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Environmental Capacity Modelling in Aquaculture Development.

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Abstract

Environmental capacity is the ability of the environment to absorb and assimilate activity and effluent and can be treated as an environmental resource, exploitable within sustainable limits. Increasingly sophisticated models seek to calculate and quantify the precise value of a given effluent component, which can be processed within the capacity limits of the environment. As the environmental impacts of aquaculture come under increasing scrutiny, managers, planners and regulators are using models to provide insight into the capacity of adjacent waters.

This paper examines the range of models used in aquaculture development to provide insight into environmental impact of actual or proposed activities. It highlights difficulties in overcoming assumptions inherent in models and reiterates that models should complement rather than replace robust and informed management and planning. In developing country applications, simple models, with effective monitoring and management systems, are likely to be more useful and effective than sophisticated modelling exercises.

Contents

1. Environmental Capacity

Definition

Definitions of environmental capacity vary in the literature, but most refer to "a property of the environment, defined as its ability to accommodate an activity or rate of activity without unacceptable impact" (GESAMP 1986).Widely used alternative terminologies include assimilative capacity and receiving capacity.

Environmental capacity describes an inherent property of the environment to provide environmental goods and to assimilate or process waste and minimise the impact of any natural or anthropogenic activities. It is the calculation of capacity based on the assimilation of waste that models are most often used and which concerns us here. The environmental capacity concept recognises that this ability to assimilate waste without unacceptable consequences is finite. It is therefore implicit that this value can be quantified and apportioned as an exploitable resource of the environment (Pravdic 1985).

Environmental capacity will vary greatly according to local conditions. For example, in coastal, lake or pond environments, well-circulated regions, subject to regular water-mass replacement, will have far greater environmental capacity than adjacent areas exposed to poor levels of circulation (Ward 1999).

It is important to distinguish environmental capacity from carrying capacity. Carrying capacity is the amount of a given activity that can be accommodated within the environmental capacity of a defined area. While environmental capacity is determined by the ecological and physical characteristics of a particular area, carrying capacity is dependent of production technology and characteristics such as effluent volumes and concentrations.
There is some confusion in the literature relating to the various terms. Assimilative capacity and environmental capacity are normally accepted as being alternative terminologies for the same thing, but assimilative capacity is also sometimes referred to directly as carrying capacity (Ward1999). In this paper we use the following terminology:

- Environmental capacity: an inherent property of the environment to provide environmental goods and to assimilate or process waste and minimise the impact of any natural or anthropogenic activities
- Assimilative capacity: an inherent property of the environment to assimilate or process waste and minimise the impact of any natural or anthropogenic activities
- Carrying capacity: the amount of a given activity that can be accommodated within the environmental capacity of a defined area

A Valuable Concept?

Industry makes use of marine environmental capacity by discharging effluents directly into the sea. Consent to discharge is usually closely associated with careful monitoring which ensures that environmental capacity is not exceeded. The difficulty with this (monitor-response) approach is that monitoring will often show an impact only once the finite environmental capacity has been exceeded. This means that efforts to reduce discharges of potentially damaging effluents may only be implemented once environmental damage has occurred (Gray, 1998). Industry can find themselves in a poorly informed and ultimately expensive cycle of cheap waste disposal followed by expensive remedial action and in some cases compensation in line with the polluter pays principle.

As a result of the inadequacy of the “monitor-response” approach, there is increasing interest in more precautionary-based approaches to environmental management. In order to be of value in a regulatory and policy framework environmental capacity must be determined in advance to ensure that it is not exceeded. Informed science can be used to predict the ability of the environment to assimilate wastes, and to set limits to discharges to ensure that environmental capacity will not be exceeded. Environmental capacity is thus used in a proactive or anticipatory rather than a reactive or retrospective fashion. In order to be used in this way, environmental capacity must be quantified at the planning stage. The process of quantifying can be both complex and varied, and capacity values will differ according to local physical and ecological conditions.

Mathematical modelling in ecological science has developed rapidly at the same time as environmental capacity has received increased attention as a regulatory tool, and models have been proposed as a way of exploring these varied and complex interactions. Lessons learned from environmental capacity models used in fields other than aquaculture can provide valuable insight, but there is also an opportunity for aquaculture to lead the way in practical adoption of modelling as part of a sustainable planning process.

2. Environmental Capacity Models

Model Types

Modelling can be described as an interdisciplinary approach to problem solving. Models can be used to help understand natural events, answer questions and predict future impacts. The value of a model is dependent upon the knowledge and understanding that goes into it and all interpretations or conclusions must take full account of the assumptions implicit in model construction.

All models whether simple or complex must go through the same process of model formulation and validation. Typically this pathway involves identifying the problem or question to be studied, determining the availability of suitable data and undertaking system structural analysis to enable model construction. At this point the model should undergo extensive validation and behavioural tests by reconciling with empirical data. Ideally the modelling process should be fully iterative and any lessons learned in validation should be continually fed back into the model design. Only at this point can the model be taken forward for practical use and policy design. Biological systems tend to be non-linear, which immediately rules out certain kinds of models for use in ecological modelling. In spite of this there remain a wide variety of models that can be used and each is appropriate to different situations or different levels of information. The more common types of model used in ecology include time trend models, regression models, bio-energetics models, mass balance models, and simulation models.

In biological modelling it is important to consider the impact of random and unpredictable events. Building deterministic models or simply including a stochastic term enables random perturbations to be ignored or averaged out. The difficulty that remains inherent with this approach is in determining a suitable scale and frequency for events, which by their very nature are random. Data randomness can also be incorporated into the model design by incorporation of Monte Carlo analysis. In short this means that the data set is expanded with huge combinations of randomly derived fictional data, arranged around the existing mean and distributed broadly within the existing standard deviation. This approach can assist with accommodation of random biological events in particular in data poor situations. In biological systems there are often relationships which can not be easily or accurately quantified. This uncertainty can lead to imprecision. In non-biological models the solution to this problem has been to employ fuzzy logic, which enables a certain degree of categorisation and generalisation into otherwise purely quantitative exercises (Lee & Wen 1996). This approach is likely to be applied increasingly in ecological modelling.
The spatial dimension of models is also a critical consideration. Models that address the distribution of an organism in an open space and take full account of migration, grazing and other movement are exceptionally difficult to construct. The danger is that in attempting to incorporate all interactions over a huge spatial scale, the model becomes over complex and the results nonsensical. Determining the appropriate spatial dimension is particularly crucial for models of marine ecosystems.

Use of Models Aquaculture

Can models be of value in the strategic development and environmental management of aquaculture? The first thing to consider is the complexity of a marine environment. Even within a closed production system such as shrimp ponds there are complex biological, physical and chemical pathways and interactions which must all be considered in any attempt to model cause and effect. In the open marine environment these pathways become more numerous and complex and the level of information required to construct a valid model increases.

