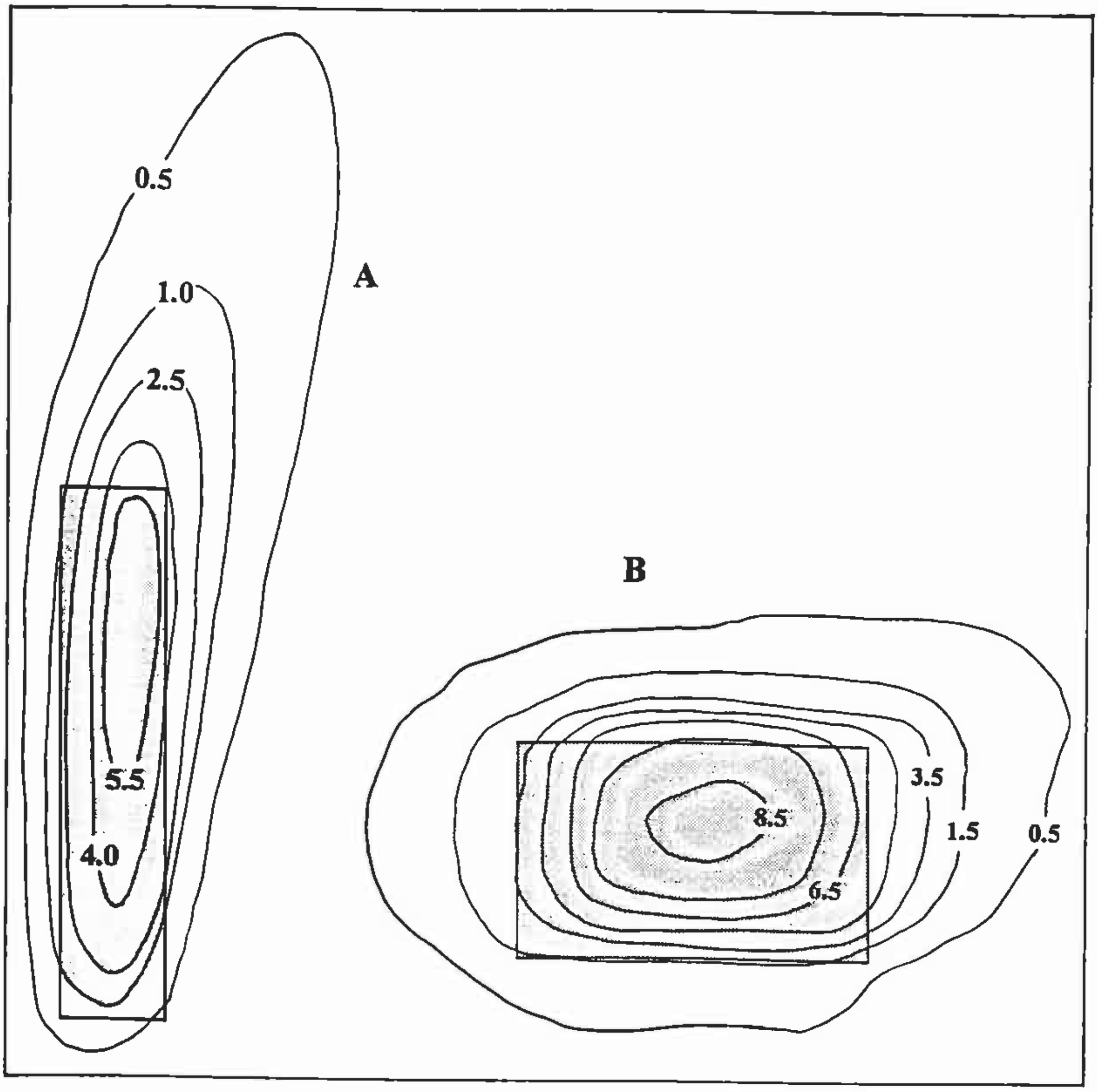


Figure 4. Two examples of the output from a simple, organic waste dispersion model. In both examples the shaded area represents the net-cages. Example A: water depth 21 m; maximum current speed 31.3 cm s^{-1}; constancy factor (an estimate of the persistence of the direction of flow) 97.3%; the long axis of the cage group represents 54 m. Example B: water depth 20 m; maximum current speed 15.6 cm s^{-1}; constancy factor 35.4%; the long axis of the cage group is 30 m. (From Gowen *et al.*, 1988)

There are five potential applications for sedimentation models:

(i) Site selection : For any given production level the model would be used to establish the deposition rate of particulate wastes at alternative sites with reference to established environmental quality standards.

(ii) Defining site limitations : In the converse application of the same models, it is possible to establish the maximum production attainable at a site, given a maximum permissible loading of particulate matter deposition.

(iii) Determining responsibility : Organic material is introduced into aquatic environments by a variety of natural and anthropogenic sources. If adverse ecological consequences of enrichment are observed, a sedimentation model could be used to determine the relative contribution from each source.

(iv) Optimizing production : Organically-enriched sediments can have adverse effects on growth, health and survival of the cultured species through mechanisms such as a reduction in dissolved oxygen concentrations, production of hydrogen sulphide, persistence of antibiotic residues or in serving as a reservoir for pathogenic micro-organisms. Sedimentation models can be used as an indication of the extent to which enrichment has a deleterious effect on production.

(v) Design of monitoring programmes : The magnitude of predicted carbon loading can be used to establish an appropriate intensity of monitoring (i.e. how frequently, number of variables measured). Model predictions of prevailing particle trajectories could be used to establish the locations of sampling sites required to identify areas of greatest predicted impact. Reference stations would need to be located at a greater distance from the farm when in a down current direction. Model predictions might also be used to establish the extent of a benthic mixing zone. Some regulatory authorities have accepted degradation within a specified area around a given farm and directed monitoring efforts at the perimeter of the mixing zone in order to verify compliance.

4. GENERAL PRINCIPLES OF MONITORING

There are a number of recent texts which discuss the design and application of monitoring programmes for coastal waters (see for example, National Research Council, 1990; Underwood, 1993). This section summarises the main features which must be considered when designing monitoring programmes to detect the effects of aquaculture wastes.

4.1 Environmental Capacity

The current state-of-knowledge often allows estimation of the likely scale of environmental effects from individual farms in coastal embayments or coastal areas. This permits the setting of effluent or receiving water standards and the design of an appropriate monitoring programme. However, as the number and size of the discharges increase, it becomes increasingly difficult for each site to be regulated independently and it becomes desirable to view the entire water body as a whole with a finite environmental capacity for pollutants.

Environmental capacity has been defined as: 'a property of the environment and its ability to accommodate a particular activity or rate of activity (e.g. volume of discharge per unit time, quantity of dredgings dumped per unit time, quantity of minerals extracted per unit time) without unacceptable impact' (GESAMP, 1986). In the case of aquaculture, the environmental capacity may be: the rate at which nutrients are added without triggering eutrophication; the rate of organic flux to the benthos without major disruption to natural benthic processes; or the rate of dissolved oxygen depletion that can be accommodated without mortality of the indigenous biota.

Quantification of environmental capacity is of value in coastal area management including aquaculture development and may afford better protection of the natural environment than current operation-specific standards. This is because environmental capacity allows for the evaluation of successive developments against one or more standards set for the coastal area in question. If the capacity of the water body for a particular substance can be determined, the permissible loading can be apportioned among all discharges. Under the current operation-specific standards, there is a risk that the cumulative effects of multiple pollutant inputs could cause ecological degradation of an entire water body. However, the environmental capacity approach has not been widely implemented in relation to the effects of aquaculture wastes. The reason for this is, in part, because it is difficult to link waste inputs with biological consequences.

4.2 Environmental Quality Standards

The majority of coastal aquaculture monitoring programmes now in place specify the variables to be measured (e.g., dissolved oxygen, ammonium, infaunal species richness) without defining a threshold of impact beyond which mitigation is necessary. Often the responsible regulatory body, or even society as a whole, has made no *a priori* determination of what constitutes an acceptable level of change in a variable, yet extensive and expensive monitoring of that variable may be required of the grower. At best, such monitoring is a data gathering process by which regulatory authorities may establish thresholds of effects (standards) at a later date. While such an approach may be necessary for new culture technologies, it often continues long after a substantial data base has been developed and, in effect, becomes monitoring for its own sake. Monitoring is most effective, and the natural environment best protected, if environmental quality standards are established and monitoring is designed to verify compliance with these standards.

For some variables, the literature provides adequate justification for a uniform, numerical standard. For example, permits for some net-cage farms in Washington State (USA) specify: "[At a distance of 30 m from the cages, dissolved oxygen shall not be below] a minimum of 7.0 mg l^{-1} or if [ambient is] less than 7.0 mg l^{-1}, no decrease below ambient of more than 0.2 mg l^{-1} ". Such standards can be applied to all farms over a broad geographical area.

To evaluate and limit biological change, biological objectives and/or standards are required. In relation to nutrient enrichment and eutrophication, for example, standards for nitrogen and phosphorus may be easier to define and measure but chemical variables are only surrogates for biological variables and their interpretation may be ambiguous. In this example, it is because nutrient enrichment is not an ecological problem in itself (except perhaps where unionized ammonia concentrations are high as in some coastal areas supporting intensive shrimp culture) and, nutrient enrichment will only stimulate phytoplankton growth when that particular nutrient is limiting (see Gowen and Ezzi, 1992). The establishment of a scientifically-based biological standard such as chlorophyll for eutrophication requires an understanding of the link between nutrient availability and phytoplankton growth and biomass. It further requires that the effects of changes in phytoplankton ecology on other trophic levels is known. This is generally not the case and predicting trophic interactions is at the limits of our understanding of how coastal ecosystems function.

Despite these problems, there have been some moves to set a chlorophyll standard. For example, in the U.K. a task team set up to undertake a comprehensive study in relation to Article 6 of DIR 91/271 EEC, (The Urban Waste Water Treatment Directive) suggested a chlorophyll standard of 10 mg m^{-3} for estuaries and more open coastal waters (CSTT, 1994). This standard is probably based on informed scientific opinion (rather than an understanding of the broader ecological effects of a phytoplankton biomass represented by 10 mg chlorophyll m^{-3}). Nevertheless, the Working Group considers that the formulation of biological standards, even working or tentative standards, is urgently needed.

With respect to organic enrichment of the benthos, there is an inadequate scientific basis to establish the ecological consequences of an observed change of a given magnitude in a variable (e.g., sediment total organic carbon, macro-invertebrate species richness). In those cases, nevertheless, a standard may be established relative to reference sites such as "the number of macro-invertebrate species shall not be less than 50% of the species richness at reference sites". Each farm, in effect, is regulated in accordance with a farm-specific standard. This approach of course assumes the availability of appropriate reference stations (see Item 4.4 below).

4.3 Baseline Studies

The aim of a baseline study is to provide a reference data set which can be used to:

a) design an appropriate monitoring programme including the location of sampling and reference stations, and

b) establish conditions of the local ecosystem (which may include assessment of temporal and spatial variability) against which future change can be compared.

Baseline studies are an essential element of a monitoring programme for regulatory purposes and must be undertaken prior to the establishment of a culture facility or expansion of an existing farm. The scope of the survey should take into account the expected seasonal variation as well as the size and potential impact of the planned facility. Where background data are available from other sites in the vicinity, or only a small farm is planned, a brief survey may suffice. Alternatively, if seasonal variations are likely to be important and have not been previously documented, the study may need to span at least a year. In British Columbia, Canada, special allowances have been made in the procedure for site approval to allow a proponent a full year to study seasonal variation without loss of his priority claim to the site (Black, 1991, Black and Truscott, 1994).

4.4 Reference Stations

Ecosystems are dynamic with considerable natural, temporal variability in physical, chemical and biological properties. As one of the primary purposes of monitoring is to quantify ecological change attributable to one or more culture operations, it is imperative to minimise the likelihood that change attributable to natural variability would be falsely identified as a consequence of culture. Recently there have been new approaches to sampling designs, such as the so-called Before-After-Control-Impact (BACI) designs (Stewart-Oaten *et al.*, 1986; Underwood, 1991 and 1993). An essential aspect is that there should always be a number of reference stations for comparison with impacted stations.

A further problem in relying upon a single reference site is that it may become impacted by anthropogenic activities other than the farm for which it is serving as a reference. This concern is particularly acute in regions where there is large-scale aquaculture development or multiple sources of pollution. For example, a second farm or other sources of organic enrichment may subsequently be located in close proximity to the designated reference station for the first farm. That station then no longer functions as a representative of undisturbed conditions and the first farm is thereafter evaluated in relation to a potentially degraded site. Substantial degradation over a broad area could theoretically occur even though no one farm violates environmental quality standards. Estimation of environmental capacity provides the best protection against such cumulative effects.

Ideally, reference stations should be identical to the site of potential impact in all respects except for the presence of the aquaculture facility itself. Reference sites may only be 50 m from the aquaculture facility in some benthic monitoring programmes, or may be more than one kilometre away when there is large-scale aquaculture development or when monitoring some water quality variables.

Despite the fact that the concept of a "control" is well-entrenched in scientific methodology, it is not unusual to find monitoring programmes which lack appropriate reference sites. This may be due to oversight, but more often results from unanticipated spatial variability that makes the intended reference sites dissimilar to the culture area in some critical characteristic. In benthic monitoring, for example, there may be differences in substrate type between the reference and potentially impacted areas that make any comparisons of macrofaunal communities of doubtful validity. In practice, however, the use of multiple reference sites is atypical and cost constraints often limit the investigator to a single reference site.

4.5 Delegation of Monitoring Responsibility

Monitoring to protect the natural environment is generally performed by one or more of the following parties: 1) a contractor employed by the farmer; 2) the farmer; 3) staff of a regulatory authority; or 4) a contractor employed by a regulatory authority. In developing countries, where monitoring is undertaken, it is usually done by the regulatory authority. In contrast, in most developed countries, the regulatory authority rarely does the monitoring but delegates this to the farmer. It is most common in such cases for monitoring to be done by a contractor selected and hired by the farmer. The local regulatory authority usually determines the details of the monitoring programme including sampling, analytical methodology, data quality assurance and undertakes limited audit monitoring as a form of quality control. Unfortunately, there are few countries where these consultants must be accredited for such work, with the result that there can be considerable variability in the efficiency and reliability of monitoring. Public confidence in monitoring as a mechanism to safeguard the natural environment depends on the perception that the monitoring programme is adequate and those responsible for its implementation are competent.

Frequently, the farmer/operator may undertake the monitoring, with occasional checking by the regulatory agency. This approach has advantages where measurements are required on a daily, weekly or monthly basis, where the cost of maintaining equipment and laboratory facilities as well as the cost of training staff becomes justifiable and travel costs for an outside investigator are prohibitive.

In the State of Maine (USA) fish producers are assessed a fee of one cent per pound of the whole weight of all fish harvested. These funds are used by the State to conduct monitoring either by State employees or through a single contractor. A team under the direct control of the regulatory agency moves from farm to farm. Such a team, with their own equipment, backed up by adequate laboratory facilities, provides not only experience and consistent methodology, but also considerable economy in equipment and laboratory facilities. Such an approach is attractive because it largely overcomes the public credibility concerns that inevitably accompany self-monitoring. It also ensures a high degree of standardisation in sampling and analytical methodologies for all farms and a uniform format and quality for monitoring reports.

An important means of ensuring that monitoring is undertaken properly and reliable data are obtained, is through a requirement for individuals and/or laboratories to be accredited for particular methods of sample collection and laboratory analysis and appropriate quality assurance procedures. In New South Wales, Australia, for example, all individuals involved in monitoring must be accredited before they are allowed to undertake such work.

4.6 Mixing Zones

The mixing zone concept has commonly been applied in environmental management of a wide variety of municipal and industrial discharges. Within the past few years mixing zones have been defined for some coastal aquaculture enterprises in the United States of America and the United Kingdom.

A mixing zone is the volume of receiving water within which it is permitted to exceed an environmental quality standard. Based on the expected degree of dilution, the discharge permit will specify the area of the mixing zone (e.g., 100 m radius from the point of discharge) which unless specified otherwise, includes the entire water column from the sea surface to the sea-bed. Compliance with environmental quality standards is determined by sampling at the edge of the mixing zone rather than at the point of discharge. A mixing zone is utilised as a regulatory tool when effluent treatment technology is unavailable, impractical, inadequate or prohibitively expensive and dilution by receiving waters is necessary before water quality standards are achievable. In establishing a mixing zone, the regulatory body implicitly accepts the possibility of environmental degradation within the zone but permits the discharge because the socio-economic benefit of the culture operation is deemed to outweigh the ecological cost, i.e. there is "net benefit" from the practice.

In net-cage culture, for example, regulatory authorities may designate a mixing zone for solid wastes as the area directly beneath the farm or as the total area of the lease (typically many times the size of the actual net-cages). A mixing zone for dissolved wastes may be established at some arbitrary distance from the farm as in Washington State (USA) where, for some farms, compliance with water quality standards is determined by sampling 30 m from the net-cages. Mixing zones for dissolved wastes from aquaculture may attain a size of over 1 km^2 (AECOS, 1991). When a mixing zone is established it is, in principle, unnecessary to monitor within it.

It is sometimes argued that if culture practices and environmental conditions allow, use of a mixing zone should be avoided since it implicitly allows localised changes in the quality of water and/or sediments. A mixing zone, however, may be a necessity if coastal aquaculture is to develop in some areas. Management based on a mixing zone approach does provide some controls on the extent of coastal development. Mixing zones for two or more discharges of pollutants should not overlap and, thus, the delineation of a mixing zone precludes additional development nearby. For example, if an upland aquaculture facility along an open coast is given a mixing zone extending 1 km along the shoreline in both directions, a second farm of comparable size would have to be at least 2 km distant in order to avoid overlapping mixing zones. In practice the actual separation would likely be even greater since subdividing a water body into abutting but non-overlapping mixing zones would not be prudent.

4.7 Detecting Ecological Change

When the potential for ecological change is low, minimal monitoring may suffice, such as an annual survey by divers to observe and record qualitative changes. In most instances, however, monitoring is of little value if measurable change can not be unambiguously linked to a particular culture facility and distiguished from natural variability and the effects of other sources of contamination. Such a linkage requires a rigour (i.e. strict adherence to good scientific methods) rarely found in current programmes for monitoring the effects of coastal aquaculture wastes. In designing a monitoring programme, greater consideration should be given to baseline comparisons (Item 4.3 above), impact assessment relative to reference sites (Item 4.4 above) and replication necessary to detect a pre-defined magnitude of change.

The ability to detect an ecological change is inversely proportional to the magnitude of the impact. That is, a monitoring programme required to detect a small change will require more replicates and more precise measurements than would be required to detect a large impact. The cost will be correspondingly high. An essential consideration in the design of a monitoring programme is therefore the power of the sampling regime to provide data which allow detection of change. For example, there may be no statistically significant difference between mean chlorophyll biomass at two separate occasions. This may be true or may be due to the fact that the sampling design is too poor to detect a change which actually occurs (Type II error). Power analysis enables *a priori* assessment of the probability of detecting a statistical change with a given sampling design or alternatively to determine how many samples are needed to detect a change at a certain power (Cohen, 1988; Peterman, 1990; Fairweather, 1991).

In many instances, statistical analysis of the data will be necessary to support the assertion of cause and effect. Commonly used statistics include analysis of variance to compare concentrations of dissolved nutrients or phytoplankton biomass and measures of species richness to assess changes in the population structure of the macrofauna. There are many texts on statistics and their use in experimental design (see Green, 1979; Sokal and Rohlf, 1981); however, detailed discussion of the utility of specific tests is beyond the scope of this report.

4.8 Feedback

It is clear that monitoring achieves little unless the responsible agency is in a position to take action when there is a departure from pre-determined standards. Nevertheless, there are numerous cases in which monitoring reports are filed as records, only to be examined in the event of a major complaint or legal action. Thus, the process by which reports are reviewed and results evaluated should be well-planned and regarded as an integral part of monitoring.