As the level of aquaculture production and associated research have increased in recent years the understanding of these complex systems and the associated data bank have increased. The understanding has begun to reach a stage where it is reasonable to expect accurate and informative models to begin to provide valuable information to the producer and regulator alike. At its simplest this may be in predicting impacts of one particular nutrient. Some scientists believe that understanding and data are now sufficient to support more complex modelling to enable predictive assessments of whole environmental consequences (Silvert & Sowles 1996), though few models capable of doing this exist and even fewer have been substantially validated.
When looking at the fates of aquaculture wastes, there are a number of different pathways to be considered. Uneaten food, fish faeces, and other metabolic wastes can be dispersed into the environment in either particulate or soluble form (Chen *et al* 1999). These two main pathways are influenced by vast numbers of factors. The physical and chemical characteristics of the receiving water will have a huge influence on the scale of impact of aquaculture production and the relative importance of these two main pathways, as will the fate and ultimate sink for the nutrients within these wastes.

Models have become more complex over time, evolving from initial simple mathematical equations which give an idea of waste build-up, to more complicated ecological models which take account of dispersion and assimilation processes. This evolution has enabled more complicated effects of aquaculture to be estimated including different production levels, variation of feeding regimes and treatment with chemotherapeutants.

A large proportion of the modelling over the last thirty years has focused on the question of particulate deposition on the local benthic community. Studying this area of aquacultural impact has the advantage of being observable and models can be readily verified with field measurements. Smaller but significant numbers of studies have either incorporated or concentrated upon the fate of soluble fraction of the waste product. This is more complicated to model and considerably harder to test and improve models in the ideal iterative fashion.
With advances in the modelling capability, much recent research has begun to consider the impact of increasingly small and numerous chemical reactions associated with aquaculture production. This field of modelling is more complex than prediction of either particulate or soluble wastes. Chemical processes occur on minute scales and many are poorly understood. Numerous interactions within the environment further complicate the accurate prediction of chemical fates over wide distances.

<u>Particulate waste models</u>

The main body of modelling work associated with the impacts of cage aquaculture has been in determining the effects of particulate sediments deposited on the seabed. Many models began life as simple research tools or intellectual exercises. It is perhaps unsurprising therefore, that most investigators opted to study the relatively easier area of particulate fate and effects. As this was the initial association between modelling and aquaculture, many lessons and refinements have occurred and modelling of particulates now leads the way in terms of widespread application in aquaculture management.

At its simplest, modelling of particulates fate includes a determination of the amount of waste food or faecal matter, tied in with sinking rates which takes into account the distribution effects of the current. Increasing complexity or reality may be added to the model through incorporation of factors such as variable waste production, digestibility and sinking rates or extend to predict the impact on benthic communities or more complex hydrodynamic effects. Progress has also been made on modelling increasingly complex and diffuse sources of effluent. At first models were only capable of dealing with single point sources but, with advances in modelling technology and theory, diffuse sources can now be modelled which is of greater value, in particular to cage aquaculture.

Initially the models used were simple summaries of waste volume, settling velocity and current flow. Many of the assumptions that have traditionally been relied upon have been shown to be unreliable and often lead to inaccurate estimates of effect. In spite of this, the majority of particulate models today still rely upon calculating those same three basic factors, although the mathematics behind the model may have become more refined and the models may now concentrate different areas.

Several studies have highlighted the shortcomings in assumptions in particulates modelling and this has often provided the impetus for refinement of models or incorporation of an extra term in the calculation to include an extra factor (Gowen *et al* 1994; Chen *et al* 1999). Many of the early models of particulate distribution failed to consider realistic bathymetry, vertical gradients and mixing in their calculations (Gowen *et al* 1995). The fact that these factors are now considered and incorporated in all but a few models that are expressly designed for use in shallow areas or closed systems, indicates the evolution of this field of science. It is likely that in the future models will give increased consideration to salinity, digestibility of diet, pellet friability and water absorption properties as these have been highlighted in recent years as factors that influence the settling velocity (Chen *et al* 1999). Additionally, other factors influenced by the physical environment are also being considered such as the influence of tidal currents of movement of fish cages and turbulent mixing due to wave action (Corner, unpublished data).

Some of the particulate waste models that are in current use in aquaculture regulation and management have built upon previous model inadequacies to provide relatively robust and accurate predictions of particulate fate and resulting impact. The Scottish Environment Protection Agency now make full use of the DEPOMOD model (Cromey *et al* 2000), which predicts the sedimentary deposition of waste feed and faecal matter from cage finfish aquaculture in the Scottish Sea Lochs. The model takes into account cage positions, feed regime and food conversion ratio (FCR) to determine waste amounts. The model includes routines to track particles during initial settlement and subsequent re-suspension. In addition the prediction of impact extends to the trophic changes in benthic community components, relating output from the models to the estimated Infaunal Trophic Index (Word, 1979), which can be used as an indication of disturbance on benthic macrofaunal communities.

An important debate and one which will be returned to later is the question of the dependence that should be placed upon models in the environmental decision making process. In short, to what extent should objective modelling overrule informed subjectivity? Cromey *et al* (2000) have sought to answer this question by incorporating an expert user interface into the DEPOMOD framework. This user interface allows expert knowledge to be incorporated to inform the final conclusion of the model. However, due to the complexity of DEPOMOD, and its requirement for environmental data, it is often not used to its full potential. At present production information and environmental management of fish farms is not taken into account and modelling is based on an assumption of consistent (steady state) maximum biomass. Presentation on data and direct use of such models in the context of coastal management is being addressed by using geographic information system (GIS) technology and software for particulate waste dispersion models, using multi-cage systems in enclosed areas (Perez *et al* 2002). Here models, such as those for waste dispersion calculation, can be used directly as a “layer” within a GIS coastal management model rather than simply supplying secondary data for *post hoc* input. Thus aiding the decision making process (Perez *et al*, in press).

Another discussion that will also be addressed later is the problem of complexity. With increasing pressure from all angles to consider all possibilities and environmental factors in the model design, complexity often reaches the limits of realistic contemporary computational capacity. This is a debate to be included in a wider discussion of the role of models but it is also a debate that has featured in the development of particulate models.

A particular difficulty is in structuring particulate models to accommodate sufficient locally specific parameters without leading to excessive complexity. One solution to this problem is to use a hierarchical approach. This enables certain common parameters to provide the foundations to the model regardless of the system being assessed. On top of this foundation, locally specific parameters can be incorporated into the model. In this way the hierarchical approach provides a compromise between the demands of site-specific and globally common models.