Monitoring is the main regulatory tool for assessing ecological change but the data are also of value in evaluating and modifying the monitoring programme itself. If this feedback is neglected, an inappropriate monitoring programme may continue for long periods without modification. Continuous evaluation shows where the monitoring programme should be intensified or scaled back as the results dictate. Further aquaculture development or expansion may be allowed if measurements show that observed ecological change is well within pre-determined limits.

Where applicable, data from monitoring programmes should be used to validate models which have been used in the EIA to predict the scale of ecological change and for refining environmental quality standards. Furthermore, monitoring data may be of value for building a database of benefit to the local aquaculture community.

4.9 Flexibility in Monitoring Intensity

The intensity of monitoring (e.g., number of stations and variables) should depend upon the size of the culture facility (commonly measured as annual production of biomass) and the sensitivity of the receiving waters. The most intensive monitoring requirements should be imposed on larger farms releasing wastes into waters of particular ecological sensitivity or significance. Intensive monitoring of small farms is unwarranted unless, by virtue of their density, combined or cumulative effects are likely to threaten the ecological health of the water body. Flexibility in design of monitoring programmes and scaling of monitoring requirements to farm size and site sensitivity can be achieved when programmes are developed on a case-by-case basis between the culturist and regulatory body. In some monitoring programmes this flexibility is built into standardised monitoring protocols (e.g., monitoring requirements imposed on salmon net-cages in Canada; Black, 1991, Black and Truscott, 1994). However, some regulatory authorities have imposed uniform monitoring protocols upon all producers of a particular culture type. This approach is inappropriate in that it requires excessive monitoring of small producers without adequate ecological justification.

4.10 Monitoring Effluents Versus Receiving Waters

Until recently, most monitoring of point sources has concentrated on effluents. This approach fits naturally into monitoring the effects of wastes from land-based culture systems which usually have a readily-defined output source. In this case, the primary purpose of monitoring is to determine compliance with standards or conditions laid down under discharge licences. For cage or pen culture, on- or off-bottom bivalve and seaweed culture, there is no well-defined effluent and it is therefore necessary to make direct observations on the effects of the operations at the site and within the farm permit area.

There is growing agreement among regulatory agencies that, in all cases, whether or not there is a definable effluent, it is necessary to monitor the receiving waters. First, this ensures that effluent standards, if any, adequately protect the receiving waters. Second, it may avoid the possibility that combined discharges from nearby farms or other coastal activities will cause receiving water standards to be exceeded. This changing perspective among agencies also supports the development of procedures designed to estimate the environmental capacity of defined areas and encourages a more predictive approach to monitoring, for example by employing hypothesis testing procedures (Morrisey, 1993).

Effluent monitoring may remain the only viable approach in areas heavily affected by a wide variety of contaminant sources. In such cases, relating impacts on the receiving water to any particular source is difficult and effluent monitoring is the best approach to establishing relative responsibilities. Care must be taken however, to ensure that sources with defined effluents are treated equitably in relation to non-point sources of the same materials. While point sources are more easily monitored and controlled, their contribution to contaminant loading may in some cases be inconsequential in comparison to non-point sources, with the result that no amount of effluent control will have a measurable effect on the quality of the receiving water.

4.11 Developing Aquaculture-specific Guidelines for Ecological Monitoring

Monitoring is not being undertaken in the majority of current coastal aquaculture practices. Consequently, efforts to promote aquaculture-specific monitoring may need to be initiated for many locations. Whenever such monitoring activities are to be started, it is recommended that farm managers, regulators, administrators, and scientists work together to develop their own specific guidelines for ecological monitoring of aquaculture waste. The following basic considerations are proposed as reference for the development of such aquaculture-specific monitoring guidelines:

- when formulating programmes or requirements for environmental assessments and monitoring, due consideration should be given to the diversity of aquaculture practices (including, in particular, the species used and the culture methods applied) and their environmental settings;

- any environmental assessment and monitoring effort should be related to the scale of perceived impact of a given aquaculture operation;

- in many cases, particular emphasis will need to be given to simplicity, flexibility and affordability of environmental assessments and monitoring, in order to facilitate the acceptance and enforcement of such measures. Consultation and participation of interested and affected parties in the formulation of requirements for environmental assessment and monitoring should be encouraged. A detailed evaluation of financial, manpower and time requirements for any such effort should precede their implementation to demonstrate their cost-effectiveness and feasibility;

- the ecological component of an environmental impact assessment should be designed such that all significant impacts of wastes are identified and an appropriate monitoring programme constructed;

- monitoring should preferably be undertaken within a framework of established Environmental Quality Objectives and Standards;

- monitoring for ecological protection should be regarded as an integral part of managing aquaculture operations. In particular, the results derived from monitoring should be used to evaluate the ecological effects of the operation, the suitability of relevant EQSs and the utility of the monitoring programme itself.

5. MONITORING PRACTICES

Despite the diversity of coastal aquaculture practices around the world, a relatively small range of variables has been routinely used to monitor the ecological effects of wastes. The most commonly used variables are listed in Table III together with qualitative estimates of how frequently they have been incorporated into monitoring programmes, the relative cost typically charged for analyses, and the relative interpretative value of the variable as an indicator of ecological change attributable to aquaculture. In many cases, biological variables are given the

Table III

Variables often used to monitor the ecological effects of coastal aquaculture wastes for the purposes of environmental protection.

Variable	Usage[1]	Cost[2]	Value[3]	Comments
Sediment chemistry				
Redox potential (Eh)	M	L	H	Measurement is rapid and at low cost beyond initial equipment purchase, thus permitting extensive spatial surveys.
Depth of visual Eh discontinuity	L	L	L	Only semi-quantitative and sometimes difficult to determine, but can be measured at no cost beyond sample collection.
Total Organic Carbon (TOC)	H	M	M	Most easily measured by elemental analyzer, if available, but can be also done by wet chemistry.
Total Kjeldahl Nitrogen (TKN) Total Nitrogen (TN)	L	M	M	Provides similar information to TOC. TN data are provided simultaneously with TOC if elemental analyser is used.
Benthic biota				
Visual presence of *Beggiatoa*	H	L	M	*Beggiatoa* is a filamentous bacteria that forms a white mat on the sediment surface in areas of intense organic enrichment. A highly reliable and easily observed indicator. Its presence suggests substantial disruption of natural benthic processes.
Macrofaunal community structure	H	H	H	Expensive and requires taxonomic expertise that may not be widely available, but provides evidence of benthic impact.
Visual survey of large invertebrates and demersal fishes	H	L	L	Documentation by still photos or videotape. Observations typically only qualitative.

Variable	Usage[1]	Cost[2]	Value[3]	Comments
Water chemistry				
Dissolved Oxygen	H	L	H	Measurement by wet chemistry or probe.
Biochemical Oxygen Demand (BOD)	H	L	H	Widely used for measurement of effluent quality for land-based systems.
Suspended solids (SS)	H	L	M	Widely used for measurement of effluent quality for land-based systems.
pH	L	L	L	Useful as a measure of impact only in some forms of pond culture, but often measured as an ancillary variable.
Transparency (Secchi disk, NTU)	L	L	L	Non-specific indictor of questionable value in many locations.
Dissolved inorganic nutrients (NH_4^+, NO_2^-, NO_3^-)	H	M	M	Appropriate in particular when the ecosystem is N-limited.
(PO_4^{3-})	L	M	M	Appropriate in particular when the ecosystem is P-limited.
Phytoplankton Biomass				
Chlorophyll	L	M	H	Useful as an index of eutrophication, but unlikely to be of value in monitoring a single facility.

[1]Usage, this is the frequency of use of a particular variable
[2]Cost is cost per sample and does not include capital costs for equipment
[3]Value, this is an estimate of the interpretative value of the measurement

L= Low; M = Medium; H = High.

highest ranking in the ecological value category. This is because biological change is the primary concern and chemical measures are only more expedient surrogates and may at times be misleading. The table lists variables commonly employed to determine compliance with regulatory mandates but excludes more specialised variables that might be measured for research purposes (e.g., primary production) or in unusual situations. Furthermore, it does not include those variables that may be required for interpretation of the listed variables which, by themselves, are not particularly useful for monitoring. For example, temperature is necessary to express dissolved oxygen as a percentage of saturation; grain size data are critical to interpretation of macrofaunal data which, when used alone, are typically a weak measure of ecological impact.

It should not be concluded that a properly designed monitoring programme must include all of the listed variables. The table provides a list of common options. The subset selected in any one instance will depend upon site characteristics (e.g., a hard substrate would preclude Eh measurement), farm size and site sensitivity (the number of measured variables should be directly related to both of these criteria). Generally the list of monitored variables should be scaled according to the level of ecological concern. For example, small farms in areas of rapid waste dispersal and dilution may require no or only relatively limited monitoring. At other sites, where the likelihood of potential impact is greater, monitoring may, for example, emphasise the presence of the filamentous sulphide-oxidising bacteria, *Beggiatoa*, or depth of the visual redox potential discontinuity (depth in the sediment below which the interstitial water is anoxic and characterised by a change in sediment colour from brown to black). Moderate levels of concern may require inclusion of chemical measures of enrichment (TOC, TON) or measurement of redox potential. In locations where any impact on benthos is undesirable, analysis of macrobenthic community structure as a supplement to the previous approaches provides the most appropriate basis for protection.

Section 6 provides additional guidance on how to select the relevant variables from this list for designing specific monitoring programmes.

6. HYPOTHETICAL MONITORING PROGRAMMES

In the preceding sections an attempt has been made to describe the basic principles of monitoring programmes and to assess the relative interpretative value of commonly measured variables. It is not possible however, to recommend standard monitoring programmes. The reason for this is that the type and scale of ecological change will depend on the type of aquaculture, the size of the operation and the sensitivity of the recipient water body. Furthermore, the level of change to be detected will govern the intensity and scope of any monitoring programme.

To provide examples of how some of the commonly used variables may be incorporated into monitoring programmes, five hypothetical scenarios are presented. In each example the general objective is to maintain the structure and function of the aquatic ecosystem in the coastal region.