Borsuk *et al* (2000) tested a Bayesian hierarchical model for organic matter loading and benthic oxygen demand in 34 estuarine and coastal systems. This showed that by adding locally specific parameters, the hierarchical model looses some precision by effectively reducing the amount of information that is available for each parameter. In spite of this, the inclusion of locally specific data means that the model is more realistic and represents a fairer translation of our knowledge. The combination of generality and simplicity that the hierarchical model offers means that it is particularly suitable for modelling of particulates from aquaculture.

<u>Soluble waste models</u>

It is increasingly realised that the modelling of particulate wastes is essential but insufficient: there is a need to consider the dilution and dispersion of soluble nutrient and chemical wastes. Scientists are increasingly recognising the importance of understanding long-term and large-scale nutrient impacts from aquaculture and other activities – on other users, on aquaculture itself, and on the quality of the wider environment and ecosystem services. The lack of understanding and the remaining uncertainty in this area has become more apparent in the light of the recent advances in understanding that have been achieved in relation to particulate benthic deposition. Gillibrand & Turrell (1997) recognised this and targeted their modelling work on reducing the uncertainty surrounding the fate and impact of the soluble fraction of the aquaculture effluent. In particular their work focused on the dissolved ammoniacal nitrogen excreted by the cultured fish. This work looked at the effects of the soluble fraction of waste from Scottish salmonid production. The model makes simplifying assumptions, in particular in relation to the loch hydrography by representing them as rectangular boxes with characteristic mixing and flushing times. These assumptions would need to be carefully validated before applying to other systems. The equilibrium concentration enhancement model characterised the exchange of water and the rate of ammonia excretion. This excretion rate is taken as standard but can be adjusted for use with higher energy, lower protein diets. Any dissolved nitrogen is also assumed to be conservative. Because of these assumptions Gillibrand and Turrell (1997) recognise that any absolute values predicted by the model are unlikely to be totally realistic, but this does not necessarily devalue the models use as a management tool. Indeed the system overcomes this by presenting the results as a ranking series highlighting the key areas where nutrient enhancement is most likely to be a problem.

Other studies have gone into greater detail relating to soluble waste fates, however certain underlying assumptions remain. Gillibrand revisited the question of the fate of soluble waste, this time comparing two different approaches to modelling soluble dispersion in Loch Fyne (Gillibrand 2001). This study went further in to determining residence of nutrients based upon exchange times with greater depth consideration and through use of a two dimensional laterally integrated numerical model, which could be improved with inclusion of a simple return flow factor. This study was validated with the use of passive conservative tracer, which revealed that the model was of greater value than the commonly used tidal prism method.

More advanced and complex spatial models have been developed for investigating the environmental fate of dissolved nutrients. One such model for fish farming effluent in the Baltic

Sea has been developed from hydrodynamic models specific to this area; FINNFLOW and FINESS models (Inkala and Myrberg, 2002). These models have been developed further into 2 and 3 dimensional models for the Gulf of Finland by EIA Ltd. These models have been used to good effect in the siting on intensive fish farms growing seatrout in the Strom Strait looking at the dispersion and dilution of soluble nutrients wastes (see http://www.eia.fi/).

Again the trade off between the need for locally specific detail and a limitation on the level of complexity that can practically be incorporated is evident. This trade-off can be regarded as a recurrent theme in projects to create environmental models used in aquaculture. Gillibrand makes the point that an assumption over a uniform exchange time is necessary so that the models can be applied quickly to enable rapid decision making. This removes the complexity that is evident in the multi-box models seen elsewhere in aquaculture modelling but still produces a realistic result. In spite of this adopting a uniform flushing time means that the model is highly locally specific and would need to be adapted for use in any other system, particularly in larger lochs where individual basins would need to be considered separately (Gillibrand 2001).

Chemotherapeutant dispersion models

Increasingly with modern intensive aquaculture, public fears are raised by the notion of chemicals, or chemotherapeutants, being added into the marine environment with poorly understood, but none the less damaging consequences. It is increasingly recognised that in order to allay public fears and to ensure long term environmental sustainability that the uncertainty surrounding many of the smaller and more complex chemical processes associated with marine aquaculture are addressed.

Initial work relating to chemical transport from aquaculture production has been based upon estimates of the fates of conservative chemicals. Relatively speaking these models are simple but none the less generate useful predictions based upon a large number of assumptions. Work by the Fisheries Research Services Laboratory in Scotland looked at the fate of dichlorvos sea lice treatments as an extension to work studying the fate of dissolved excreted ammoniacal nitrogen. This work aimed to model the dispersion of dichlorvos in order to calculate the total annual amount that could be used without exceeding the proposed environmental quality standards. In addition a second model predicted the typical dispersion of the chemical following a lice treatment (Gillibrand & Turrell 1997). Although dichlorvos has a half-life of 6 days, as most of the lochs have flushing times of less than 6 days for the purposes of the model, dichlorvos is assumed not to decay. The flushing model enables the total amount of water replaced during a year to be calculated and this enables the maximum total dichlorvos amount to be calculated. To determine the fate of the dichlorvos from a single treatment, a simple advection diffusion model was used. Advection is simulated using the particle tracking approach and diffusion is simulated using a Gaussian equation, which, in this case, includes a factor for dichlorvos decay. This model considers the mass released at source, the nature of the surface mixed layer, the half life and diffusion coefficients to calculate the concentration at a set location after a set time (Gillibrand & Turrell 1997).

Although useful, these models are essentially modelling the circulation patterns of the water body along with the typical dosage patterns. There is no consideration of reactions that occur in the water body. By choosing to look only at conservative and passive chemicals, this work avoided many of the harder questions that need to be answered to reduce uncertainty over the effects of aquaculture. By contrast attempts at modelling reactive chemical fate tackles the largest remaining area of uncertainty head on.

The calculation of water borne chemical reactive transport to determine the fate of numerous chemical reactions associated with aquaculture production is perhaps the most detailed and complex form of aquacultural modelling. Perhaps unsurprisingly, the work in this field is more likely to come from general environmental modelling and computer science areas of study than from aquaculture. The goal of aquaculture will be to interpret this work and apply it accordingly. Cheng, from the Department of Nuclear Science at National Tsing Hua University, Taiwan, in association with scientists from Pennsylvania State University recently published details of a numerical model that simulates the reactive transport of a number of chemicals in a shallow water domain (Cheng *et al* 2000). This model includes more chemical reactions than before and considers chemical kinetic reactions dissolved in the water column, within interstitial spaces of the sediments or sorbed to particulates.

The fact that it is a two dimensional depth averaged model will open it to criticism due to its inability to be used in most bathymetrically complex marine systems (Gowen *et al* 1994; Roth *et al* 2000). These limitations do indeed represent some obstacle on route to application of the model in aquaculture, but this publication does represent a significant milestone on the road to comprehensive chemical modelling of the fates of marine wastes. As the authors rightly point out, "no existing water quality model, to our knowledge has used a fully mechanistic approach to estimate non-conservative chemical transport in shallow water domains".

Environmental Capacity Models

Environmental capacity models represent an extension of the models described above, to generate practical guidance on appropriate limits to aquaculture effluents. They compare predictions of impact made using the above models with environmental quality standards. This allows for appropriate limits to effluents or levels of aquaculture activity (such as production) to be estimated.

There are two basic approaches to environmental capacity modelling. The first is to consider the capacity of the environment to absorb, dissipate or disperse a single component with no discernible effect. This single component is regarded as an indicator for the whole environment. Choice of indicator is vital as this must provide a representative indication of environmental impact, and when possible provide initial warning of more serious and longer term impacts. The second approach to environmental capacity is to attempt to map, model and understand all major processes that may influence the complex relationship between man's activities and the environment. This includes consideration of all major particulate, soluble and chemical pathways to identify and predict potential impacts of an activity.

The decision over which approach to take to the modelling of environmental capacity is essentially a trade off argument. The benefit of concentrating on one component as an indicator for other environmental processes is the relative simplicity of modelling. The obvious trade off is that other environmental consequences may go unnoticed if the indicator is not sufficiently representative of all processes involved. By contrast, including many more complicated pathways into consideration avoids accusations of overlooking important processes, but adds tremendous complexity to the model that could reduce, rather than add to clarity.

Simple Capacity Estimates

Perhaps the simplest form of environmental capacity estimation, and for this reason some would argue the most robust, is linear regression based on the relationship between an influencing factor (e.g. aquaculture production; nutrient load) and a perceived impact (such as sediment quality, water quality, biodiversity). This requires a historic data set of both the influencing factor and the perceived impact. Determining exactly what are the most important influencing and impacted factors is not always easy, and multivariate analysis, correlation analysis and/or many linear regressions between different factors may be required in order to identify critical relationships.

Once a correlation between the main input and changes in the environmental condition has been established a historical data set is required for these two factors. In the past this approach to environmental capacity determination has frequently been used for industrial effluents which often have one particular component of concern. Often this will be something toxic or conservative that has a direct pathway to the human food chain, such as a heavy metal. A good example of this approach to capacity modelling was Krom *et al* (1990) who used a simple linear regression to relate inputs of mercury from a chlor-alkali plant effluent into Haifa Bay' Israel, with the mercury content in the muscle tissue of *Diplodus sargus* - a commercially important species with a critical path to man. The relationship derived allowed for the levels of effluent that caused harmful levels in the muscle tissue to be established. In the Firth of Forth Estuary in Scotland a linear regression approach was used to determine the relationship between industrial mercury discharges and mercury concentrations in skeletal muscle of the territorial benthic eelpout (*Zoarces viviparus* L). Again this revealed a significant linear relationship This was compared with environmental quality standards (EQS) to determine the level of discharge at which EQS would be exceeded (Mathieson *et al* 1995). The work of both of these teams is notable due to its simplicity and practicality. Environmental capacity is taken as the level at which the EQS is exceeded for a single discharge component accumulating in one species. This may be useful in situations where there is one major effluent of concern and perhaps one main or representative commercial species. However, there is no consideration of wider ecological implications or indeed other effluent components.

In the context of aquaculture this approach may be of use for the testing of the impacts of certain persistent toxicants in chemical therapeutants that are increasingly used in intensive production. From a holistic management approach, particularly which seeks to enable pre-emptive planning, the value of this approach is limited. Detailed determination of environmental capacity should ultimately seek to consider a wide range of effluents, over a wide range of ecosystems and ecosystem components. This inevitably leads to more complicated models incorporating a wide range on environmental parameters and pollution pathways.

<u>Complex Capacity Estimates</u>

Where detailed historic data on relationships is lacking, it is more usual to make an estimate of capacity by combining a series of deterministic models in order to build up a picture of the complex interactions that will influence the scale of environmental impact. Most of the models, which consider particulate, soluble or chemical pathways incorporate a hydrodynamic aspect. This allows for greater precision in describing spatial variations in impact, related to source location and effluent dispersal within a wider system.

Ward, of the University of Texas at Austin has worked on combining these various elements of impact from aquaculture into a single environmental capacity estimation. Estuarine circulation is influenced by morphology, tides, freshwater flows, meteorology and density flows. Because circulation is the major factor influencing the assimilative capacity of an estuary it is important that all of these variables are incorporated into any assimilative capacity models. Any capacity calculation also requires a determination of water quality parameters and their acceptable values, before developing a model to establish the conditions critical to maintain these values (Ward 1997). Ward developed a combined mass balance and hydrodynamic model that is sensitive to both spatial and temporal queries and is based upon real field data. The model is able to determine the assimilative capacity of the estuary by determining the maximum amount of waste that can be discharged without having a negative environmental consequence. This result of the calculation varies according to the position in the estuary and will be influenced by local and larger scale hydrography. Initial case studies in Golfo de Fonseca in Central America indicated that existing shrimp culture is approaching self-limiting scales (Ward 1997). Later work in Honduras in association with Peace Corps volunteers aimed to establish assimilative capacity in relation to shrimp mariculture in estuarine systems. The project resulted from concerns relating to the intensive development of the industry in recent years and the prospect of an approaching 'self-limiting density' caused by accumulation of waste by-products. This project identifies this self-limiting density as a basic management parameter and attempts to develop models capable of determining its level in order to ensure the viable future of the industry (Ward 1999).

The project evaluated existing impacts of current shrimp farms in the estuarine study region through data collection and modelling. The level of complexity of data and model is commensurate with the complexity of the system and is dictated by spatial and temporal variations in the estuarine dendritic tidal flats. The model includes determination of the hydrodynamic characteristics of the system and the mass balance of water quality constituents through the use of two models. The first is a section-mean hydrodynamic model which describes the rapid fluxes in tidal and current flow and the second is a section-mean longitudinal mass budget model which describes slower, longer term concentration profiles.

A range of different scenarios tested illustrate that tidal flow, freshwater throughflow and topography are at least as important as the nature of chemical usage. This means that any impact is influenced substantially by location as well as load and reinforces the point that the discharges have effects a great distance away from the source (Ward 1999).

In Norway a comprehensive system is now in place which brings together estimates of the environmental capacity of different coastal zones with farm related modelling of loadings, dispersion and dilution. The LENKA programme divided the Norwegian coastline into distinct areas. For each a capacity, such as for dissolved or particulate nutrients, was calculated and allocated to the various contributing activities (aquaculture, agriculture, forestry, sewage etc) (Bergheim, et al., 1991; Ibrekk et al., 1993). MOM (Modelling – Ongrowing fish farms – Monitoring) is then used to estimate the holding (carrying) capacity of the site by modelling waste dispersion and dilution, and taking account of environmental capacity assigned under LENKA (Ervik et al. 1997). Further information gathered during monitoring is used to refine the model and thus the consented fish production. MOM integrates elements of environmental impact and environmental quality standards (EQS). The monitoring regime is governed by the degree of exploitation, where higher biomass sites require greater amount of monitoring. There are three levels of complexity.