6.1 Scenario 1

A sea bream net-cage farm with an annual production of 100 t y^{-1} is to be placed in estuarine waters at the entrance to a small bay (Figure 5). The land immediately surrounding the bay has a moderate to steep gradient. An access road to the bay area enters from the west along the northern shoreline, and there are ten houses, six of which are occupied only in the summer months. The summer residents come primarily to fish in the bay and enjoy its relative seclusion. There is no commercial activity in the area other than the proposed cage farm.

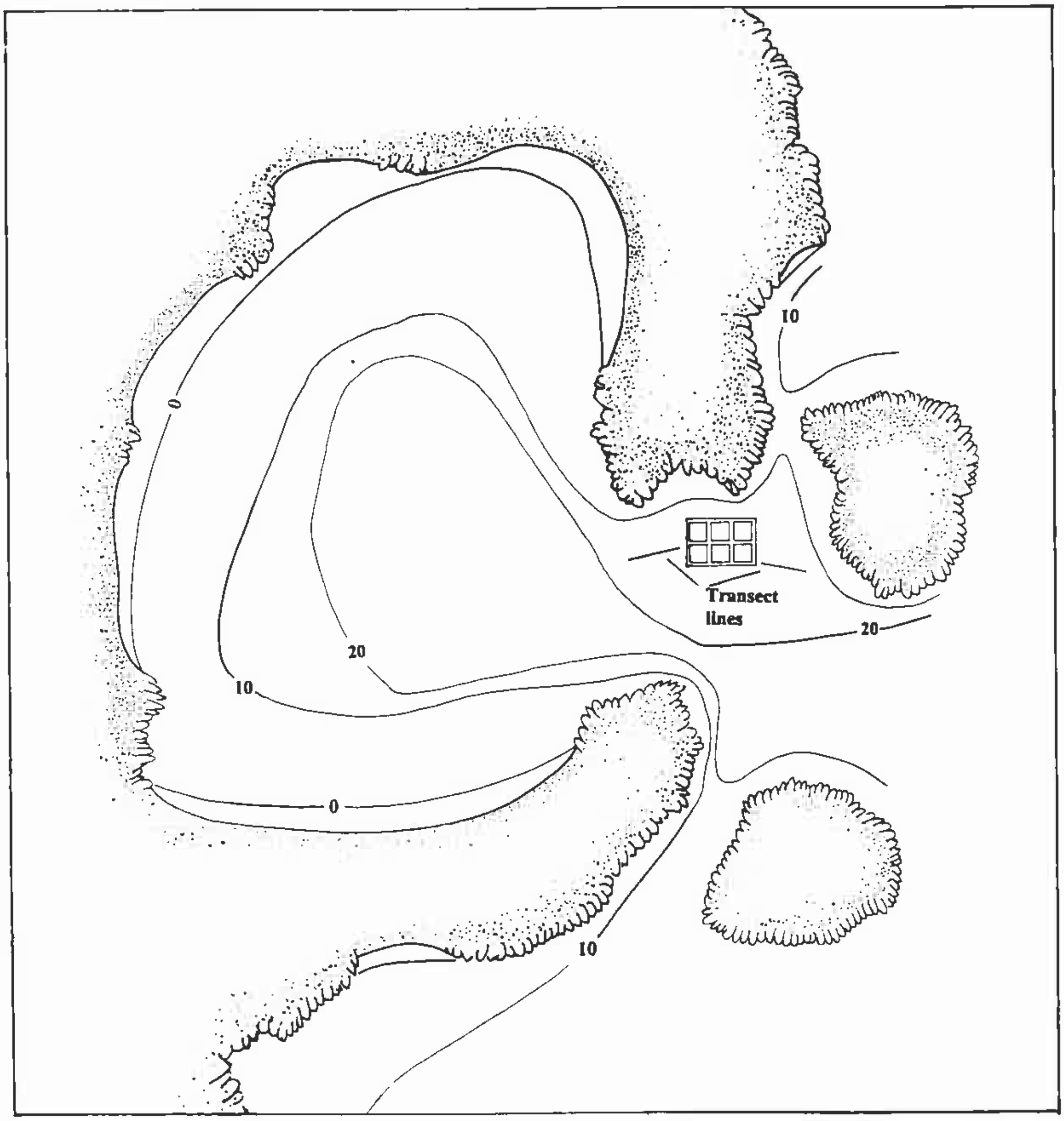


Figure 5. A diagram of the bay described in scenario 1, showing the location of the fish farm and the transect lines used in the monitoring programme (depth contours in metres).

The cages will be located in a constricted channel connecting the bay (high water area of 4 km^2) to an estuary to the east. Results of an environmental assessment indicate that the tidal amplitude is 3 m and that tidal currents commonly exceed 50 cm s^{-1} in the channel. The site was selected because of its high current velocities and hence good supply of oxygen to the farm. In addition, the shape of the bay and its shallow depth ensure there is good exchange of water between the bay and the estuary throughout the year. The surrounding waters are rich in nutrients most of the year although, during the summer, low concentrations are measured in near surface waters. Water depth at the farm site is 10 m and the mean depth of the bay is 12 m. The benthic biota is dominated by attached macro-algae, large herbivores, such as limpets and sea urchins, and attached suspension feeders such as sea anemones.

Recommended monitoring programme

Owing to rapid dilution resulting from the fast currents and turbulent mixing, direct measurements of nutrients coming from the farm would be difficult. An estimate of the flushing time of the bay can be calculated from the high water volume ($4 \cdot 10^6$ m^2 $\cdot$ 12 m, or $4.8 \cdot 10^7$ m^3) and the tidal volume ($4 \cdot 10^6$ m^2 $\cdot$ 3 m). It would take about four tides to replace the water of the bay and since there is a diurnal tide, this means the bay is flushed every 2 days. Given such rapid exchange it is considered unlikely that nutrient enrichment of the bay water would occur, or that phytoplankton would remain in the bay for a sufficient time to utilise the additional nutrient, grow and for biomass to accumulate. The effect of nutrients released from the farm is considered to be negligible.

The benthic fauna are mostly herbivores and suspension feeders. The potentially fast currents are likely to prevent significant accumulation of feed and faeces, but the suspension feeders could be sensitive to sedimentation during periods of slack water. There is little information available on the impacts of cage culture in high currents or over rocky substrates. For this reason a low level of monitoring of the substrate would seem appropriate. This could be most efficiently carried out by divers during an annual survey. Given the narrow channel and strong currents (which may disperse organic waste over considerable distances) a transect should be established along the axis of the channel extending 100 m in each direction from the edge of the farm. Divers should make and record observations (sedimentation type, macroalgal density, common invertebrates, demersal fishes and the presence or absence of feed or faeces) and take photographs every 20 m along the transect. Alternatively, a video recording could be made along the length of the transect. The monitoring programme is run for two years and then evaluated.

6.2 Scenario 2

For this scenario the modelling approach recommended by CSTT (1994) is used as an initial screening approach to assess the potential for eutrophication.

A proposal has been submitted for a large (500 t) land-based fish farm on the shore of the inner basin of a fjordic estuary (Figure 6). Objections have been raised by a local conservation organisation which cites the Government's Environmental Quality Standard of 10 mg chlorophyll m^{-3} and general guideline that only in exceptional circumstances would planning permission be granted if the EQS was likely to be exceeded. To support their case, the conservation organisation employs a consultant to assess the effects of soluble waste from the proposed development. To predict the level of nutrient enrichment and the effects of additional nutrients on phytoplankton biomass, the consultant uses the modelling approach mentioned above together with data on phytoplankton ecology which have been collected by staff of a local marine institute.

The fjordic estuary in question is some 35 km long and on average 1 km wide. A plan and bathymetric profile of the estuary are shown in Figure 6. The basic physical variables of the fjord basin in which the proposed fish farm is to be located are:

High water area (km^2)	1.35
Low water area (km^2)	1.18
Mean tidal amplitude (m)	2.0
Mean high water volume ($\cdot 10^6$ m^3)	13.5
Mean low water volume ($\cdot 10^6$ m^3)	11.8
Mean daily river inflow ($\cdot 10^6$ m^3)	0.284
Mean inner basin salinity (ppt)	31.14
Mean main basin salinity (ppt)	33.87

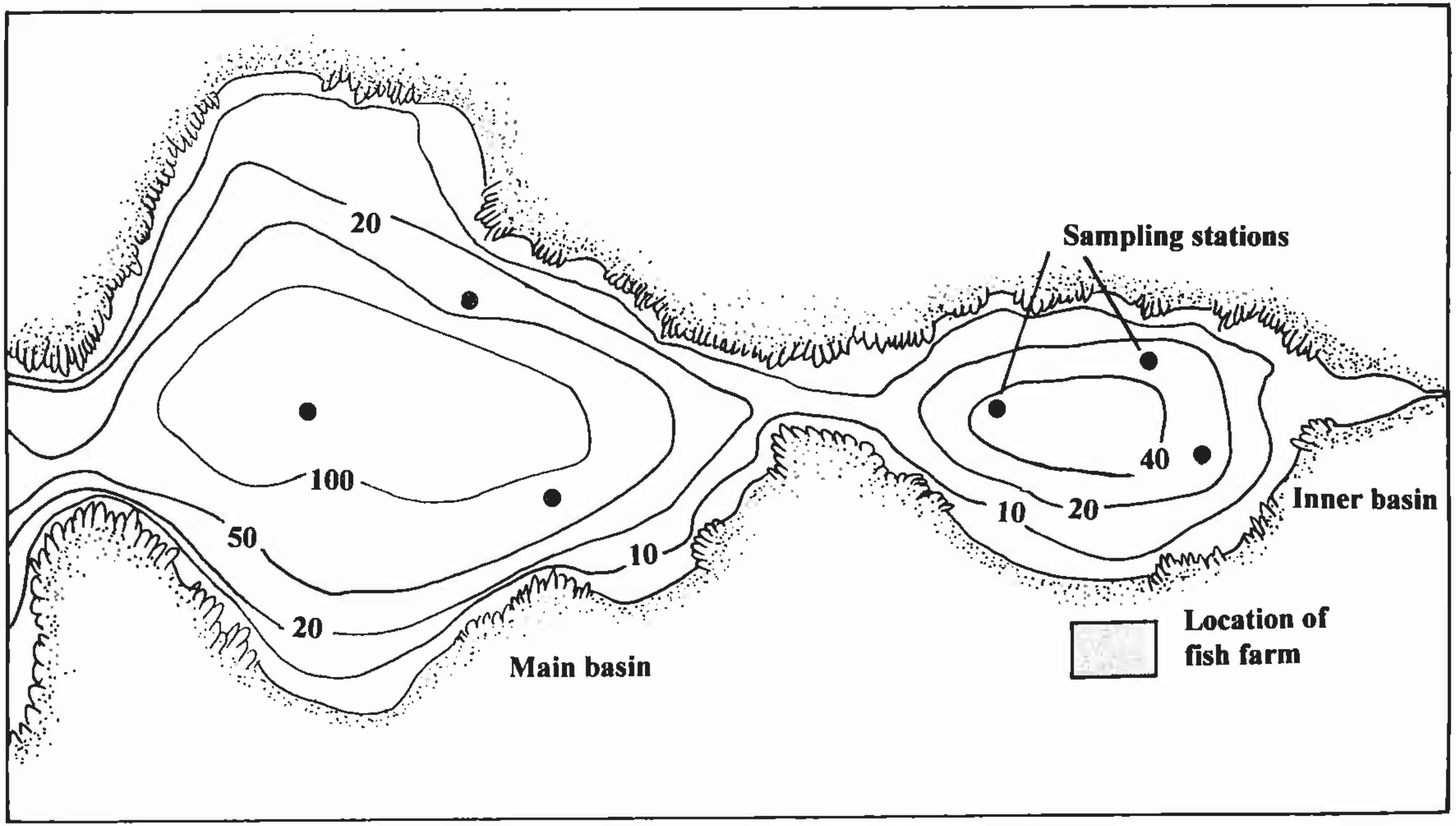


Figure 6. A diagram of the fjord described in scenario 2, showing the location of the fish farm and the sampling stations. (depth contours in metres)

There is limited human activity on the shores of the fjord, notably a few houses which are occupied during the summer and a small hamlet on the shore close to the fjords entrance. The only agriculture practised within the catchment area is rough grazing of sheep and cattle.

The report by the consultant sets out the 6 steps in the modelling procedure as follows:

Step 1: *Defining the volume of the basin into which the waste is dispersed*

The sill to seaward of the inner basin has a mid-tide depth of 11 m, which approximately corresponds with the depth of the pycnocline. As such, soluble nitrogenous waste (ammonia) from the fish farm is assumed to be dispersed over a depth of 10 m and throughout the basin.

Step 2: *Estimating the rate at which the estuarine segment exchanges with the adjacent waters*

In this case the adjacent waters are taken to be the main basin of the fjord. Two methods were used to estimate dilution of the inner basin. The first is based on river inflow and salinity (Bowden, 1967; Saelen, 1967) where dilution rate D (per day) is given by:

$$D = (R \cdot S_o) / (V \cdot (S_o - S))$$

R is river inflow ($m^3\ d^{-1}$), S_o and S are the mean salinities (ppt) of sea-water flowing into the basin and of the basin respectively. V is the volume (m^3) of the estuarine segment under consideration.

The second method is based on the volume exchanged each tide (Bowden, 1967; Edwards and Edelsten, 1976) where dilution D (per tide) is given by:

$$D = (V_h - V_l) / V_h$$

in which V_h and V_l are the high and low water volumes of the basin segment. Using the data given above, the estimate of dilution based on river inflow and salinity is 0.28 d^{-1}. For the tidal volume method the estimate is 0.13 $tide^{-1}$ and given that there is a diurnal tide in the area this gives a dilution rate of 0.26 d^{-1}. Thus, the mean dilution rate is taken to be 0.27 d^{-1}, that is the upper 10 m segment of the inner basin is flushed approximately every four days.

Step 3: *Calculating the maximum steady state nutrient concentration*

$$N_m = N_a + (A_d + N_i)/(d \cdot v)$$

N_a the nutrient concentration in the adjacent waters is taken to be 0.4 mmol m^{-3} (nitrate + nitrite + ammonia). A_d the daily input of nutrient (ammonia) from the fish farm is taken to be $19.6 \cdot 10^6$ mmol d^{-1} derived as follows:

1) 500 t fish fed 1.2% body weight per day : 6.0 t
2) Food consumed by fish as a proportion of food fed is 86% : 5.16 t
3) nitrogen content of food is 7.6% : 0.39 t
4) nitrogen excreted as proportion of nitrogen consumed is 70% : 0.275 t

The above does not include organic nitrogen in uneaten food and faecal material which may be remineralised from the sea-bed as soluble nitrogen.

N_i the daily input of nutrient from rivers is taken to be $0.1 \cdot 10^6$ mmol. This is based on the assumption that the concentration of dissolved nitrogen in the river in question is similar to that of other rivers in regions of comparable geology and agricultural practice and for which there are published data. The volume exchanged daily with the main basin, $d \cdot v$ is estimated as $3.4 \cdot 10^6\ m^{-3}\ d^{-1}$.

Thus, the steady state concentration of nitrogen (N_m) is

$$0.4 + (19.6 \cdot 10^6 + 0.1 \cdot 10^6) / (3.4 \cdot 10^6) \text{ or } 5.9 \text{ mmol m}^{-3}.$$

Step 4: *Estimation of the potential maximum chlorophyll (X_m)*

$$X_m = X_a + q \cdot N_m$$

where X_a the chlorophyll concentration in adjacent waters (the main basin) is taken to be 3.4 mg m^{-3}, and q the yield of chlorophyll from nutrient is taken to be 1.1 mg chlorophyll (mmol N)$^{-1}$ (See section 3.1 for discussion of q).

The estimated increase in chlorophyll is therefore 6.5 mg chlorophyll m^{-3}.

Step 5: *Predicting the worst-case effects of nutrient enrichment compute the light-controlled growth rate ($\mu(I)$)*

$$\mu(I) = \alpha(\langle I \rangle - I_c)$$

where, α is photosynthetic 'efficiency' taken as 0.012 (μE m^{-2} s^{-1})$^{-1}$, I_c is compensation irradiance taken as 12 μE m^{-2} s^{-1}, $\langle I \rangle$ is mean layer illumination given by:

$$m \cdot I_0 / k \cdot h$$

where, m is a correction factor (0.82) for near surface losses of solar radiation; I_0 is the 24-hour mean sea-surface photosynthetically active radiation (PAR); h is the thickness of the surface layer in which phytoplankton grow, which for the inner basin is 16 m; k is the diffuse attenuation coefficient for submarine PAR, taken to be 0.197 for the inner basin during summer.

Using a value of I_0 of 357 mE m^{-2} s^{-1} gives a value for $\mu(I)$ of 0.97.

Step 6: *Compare light controlled growth rate with loss rates*

$$m(I) >> D + l$$

where l is local loss - zooplankton grazing : Since there are no data on typical biomass of zooplankton in the fjord, grazing loss is assumed to be zero. The main loss is dilution given the dilution rate of 0.27 d^{-1}.

The estimated growth rate is higher than the dilution rate, thus even allowing for some grazing it is likely that phytoplankton would stay in the basin long enough for growth to result in an accumulation of biomass.

Conclusion

The assessment undertaken by the consultant indicates that the fish farm would cause a significant increase in summer phytoplankton standing crop bringing the level of chlorophyll close to the Government EQS. The report is accepted by the local regulatory authority who refuse planning permission for a 500 t farm. However, the regulatory authority grants an application for a smaller farm (250 t annual production) on the condition that a monitoring programme be established. The licence stipulates that if the results of the monitoring programme indicate that the EQS is being exceeded the farmer will be required to reduce the size of the farm. Since the regulatory authority does not want to unnecessarily restrict development, the authority intimates that an increase in annual production may be allowed if the predicted increase in phytoplankton standing crop is not realised.

Recommended monitoring programme

The monitoring programme is designed to fulfil three objectives: ensure compliance with the EQS; identify any changes in bottom water oxygen concentration; validate the model.

(i) Compliance with the EQS

This is the main objective of the programme. Since the EQS is based on phytoplankton chlorophyll this is the obvious variable to monitor and there are no substitutes. The regulatory authority could opt to simply monitor levels of chlorophyll in the inner basin to make sure that they do not exceed the EQS of 10 mg m^{-3}. However, a comparative approach is considered better because it will provide more insight into how the phytoplankton respond to the additional nutrients and allow a more informed assessment of the effects of the farm and any possible expansion. The comparative approach will also address the question of whether any measured increase in chlorophyll in the inner basin is natural or due to the farm. The basis of the monitoring programme is therefore to compare historical data with data collected during monitoring and to compare chlorophyll concentrations in the inner and main basins of the fjord. The outer basin represents the control, but may be compromised because of exchange with relatively nutrient rich water from the inner basin.

An indication of the potential for this can be determined by repeating step three, but for the fjord as a whole. If the model predicts nutrient enrichment in the main basin, there is little point in using the basin as a control. It will be necessary to locate reference stations in adjacent coastal waters or a nearby fjord, or to seriously reconsider whether the location of the farm is suitable and the modelling appropriate. With respect to using adjacent coastal waters as a control, the ecology of phytoplankton in coastal waters is likely to be markedly different from that of the fjord, making comparisons difficult. The same argument may apply to nearby fjords, although it is more likely that fjords with similar physical characteristics (such as dilution rate and water column structure) will support similar populations of phytoplankton.

For the purposes of this scenario, it is assumed that the main basin of the fjord will not be enriched. To monitor changes in phytoplankton chlorophyll therefore, three surveys of the fjord are to be undertaken at approximately monthly intervals during the summer each year. During each survey a total of six stations will be worked, three in each basin. At each station water samples will be collected from depths of 2, 5 and 10 m for estimation of chlorophyll.