More commercial products are now also becoming available for making assessments of the impact of industrial processes including aquaculture. Many offer a suite of packages which when run together provide all the information necessary to make assessments about the environmental capacity. For example, a combination of a hydrodynamic model, a particulates model and a chemical processes model provides a fairly comprehensive indication of the impacts. Many of these packages are available in downloadable form from the Internet, and some are free. These packages may originate from academic institutions while others are commercial products. Providers include for example Powergen (UK), the Danish Hydraulics Institute, WRC (UK)
For example, Powergen offers a suite of tools for environmental capacity estimations. This suite comprises TRISULA and DELWAQ both of which model heat dispersion, sediment transport , water quality and heavy metals in combination with CORMIX which models near mixing to give an indication of plume characteristics. This suite of programmes can relate effluent levels to predetermined environmental quality standards and inform management questions such as optimum outfall location, interaction of discharges and levels long term accumulations.

There are examples of these sort of combination approaches to environmental capacity modelling being successfully applied. This technique was applied in the Great Vitoria region to determine the best area for sewage treatment plants. This work used a suite of models that again included CORMIX along with WASP (Water Simulation Program) and QUALBAVI to predict BOD, DO, faecal coliform, nitrogen and phosphorous data as part of the environmental capacity calculation. The majority of these suites of programmes, and in particular, any which use the CORMIX programme, are more suited to point sources of pollution. This would mean that in their current form they could not be used to model diffuse effluents from cage aquaculture. In spite of this the potential of suites of tools is clear and even in their current form these packages would be suitable to predict and the effects of point discharges from pond culture.

Taken together Ecopath and Ecosim (Christensen, Walters & Pauly, 2000) enable the construction, parameterization and analysis of mass-balance trophic models of aquatic ecosystems. These models give an overview of the feeding interactions in the ecosystem, and the resources it contains. This enables analysis of the ecosystem and simulation of various levels of exploitation, and thus allows evaluation of the relative impact of fisheries and environment. This model may be used indirectly to estimate the effects of pressure on native stocks from anthropogenic impacts, such as aquaculture, but not directly in the calculation of the carrying capacity.

New approaches are also beginning to appear which attempt to overcome the problem of imprecision and uncertainty that is inherent in all environmental capacity modelling by use of fuzzy logic modelling techniques (Lee & Wen 1996). These fuzzy techniques have significant advantages in providing robust solutions in imprecise environments. The disadvantage is that fuzzy models are often data hungry and always require considerable advanced computational capacity that inevitably adds to the cost of the project.

Fuzzy logic resolution of water quality problems may be particularly appropriate when viewed in comparison to conventional deterministic mathematical programming models. These introduce uncertainties at all stages from model formulation through to computational procedures and error rates of these models may be as high as 300% (Lee & Wen 1996). In addition, conventional approaches to environmental capacity aim to relate the impacts against imprecise environmental goals and subjectively determined environmental quality standards. Fuzzy systems are ideal for overcoming this degree of uncertainty and subjectivity and seek to make general but robust conclusions. In the future this is undoubtedly and area of environmental capacity modelling that will receive increased focus.

<u>Non deterministic alternatives</u>

Some have called into question the whole deterministic approach of conventional environmental monitoring and assessment. In particular, the inherent acceptance of a large amount of uncertainty and the packaging of subjectivity into a highly objective framework has been questioned (Lein 1991, Hogarth and McGlade 1995, Lee & Wen 1996). These questions apply equally to deterministic assessment of environmental capacity. The driving force for any alternative approach is principally the view that conventional approaches have failed to adequately deal with the grey area of uncertainty and in particular how this is incorporated into the assessment procedure.

As can be seen from previous sections of this paper, some of the bottlenecks that existed in the determination of environmental capacity have over the years been removed as modelling, data collection and computational ability has increased. In spite of this there remains an argument that this advancement has simply disguised rather than removed the problem of uncertainty.
James Lein, from the University of Ohio stated that uncertainty is an inherent feature of any environmental system and that this should be used to the advantage of the decision support systems rather than being regarded as an obstacle (Lein 1991). A prototype knowledge based expert decision support system is proposed as an alternative to deterministic approaches to defining a numerical value of carrying or environmental capacity. This system recognises that capacities are heavily influenced by human subjective appraisal of environmental quality standards and environmental processes.

A knowledge based solution to determining environmental capacity still requires a similar set of initial considerations, namely, the selection and measurement of key components and a study of their interrelationships. This approach differs however in that it aims to qualitatively evaluate the human-environment balance in order to determine the point at which the equilibrium is upset, rather than attempting to define a fixed numerical value for environmental capacity. A knowledge-based expert system is a logically arranged collection of facts, experience and insights that yield specific conclusions to a series of user operational prompts. Lein developed the C-CAPACITY prototype and tested it in Kenya's eastern ecological gradient, building up the knowledge base through interviewing of local experts. This work clearly showed the potential of this prototype for further development in order to provide valuable insight into environmental capacity planning. The commercially based SIMCOAST package available in the UK combines many of the features of Lein's C-CAPACITY prototype but stretches further by including some fuzzy logic deterministic queries. Again this recognises the value of informed subjective opinion provided by experts, but supplements this with fuzzy analysis of objective data sets. SIMCOAST describes itself as a transect based decision support system for improved coastal management which approaches the problem of trans-boundary pollution analysis from a highly multidisciplinary viewpoint. Because this tool is so multidisciplinary there is a high data requirement, but the rule based expert system and the fuzzy calculations enables the package to cope with a certain amount of data inaccuracies.

The approach of Lein (1991) and the SIMCOAST team (Hogarth & McGlade 1995) initiates a debate over when is it best to rely upon subjective informed judgement and when on objective modelling based on necessarily uncertain parameters and relationships. This issue will be returned to in the next section.

3. Future trends and potential

Level of Complexity

D.B. Lee (Jr) stated in 1973 that bigger computers simply permit bigger mistakes. Ecological models can describe events from the microscopic to whole biosphere scales. The models can simply comprise of one or two equations but may equally demand massive computer simulation. The complexity of a model is not dependent upon the spatial scales that are covered but rather on the number of variables and the precision with which they are specified (Silvert, 2002).

The temptation is often to produce big models, which consider all components of the ecosystem even though these may lead to complexity problems. Uncertainty is compounded in a chain of uncertain relationships, and the conclusions generated may be associated with huge margins of error. While most models have under gone sensitivity analysis as part of their validation, they do not offer assessments of the reliability of final outputs for each individual simulation. This drive for big models may in part be explained by a concern over criticism for leaving information out, in spite of the fact that this may lead to a more robust model providing reliable information. A more cynical point of view is that it is easier to attract funding for large and ambitious models and it is harder to find fault with them due to their complexity and implicit knowledge of understanding (Silvert, 2002).