It is anticipated that natural variation in chlorophyll concentration will occur as a result of small scale patchiness within each basin and temporal variation over the three months that monitoring is conducted. However, as the aim of the monitoring programme is to compare summer chlorophyll levels in the two basins, the data from the three surveys of each basin are combined. The statistic, analysis of variance can be used with $\log_{10}$ transformed chlorophyll data to determine if summer chlorophyll concentrations in the two basins are significantly different. Combining the data in this way allows variation (due to natural factors) within each basin to be distinguished from differences between the two basins. The null hypothesis is therefore that the biomass of phytoplankton in the two basins are similar. This hypothesis is rejected if the results of the analysis shows that the variation between the two basins is greater than the variation within each of the basins. Similarly, the data collected during monitoring can be compared with historical data from the two basins. Once the monitoring programme has been in operation for a number of years, annual variability can be assessed and an appropriate statistical test (such as analysis of variance) used to detect any temporal increase in summer chlorophyll in the main basin which would indicate that nutrient enrichment of the basin was occurring.

Monitoring nutrient concentrations during the summer would also provide information of whether phytoplankton in the inner basin are utilising the additional nutrient. For example, elevated summer ammonia levels in near surface waters of the inner basin (but not in the main basin) could be interpreted as indicating that phytoplankton in the inner basin were not utilising the additional nutrient. While this would increase the scientific basis upon which the regulatory authority would make decisions regarding future levels of farm production, summer nutrient data are not essential for interpreting the chlorophyll data. In addition, monitoring summer nutrient concentrations would substantially increase the cost of the monitoring programme. For these reasons the regulatory authority does not stipulate a requirement for monitoring summer nutrients.

(ii) Dissolved oxygen in bottom waters of the inner basin

The settlement of a greater phytoplankton biomass together with the possibility of limited flushing could result in depletion of oxygen in the bottom water of the inner basin. For this reason, monitoring concentrations of dissolved oxygen in the bottom water of both basins is to be undertaken during the summer and winter. To avoid unnecessary visits to the fjord, sampling will be carried out at the same stations and with the same frequency as sampling for summer phytoplankton biomass and winter nutrients. Four water samples are to be collected, from each station from depths of 8, 20 and 30 m and from 5 m above the sea-bed.

(iii) Verification of the model

It is not strictly necessary to monitor compliance with the chlorophyll EQS, but the regulatory authority argues that validation of the modelling approach is important. In addition, field surveys would provide an additional indication of whether or not enrichment of the main basin was occurring. It is also in the interest of the farmer to verify the model since field data may show that the model has over-estimated nutrient enrichment and therefore increased production might be permitted. This part of the monitoring programme is run for two years.

Monitoring should be undertaken on three occasions, at approximately monthly intervals during the winter and using the same sampling stations as those for summer chlorophyll. At each station a vertical profile of salinity is to be recorded and water samples, for estimation of ammonia, collected from depths of 2, 5 and 10 m. Verification of the model is as follows: the salinity data are used together with estimates of river inflow during the time of the survey to estimate dilution of the upper 10 m segment of the inner basin; using tidal amplitude data (from local tide tables) dilution based on the tidal exchange method is estimated; using the estimates of dilution together with details of farm husbandry (stock and feed usage) the predicted increase in nutrient is determined as outlined in steps 2 and 3 of the modelling procedure. The model predictions can then be compared to the field data and an assessment of the model made.

In addition to the above monitoring programme imposed by the regulatory authority the farmer is advised that there is a risk of additional nutrients stimulating the growth of phytoplankton species which are potentially harmful (toxic or likely to cause physical damage to gills) to the fish. The farmer therefore undertakes his own monitoring of phytoplankton species composition during the late spring, summer and autumn. Since it is possible that a bloom of a harmful species may enter the fjord from open coastal waters or develop within the fjord, monitoring is undertaken at one station in each basin. An integrated, 10 m water sample is collected from each station using a hose sampler. Monitoring is undertaken on a weekly basis.

6.3 Scenario 3

An intensive shrimp (*Penaeus monodon*) farm using 16 diked ponds (2500 m^2 each) is proposed for construction in the supratidal area, of a coastal region, about 150 m behind a beach (Figure 7). The total pond area will occupy about 4.5 ha with additional land use for infrastructure (approx. 15% of the pond area). The units are to be constructed above sea level and water supplied by pumping from a sea-water intake located 60 m offshore. Effluents are returned to the sea via run-off canals where they disperse along the shore, occasionally passing the intake. Daily water exchange is by partial draining and refilling with pumped sea-water at 5-10% of total volume early in the production cycle, increasing to between 25 and 30% by the end of the four month production period. Total planned production is approximately 40 t y^{-1}

derived from 2 production cycles per year with an average output of about 5 t ha^{-1} $cycle^{-1}$. Feeds used amount to 10 t ha^{-1} $cycle^{-1}$. The surplus organic sediment produced per cycle amounts to 150 t ha^{-1} $cycle^{-1}$ (dry weight). Ponds are totally drained during harvesting, resulting in 2 full water exchanges annually. Initially, the release of suspended solids to the environment during flushing at harvest will be permitted, although authorities reserve the right to require the removal of bottom sludge and appropriate disposal at a later date. The water released during daily water change may contain on average 13.5 mmol m^{-3} total P and 242.9 mmol m^{-3} total N. The concentration of suspended solids may be between $3.0 \cdot 10^4$ and $1.9 \cdot 10^6$ mg m^{-3}. Chlorophyll concentration in the effluent ranges between 20 and 275 mg m^{-3}.

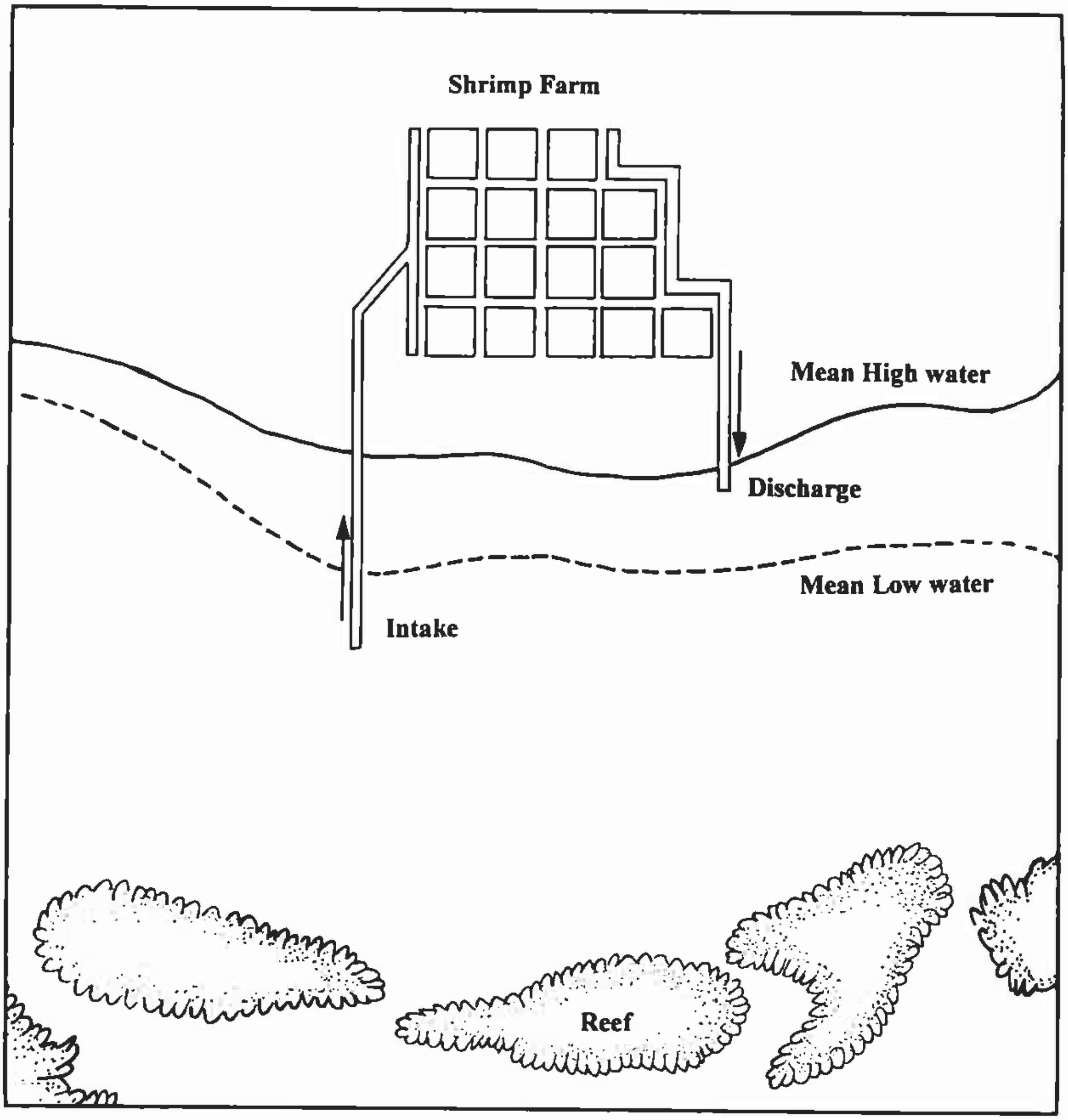


Figure 7. A diagram showing the location of the shrimp farm in relation to the reef, described in scenario 3.

Tidal amplitude of the receiving waters is small and coastal waters are low in nutrients and suspended solids. A fringing reef is located 0.5 km offshore. No prior shrimp farms have been located in the area, and aquaculture development has been limited, so authorities are dependent upon data from elsewhere to predict the environmental effect. In this respect, the results of a study have shown that the shallow sub-tidal discharge of municipal waste waters from nearby urban areas has not had any obvious biological impact.

The farm is a new development in the area and it is anticipated that additional licences in nearby areas will be applied for in the future when full-scale production has been reached.

Recommended monitoring programme

The presence of oligotrophic waters in the region indicates that the potential for eutrophication should be considered, yet nearby municipal waste waters have had minimal effect. Adequate dispersal of the municipal waste waters has apparently been achieved and the farm wastes would probably be equally well-dispersed given the open location.

If an Environmental Impact Assessment has not been done or did not compare the projected farm nutrient loads to that of the municipal wastes, this comparison should be done. Some waste output data have been provided by the applicant and additional data can be found in the literature. Nutrient loading data for the municipal waste waters may be available from local regulatory authorities, but if not, loading can be estimated from the size of the population served. The potential likelihood of nutrient impact should be re-evaluated after this comparison is complete. The effluent is likely to contain high concentrations of suspended solids particularly during harvest when ponds are completely flushed. It is not known if particulate wastes will reach the reef in biologically significant concentrations but the offshore reef could potentially be affected by turbidity or sedimentation. As a precaution, monitoring should be performed annually.

The offshore reef should be stratified into areas of high, medium and low probability of impact based on proximity to the discharge point and current data, if available. In annual sampling surveys, random sites should be selected within each stratum. Photographs of reef quadrats should be taken at the time of monitoring, and qualitative diver observations of site conditions and faunal abundance should be recorded. Two years after the farm has reached maximum production, the scale of the monitoring programme should be assessed.

6.4 Scenario 4

A salmon net-cage farm (200 t y^{-1}) is located in a large riverine estuary (Figure 8). The farm was sited in its current location seven years ago before local regulatory authorities became concerned about potential ecological effects of fish culture. Consequently, the farm has done no ecological monitoring to date. It has had consistently good harvests and the operator wishes to expand production and has applied for a permit from the regulatory authority to double production to 400t y^{-1} by adding additional net-cages on the seaward side of the existing farm. The permit review was particularly controversial because of the presence of a rocky subtidal area and associated macroalgae, epifauna and herbivorous fishes along the shoreline 500 m south and west of the farm. Such a community is unusual for the region and staff at a local university argued against expansion on the grounds that the rocky area is of special scientific significance and could be threatened by farm expansion.

Local regulators were sympathetic but ultimately approved the permit when the farmer provided predictions based on the modelled deposition of feed and faecal matter. The simulated dispersion and loading of organic waste to the sea-bed showed that the area of highest loading (and hence region of greatest ecological change) would be directly beneath the net cages. Less waste would be deposited to the northwest of the farm although waste could be dispersed up to 150 m. The model simulation indicated that there was little likelihood that solid waste would reach the rocky area to the south and west.

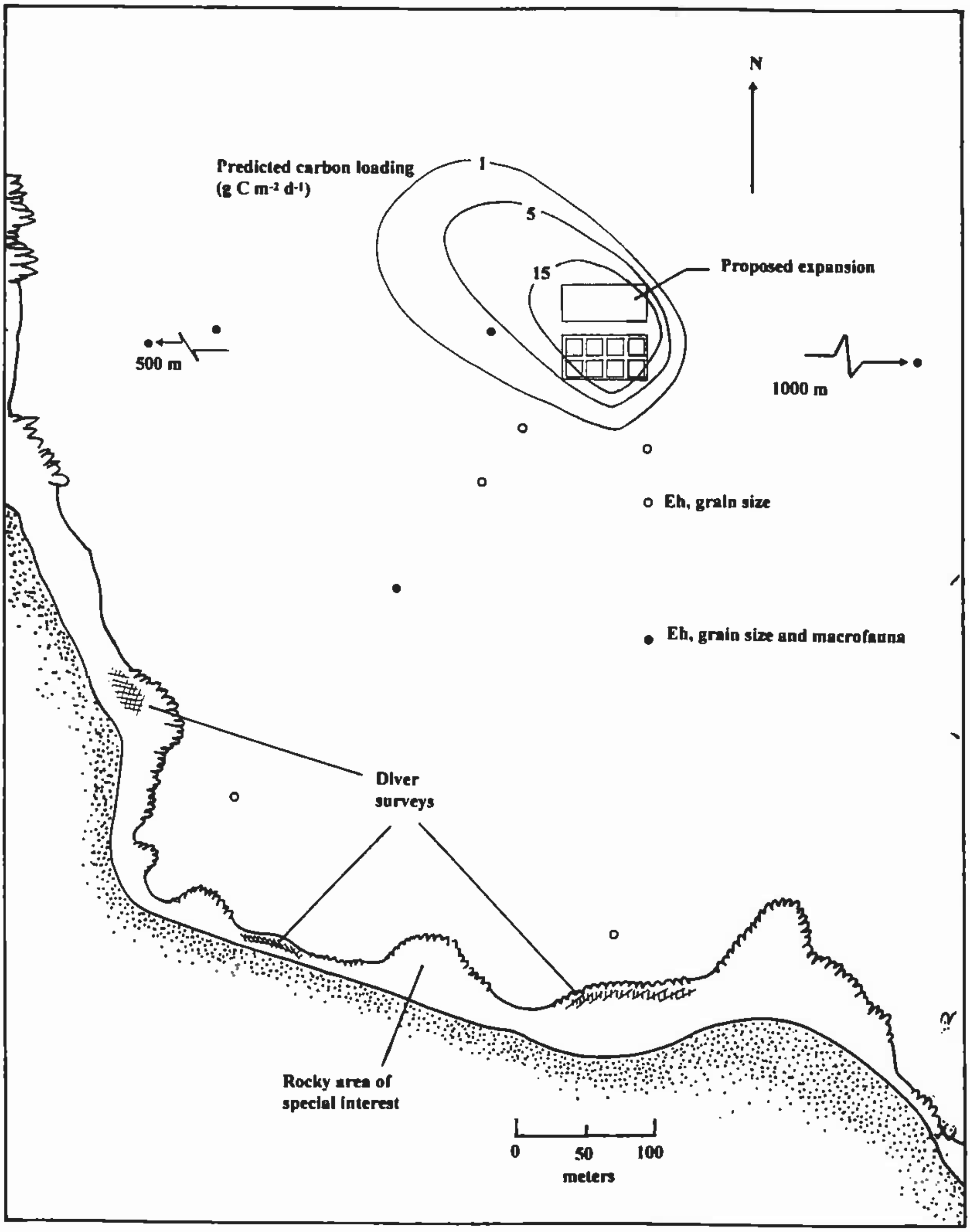


Figure 8. A diagram showing the existing farm and the proposed expansion described in scenario 4 and showing the output from the sedimentation model. The location of the benthic sampling stations (hollow and filled circles) and the diver survey areas are also shown.

Water depths in the area of the net-cages range from 15 to 30 m. Sediments are predominantly muddy sand and support a diverse assemblage of deposit and suspension-feeding infauna. Currents are predominantly tidally-driven to the west. The average velocity of 15 cm s^{-1} and a peak velocity of 40 cm s^{-1} are frequently attained with each tidal cycle. Nutrients and dissolved oxygen in the surrounding waters rarely reach limiting levels.

Recommended monitoring programme

Nutrient enrichment and consequent effect on phytoplankton growth are not likely to be a major concern given the moderate current velocities which should rapidly disperse the plume of enriched water leaving the farm, the large body of water in which dissolved wastes will be diluted and the fact that phytoplankton growth is not nutrient-limited in the estuary.

The primary concern is for the effect on the rocky subtidal area 500 m from the farm. While no obvious effects have been observed with 200 t of fish production nearby, the doubling in waste loading that would accompany farm expansion could potentially: increase nutrients in the local area enhancing the growth of the macroalgae or altering algal species composition; reduce macroalgal growth due to increased turbidity; or cause sedimentation of feed and faecal matter in the rocky area with resultant changes in floral and faunal composition.

Many past investigations of farms of this size and in comparable sites have found that the effects on the macrofauna are limited to < 50 m from the farm and the model predictions are consistent with this observation. In the absence of the rocky area of special significance, it would be justifiable to limit monitoring to an annual diver survey or forgo monitoring completely given the small area of habitat that is likely to be affected. Many regulatory authorities are moving towards allowing degradation under and in close proximity to the farm, simply because there is little alternative if net-cage culture is to be undertaken. Nevertheless, the area of special significance merits protection and a greater degree of caution in proceeding with aquaculture development. Existing sedimentation models, including the one provided by the farmer, have not been tested sufficiently to ensure complete reliability, but more importantly they do not predict the biological effects of a predicted carbon loading rate. Therefore, a benthic sampling programme is necessary to prevent an unacceptable impact.

Monitoring in the sandy zone between the farm and the rocky area would provide an early warning before particulate wastes reached the rocky habitat, allowing time for mitigation. However, the rocky area itself should also be monitored because the biota may be more sensitive to sedimentation than the soft-bottom infauna and to provide some measure of protection against the nutrient and turbidity concerns identified above. The proposed monitoring plan involves measurement of sediment redox potential (Eh) and macrofaunal community structure at some sites and then Eh only as a surrogate for biological measurements at additional stations. The principal advantage of such an approach is that it employs Eh, a rapid and inexpensive measure, to reduce costs yet achieve broad spatial coverage. Since Eh and biota are to be sampled concurrently at some stations, the approach permits inference of macrobiotic composition where only Eh data are available. If equipment for measuring Eh were unavailable, total organic carbon and/or total nitrogen could be substituted for a small additional cost.

The area of special concern is at its closest point south of the farm but, because of prevailing currents, the area to the west of the farm is probably most at risk. Three stations should be established west of the farm at distances of approximately 50, 250 and 500 m. These sample locations provide several opportunities to detect sediment enrichment well before it reaches the area of greatest concern. At each of these sites, samples should be taken by grab, or preferably, by diver for macrofaunal species composition, Eh, and sediment grain size distribution. Grain size data are not by themselves a useful measure of farm impacts but are necessary to interpret the other variables. Lacking site-specific information on spatial heterogeneity for these variables, but assuming that cost containment of the monitoring programme is important, a minimum of three, rather than five, replicate samples at each station and for each variable is recommended. Two additional transects should be established to the south (with stations at 50, 100, 200 and 400 m) to protect that region of the rocky habitat. Eh and grain size data at most sites should be adequate, with one site on each of the transects sampled for macrofauna to supplement the Eh data and avoid misinterpretation.