Almost all model authors recognise the trade off between inclusion of sufficient variables and the resulting complexity. Model designers have struggled to incorporate the major physical, chemical and biological processes into their work without compromising the overall clarity of interpretation and operation (Ross *et al* 1993). Different approaches to overcoming this problem have been used and showed to be of potential future value. Several model designers have opted for hierarchical type models which enable generic assumptions and rules that are uniform for all marine situations to form the base of the model while more detailed local specifics can be added into this structure (Borsuk *et al* 2000). This combination of generality and simplicity is proving useful in ecological modelling applications.

Perhaps in the future, with increased computational power and modelling ability the point of compromise between increasing amounts of realistic but complicating parameters and the need for practical simplicity will shift. Models may aim to seek a level of complexity that is commensurate with the complexity of the system (Ward, 1999). However, if this becomes the case, the value of the models may not necessarily increase. The concern will be that with the inclusion of terms for an increasing number of (uncertain) factors, the conclusions of the models will increasingly be accepted as objective statements of fact. There is an inherent natural caution that decision-makers use, or should use, when interpreting evidently simple models. This caution adds rather than detracts from the models by encouraging conclusions to be interpreted and implemented with informed subjectivity. The worry will be that with increasingly advanced and complicated models the assumption of the need for a subjective interpretation will disappear and reliance on conclusions would increase. The greater the complexity, the greater the temptation to forget the assumptions and believe that all factors have been considered. This point is of considerable importance in the climate of extensive model use by governmental regulators to make decisions which significantly effect aquatic resource use.

Data Limitations

Any attempts to model environmental capacity are attempts to provide answers and insight in areas of significant uncertainty. This insight aims to reduce the uncertainty and enable informed exploitation whilst remaining within the capacity of the environment and ensuring sustainable development. In short modelling should increase understanding enabling informed and sustainable resource exploitation.

This sounds ideal, but many feel that this is an exercise in worshipping false gods (Thompson 1985). The Precautionary Principle is becoming increasingly widely adopted at regional, national and international level in relation to management of environmental resources. Many see attempts to model environmental capacity as contrary to the precautionary principle, since models imply (falsely it is argued) reduced uncertainty and lesser need for precaution. They argue that regardless of modelling advances, the complexity of the environmental processes mean that data limitations will always create sufficient uncertainty to trigger precaution (Stebbing 1992). Conclusions drawn from models rarely fully acknowledge the assumptions upon which they are based and the associated levels of uncertainty.

It is vital therefore that all models clearly express the assumptions upon which they are based and in drawing conclusions highlight the areas of uncertainty that remain. Models must be seen as a means to enhanced understanding rather than providing answers. If the modelling world claims to identify hard and fast solutions from comprehensive data sets then the frequency of contrary voices that aim to discredit the models will increase.

Limited data sets

Certain pieces of work have highlighted the inherent weakness of models caused by data limitations and have looked at ways to overcome these difficulties. Models such as the non-linear iterative least squares approach has been used in India to predict environmental impacts specifically in areas of small or inaccurate data sets.(Agarwal & Sharma 2001). The hierarchical approach, which has already been mentioned in the discussion of the particulates modelling work, is another approach to overcome problems of limited local information (Borsuk *et al* 2000). This approach enables certain common parameters to provide the foundations to the model regardless of the system being assessed. On top of this foundation, some locally specific parameters can be incorporated into the model. In this way the hierarchical approach provides a compromise between the demands for site-specific information and the use of globally common data. The difficulty with situations of limited data is that the mathematics and computational effort that is required to overcome the data shortage is often huge.

Interpretation of model conclusions

It is clear that the conclusions derived from models of complex physical and ecological processes must be treated with caution. Simple models may be robust but fail to take into account a wide array of location specific influencing factors. Complex models take account of these factors, but compound uncertainties through chains of relationships and assumptions. A failure to recognise these assumptions may cause models to obscure the obvious. For example, a model may predict that a predator dies when food is exhausted triggering a chain of events. In reality, however, in the absence of food the predator would simply migrate out of the spatial limits of the model. As the model assumed a finite spatial limit it is unable to consider this eventuality (Silvert 2002). Comparative studies of the findings of different models have often highlighted this danger. In many cases where the same problem or environment has been modelled in different ways the conclusions have conflicted. For example, modellers, looking to determine carrying capacity for mussel raft systems in Loch Etive and Loch Kishorn developed two different capacity models. Both considered current, mean filtration and seasonal assimilation rates. A seston clearance model concluded that in both cases operations were well within the carrying capacity, whereas with a particulate organic matter model concluded that Loch Kishorn production exceeded the carrying capacity of the system (Karayuecel & Karayuecel 1998). This clearly illustrates the danger of placing absolute faith in model conclusions.

This variation between models is at odds to one of the apparent benefits that models are said to bring, namely consistency in planning in different locations (Lein 1991, Gillibrand & Turrell 1997). This presents a problem. If different models are to be used in different locations then the conclusions and the resulting levels of exploitation will vary. In effect this will lead to different degrees of environmental impact in different areas. Whist some models may advocate exploitation which remains well within precautionary levels, other models may draw conclusions which enable exploitation closer to the point of exceeding environmental capacity.

Clearly therefore in order to ensure consistency in planning a suitable model or suitable suite of models should be advocated for use across a wide range of situations. However promoting one model ahead of another is to face accusation of academic or commercial bias, and undermines competitive science. A better approach may be to highlight generic model types or combination suites of model types which should be used. Highlighting key considerations that should be included as standard in any model design may also be a useful way of ensuring that appropriate models are used across the board and reduce the variation in conclusions between sites.

4. Application of environmental capacity models in developing countries

Most modelling of environmental capacity related to aquaculture has been undertaken in developed countries, particularly focusing on marine cage culture of salmonids and mollusc culture in enclosed coastal embayments. In spite of the recognised importance of nutrient management for the productivity of tropical aquaculture systems, fresh and brackish-water pond production has received little specific focus in either the application of environmental capacity or the development of appropriate tools to measure and assess it.

Aquaculture systems in developing countries range from highly extensive to highly intensive, and they are often much more closely integrated with agriculture and in some cases forestry systems. Concerns that pond aquaculture, especially shrimp farming, is exceeding environmental capacity are frequently voiced, and it is often assumed that the high disease incidence in shrimp farming is related to environmental degradation. Intensification is likely to increase in the medium term and environmental capacity issues will become critical very quickly. It is vital that the tools for assessment, and appropriate management and design are in place before further problems arise. The practical and technical restrictions on the application of modelling techniques in developing countries, vary from those experienced in developed countries. Although the currently-favoured modelling techniques could be applied in developing country applications, in reality practical constraints often prevent this. Access to adequate funding, technology, training and experience may combine to reduce the applicability or appropriateness of many of the more sophisticated modelling techniques. In spite of this, there remains considerable scope for employing appropriate modelling techniques to examine and evaluate environmental capacity for aquaculture management. Simpler and less data hungry tools, less reliant on sophisticated computation may be particularly appropriate and still capture the same principles of environmental capacity which are exploited by the more technical solutions.

The low costs of skilled labour mean that simple water quality sampling may not be so expensive, and if this can be linked to the development of simple field indicators which can be used by farmers, improved environmental management techniques should develop and spread rapidly. Environmental impact and capacity estimations can be addressed in a variety of ways ways, according to local problems, conditions, and technical capacity. Research is currently on-going into the appropriateness of many lower-tech and more cost effective applications of modelling, which build upon the key principles of environmental capacity. In each case the utility of the different approaches is determined by their cost-effectiveness in generating:

- Improved understanding of the links between management practices, water quality, sediment quality, and farm performance;
- Improved understanding of the relationships between water quality in ponds, in the defined aquatic system, and in the wider aquatic environment;
- Improved understanding of the nature and value of simple indicators of environmental quality and their use for routine water quality management
- Guidance on management of ponds and/or wider aquatic systems to generate optimal conditions for aquaculture and other productive activities;
- Estimates of possible limits to intensification or increased production for aquaculture (and agriculture where relevant) within the defined aquatic systems;
- Guidance as to system changes (e.g. canal design; water gate protocols) which would lead to improved water quality and or potential for increased production without compromising the quality of the wider environment

Farmer level indicators of environmental health and nutrient status

At it's simplest, the principles of environmental capacity can be applied through extension of observations undertaken as part of existing standard husbandry routines. Farmers already use a variety of indicators to assess water quality and growing conditions, such as water colour, sediment colour, smell, presence or absence of common organisms. This can be greatly enhanced through further refinement of observation and recording protocols, including record keeping which highlights management interventions (stocking, fertilisation, feeding), climate and water events (such as heavy rain or water exchange), and measured water quality parameters (typically relating to temperature, salinity, dissolved and bound nitrogen, total and reactive phosphorus and biochemical oxygen demand).

Information on indicators, salinity and nutrient levels can be plotted over the production or annual cycle, along with major management and other events. The relationships suggested in these plots can be discussed locally between farmers to build upon the utility of indicators in pond or area water management.

Gross nutrient budgets

Total inputs to individual ponds, cages and the shared aquatic system of nitrogen, phosphorus, and in some cases organic matter can be estimated as a next step in environmental capacity assessment. Outputs in the form of harvested organisms, and nutrients and suspended matter in water remaining in the system or exchanged with the wider environment (in open systems) are also estimated. The balance of nutrients retained within the system or lost to sediments and/or atmosphere can then be estimated. These estimates will immediately provide an indication as to whether the system is a nutrient sink or source.

For those systems which are used for rice production as well as aquaculture, the nutrient budget associated with rice cultivation can also be estimated, and possible nutrient exchanges between the two farming systems assessed. Ideally subsidiary budgets should be developed for each sub-system (e.g. ponds, minor canals, major canals) so that accumulation or removal of nutrients in different components can be estimated, and management interventions agreed where appropriate.

Simple mass balance/flushing rate models

The nutrient budget estimations of the distribution of nutrients between the defined system and the wider aquatic environment, and between sub-components of the system may be refined and/or cross checked through the use of mass balance or flushing rate calculations. If nutrient concentrations within system components are known (or can be estimated from inputs), if flushing rate (water exchange/water volume) can be estimated, and if perfect mixing is assumed, then the total flux and balance of nutrients between sub-components can be calculated simply (see for example GESAMP 1986, 2001; Barg 1992; Hambrey et al 2000). Water exchange may be estimated based on tidal height differences or on the actual flow of water into and out of system components where this is controlled at a sluice gate.

Nutrient/organic matter dispersion and dilution

The final refinement to our understanding of nutrient relations within aquatic systems is to model or plot empirically, nutrient gradients and or sediment dispersal and settling within and between components. This can be costly, and is clearly only worthwhile where there are significant and important (in terms of aquaculture sustainability) nutrient gradients within sub-components, which could be manipulated through management intervention if necessary. Simple empirical data on nutrient gradients in critical parts of the system may be collected and plotted to determine the need or otherwise for more data, and the likely value of dispersion and dilution modelling.

5. Conclusions

Aquaculture production in developing countries has often expanded in an unplanned way leading to the exceeding of environmental capacity, and negative impacts on aquaculture itself and other resource users. Action has usually been taken too late, and corrective interventions have sometimes been prohibitively expensive. Shrimp farming in particular has collapsed in many regions as a result of environmental degradation and chronic disease problems. There is tremendous potential value in using models to predict environmental effects, to inform farmers and other resource users, and to set agreed upper limits to aquaculture and other resource use activities to ensure that environmental quality does not deteriorate below unacceptable levels (Gowen 1994; Rydin 1998; Barg 1992; GESAMP 1986, 2001). The alternative is to monitor environmental conditions more rigorously and continuously and to implement appropriate corrective measures when problems arise. A more precautionary approach can be taken in either case: predictive models can be used, but precautionary limits still set; or precautionary "early warning" indicators can be identified and monitored, and management responses implemented well in advance of serious problems.

The relative costs of these different approaches are debatable, and likely to vary greatly according to circumstance. There is little doubt however that any system which reduces the chances of widespread environmental degradation, chronic water quality and disease problems, will save very large amounts of income and export earnings, and greatly reduce the incidence of personal debt and bankruptcy.

Despite the clear rationale outlined above, the use of predictive models has been criticised. There is a danger that model conclusions are taken as definitive and objective, despite the fact that many of the processes are poorly understood and numerous assumptions are required. This valid criticism requires that models be seen as an aid to the planning process rather than a process replacement. The value of the tool is wholly dependent upon the institutional systems that support it and does not in anyway replace those systems (Kapetsky 1995). Whichever models are used and regardless of their levels of complexity or technology, they must be seen as part of a wider and clearly defined management structure and planning process. It is this structure rather than the eventual choice of tool that will determine the overall success of any inclusion of environmental capacity determinations in sustainable aquaculture regulation. It is also clear that models do not and cannot absolve planners and regulators from responsibility, or provide a 'get-out' clause, in the event of environmental degradation.