Reference stations representing unenriched conditions are needed against which conditions surrounding the farm may be compared. Greater confidence in interpretation is achieved with two reference stations. Due to the long history of culture at the site and likely enrichment of the area around the farms, and the fact that sampling anticipates the potential for impact 500 m away, reference stations should be at least 1000 m distant. The first is located 1300 m to the west (not shown on the figure) and the second 1000 m to the east, in areas of comparable substrate type and water depth. Monitoring of the rocky area itself can best be accomplished by divers. Short transects parallel to the shoreline near the end of the three Eh/macrofauna transects should be surveyed as well as a reference area to the west. Observations should be made of floral and faunal composition and abundance as well as any visible evidence of sediment accumulation. Photo-documentation would be desirable. Monitoring of the rocky area and adjacent soft sediment should be conducted annually for at least three years after maximum stocking density is obtained. Thereafter, the data should be reviewed to determine if a reduction in sampling frequency would be warranted.

6.5 Scenario 5

A large corporation has been operating an intensive 6 ha shrimp (*Penaeus monodon*) farm as a pilot scheme for the past three years. The farm is located 10 m from the shore of a medium sized bay (Figure 9). Water is pumped (through a 15 m intake pipe) directly into the ponds via a central supply canal and drains back into a bay via drain canals on both sides of the farm. The farm consists of 24 ponds each covering 2500 m^2, by 1.5 m deep. With a stocking density of 25 m^{-2}, the farm produces 4.5 t ha^{-1} $cycle^{-1}$ $pond^{-1}$ or a total of 54 t y^{-1}. Total feed given amounts to 108 t y^{-1} at a feed conversion ratio of 2. Water exchange is 5% at the start of the cycle, gradually increasing to 30% in the last (4th) month of production. In addition to the pilot farm, there are four small scale (with a combined annual production of 240 t) semi-intensive shrimp farms covering a total area of about 60 ha and an oyster farm producing 70 t $year^{-1}$. Human habitation is concentrated in a small village although there are two other smaller communities along the shore of the bay. There is a small, natural fishery in the bay and nearby open coastal waters which supplies fish locally and to a town some 30 km distant from the bay.

The corporation submits a proposal to increase its shrimp production by expanding its pond area by a further 50 ha in an adjacent mangrove area. The expectation is that yields similar to those produced in the existing farm will be achieved. The regulatory authority receives objections to the proposed development from conservationists who argue that there should be no further reduction in the mangrove forest. The local fishermen, semi-intensive prawn farmers and oyster grower also express concern, arguing that an increase in the discharge of dissolved nutrients and particulate-rich water into the bay will degrade water quality and have a deleterious effect on their livelihood. Any decision that the regulatory authority makes will be determined in part by a recent Government law that requires the retention of 80% of mangrove area.

The regulatory authority asks staff at the state university to assist in assessing the ecological consequences of the proposed expansion, with particular emphasis on any adverse effects on existing activities in the bay. To aid their assessment, the regulatory authority/university team use local information and the results of a five year old study which was undertaken as part of a regional classification of coastal areas together with local information. In addition, the team undertake a limited study of the bay during the course of one dry season.

The bay is shallow with a maximum depth of 17 m and a mean depth of 6 m. The entrance is shallow and constricted which together with the small mean tidal amplitude (1.5 m) means that the bay is only slowly flushed (in about 25 days). It is recognised however, that the

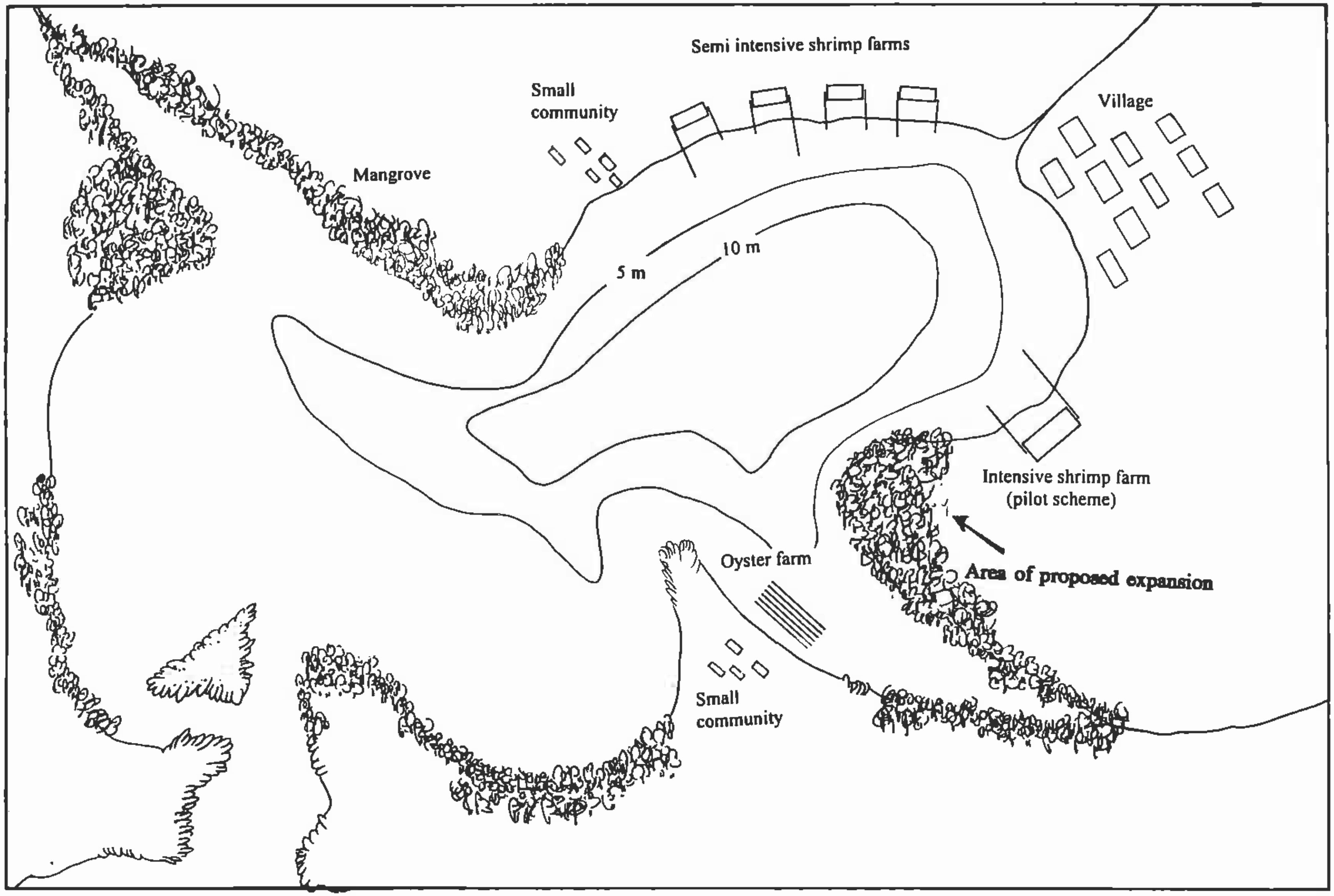


Figure 9. A diagram of the bay described in scenario 5, showing the location of the shrimp farms, the oyster farm and the village.

flushing time of bottom water in the main basin and of water in the coves isolated from the entrance is likely to exceed 25 days.

At the time of the original study the four semi-intensive shrimp farms and the shellfish farm were in full production. Results from the survey show that, during the dry season, water within the bay becomes thermally stratified, resulting in isolation of the bottom water. This is evident by the fact that an oxygen deficit builds in the deep water, with the concentration falling to 5.0 mg l^{-1} compared to 6.5 - 7.0 mg l^{-1} in near surface waters of the bay. Annual production by phytoplankton is high (800 - 1000 g C m^{-2} y^{-1}) and the surrounding mangrove forest further adds to the productivity of the bay. During the dry season, Secchi depth was only 1 m, suggesting that phytoplankton growth becomes light limited. This is supported by the fact that concentrations of dissolved inorganic nutrients were not completely depleted (nitrate concentration was measured at 1.0 mmol m^{-3}). The low Secchi depth is therefore the result of both the high phytoplankton biomass (typically 15 mg chlorophyll m^{-3} during the dry season) and high suspended solid loading which originates from the mangrove forest, rivers, and aquaculture activity.

The results of the survey undertaken by the regulatory authority and university team show that there has been a deterioration in bottom water conditions since the original survey. The minimum oxygen concentration measured was 4.0 mg l^{-1}. The results of the study reveal two additional factors. First, the area of dissolved oxygen depletion is widespread throughout the deepest part of the bay. Second, depletion does not persist throughout the dry season with periods of depletion being associated with periods of stable weather and low tidal flow. An assessment of the productivity of the bay was beyond the scope of the study and, thus, the investigators were unable to make a direct assessment of whether there had been an increase in the productivity of the bay. However, the team found that near surface concentrations of chlorophyll ranged from 10 to 25 mg m^{-3} during the dry season but maximum concentrations were higher than the levels reported in the original study. Of additional concern to the regulatory authority are the accounts of blooms (red tides) of the dinoflagellate, *Pyrodinium bahamense,* which have been observed in the bay during the past years.

On the basis of a comparison of the data collected during the two studies, the regulatory authority concludes that further enrichment of the bay had occurred. The authority attributes this enrichment to an increase in human habitation and to a lesser extent to the development of the intensive shrimp farm. Given the existing eutrophic state of the bay and the increase in dissolved nutrients and particulate waste (150 to 300 t dry weight ha^{-1} $cycle^{-1}$) which would result from the proposed expansion, the regulatory authority refuses permission for any increase in shrimp production.

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Unacceptable ecological change associated with wastes from coastal aquaculture farms can be minimized by good management practices. Any environmental assessment and monitoring effort should be related to the scale of perceived impact of a given aquaculture operation. An environmental management framework should include an environmental impact assessment (involving the use of predictive models) to quantify significant potential impacts and design a monitoring programme. Flexibility of monitoring undertaken for regulatory purposes is necessary, so that monitoring effort is related to the scale of development and sensitivity of the receiving water body. The choice of which variables to monitor must be based on the nature of the impact and the interpretative value of particular variables. Additional elements of monitoring programmes include: selection of reference stations; standardization of sampling and analytical procedures; analysis and interpretation of data; and feedback mechanisms.

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