Any useful model must be iterative. Willingness to admit that the model contains inaccuracies is vital, and models should contain sufficient flexibility to enable lessons to be constantly fed in and refinements made. Obviously this must include information from careful and appropriate monitoring which serves as a vital part of any quality control of management system. Environmental capacity models should also be seen as part of a suite of new and wider ranging policy tools which can consider and plan using environmental capacity concepts and enable focussed and informed planning at global and local level (Rydin 1998). An ideal planning process would bring together the outputs from a deterministic modelling exercise with the views of experts in local conditions and environmental capacity issues. A key issue in this regard is in the setting of environmental quality standards. This is rarely an objective process and may require an assessment of trade-offs in terms of risks, costs and returns associated with different levels of precaution and corresponding environmental quality standards.

In determining the appropriate choice of model, the level of complexity will be of paramount importance. Simple models are robust and transparent but fail to take into account a wide array of location specific influencing factors. Complex models take account of these factors, but compound uncertainties through chains of relationships and assumptions, and are not easily understood or checked by stakeholders. Despite this there is a widespread and dangerous tendency to equate complexity and sophistication with accuracy and objectivity. By contrast conclusions drawn from simple models are more likely to be treated for what they are; that is, best estimates of likely impact. This encourages the output of simple models to be used as part of an array of information and expertise in setting appropriate standards and management interventions. It also encourages greater participation in the planning and management process.

However simple or complex the model, monitoring, validation, and model refinement will be required. This is often not undertaken. Indeed the existence of a model, especially a costly and complex one, may be used as an argument for dispensing with expensive monitoring. In practice therefore, and particularly in relation to aquaculture of the developing world, simple models have many strengths, especially when developed and used in combination with other interventions. These other interventions are vital if the output of any model is to be used effectively. They include: awareness raising of possible environmental limits to activity; identifying simple indicators for farm level and aquatic system environmental monitoring; encouraging better organisation and group responsibility for environmental management.

The choice of appropriate models will be informed by many considerations. The level of existing data and the quality of the historical data set are primary considerations. More data hungry models will require greater initial expense both in initial set up and on-going monitoring. This must be viewed in light of the fact that with improving technology and increasing data availability, low-tech systems are becoming higher tech and more and more big companies are using high tech solutions in developing countries. In situations with poor quality or limited quantities of data more robust techniques such as hierarchical or fuzzy logic may be required.

Once an appropriate level of data requirement has been determined the type of model must still be resolved. Different models treat bathymetry and physical conditions in different ways. Many are depth averaged and will not be appropriate for more complex coastal locations. Others rely on plume modelling and are inappropriate for diffuse effluents from cage culture. Decisions must also be made over what must be modelled in order to gain meaningful insight into the environmental capacity of the area. Is it sufficient to simply measure particulate organic benthic loading as being indicative of other impacts or must full trans-boundary reactive pathways of dissolved nutrients and chemicals be modelled?

Many models fail to take consideration of unpredictable or even seasonal influences on water exchange and current flow such as storms or monsoons. This highlights the fact that many models are geared toward temperate application in particular in semi-closed fjordic systems. Some work in warmer climates has already highlighted the importance of evaporation, especially in pond systems, which often goes unconsidered in many temperate models and anecdotal evidence from the Shetland Isles reports that models have over-estimated the impact of cage culture by failing to consider wind induced currents.

Perhaps the best solution to the problem of selecting models and implementing appropriate associated planning structures for environmental capacity assessment will be to have clear *guidelines on key aspects* that should be considered in key areas. In effect, a decision tree could be developed to lead potential users to generic model types or suites of models that are most appropriate for use in each situation. This is likely to be a vital exercise as an ever more baffling array of environmental capacity models become available on the market, or in freely downloadable form from the Internet.

Inappropriate model selection and a failure to consider the necessary associated management and planning infrastructure could lead many coastal aquaculture regions down an inevitable road toward environmental degradation and collapse of the industry. In order for this to be avoided and for models to serve their rightful role as valuable additions to the planning process, model selection and the development of improved environmental planning and management systems must take place in parallel. Models must feed into and enhance effective planning and management regimes.

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TROPECA

Environmental Indicators for Sustainable Aquaculture Development.

Tropeca Working Paper No. 2

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Abstract

The impacts of aquaculture are well publicised and often hotly debated. In developing countries where aquaculture has enjoyed unprecedented expansion over recent decades environmental impacts have become observable and in some cases detrimental to both the production system and the local ecosystem health.

Environmental capacity measures the ability of the environmental to assimilate waste without significant detrimental impact. Models are important tools to determine environmental capacity in advance, allowing for rational and equitable allocation of this capacity, and limits to be set on activities to ensure that environmental capacity is not exceeded (see working paper No. 1). In practice however modelling is imperfect, and indicators are essential to provide warning signals that capacity is being stretched, and facilitate effective adaptive management of aquatic systems. This should enable the level of economic activity to be limited to within a precautionary margin of the environmental capacity.

An ideal indicator should represent in a proportionate way all the components of or potential impacts from an effluent. While it is relatively straightforward to define an indicator of significant impact, it is more difficult to find indicators of more subtle environmental change which can serve as early warning and be used be used for pre-emptive action.

This paper looks in detail at the role and range of potential indicators and addresses the questions that influence the selection of suitable indicators for each situation.

Contents

1 Impacts of aquaculture

Planning from an Environmental Perspective

Aquaculture development over recent years has sometimes led to, or threatened to lead to a decrease in nearby water and ecosystem quality levels. Uneaten food, faeces, pseudo-faeces and dissolved metabolites threaten nutrient enrichment in the sediment and water column and the release of therapeutic chemicals may lead to chemical pollution, that has led to habitat changes and impacts on wild fish and shellfish. These impacts are often subtle and cumulative and may be insignificant in relation to a single farm or aquaculture system, but potentially highly significant for a large number of farms producing over a long period of time. These impacts are also often poorly understood, characterised by a high level of uncertainty and ignorance.

It is now generally agreed that aquaculture development needs to be better planned and managed if it is to achieve its undoubted potential and develop in a sustainable manner. GESAMP (2001) has proposed a set of guiding principles to inform future planning and management of aquaculture development in order to promote sustainable development. These include a call for increased integration with other regional activities, increased awareness of the costs and benefits (financial, economic, social, environmental) of aquaculture relative to other resource uses, assessment of environmental capacity and increased effort in monitoring.

There needs to be increased effort in quantifying the impacts of aquaculture so that this can be communicated to aquaculturists, other users, and regulators. Increasing the level of understanding of these impacts and the potential of the environment to assimilate production by-products is a pressing need for the industry.

Environmental Capacity Considerations

Environmental capacity (sometimes referred to as assimilative capacity) describes "a property of the environment, defined as its ability to accommodate an activity or rate of activity without unacceptable impact" (GESAMP 1986). This is an inherent property of the environment to assimilate or process waste generated by natural or human activities. Implicit in the concept is the assumption that the ability to assimilate waste without unacceptable consequences is finite, and can therefore be quantified and apportioned as an exploitable resource of the environment (Pravdic 1985).

Environmental capacity is likely to vary greatly according to local ecological, physical and hydrodynamic conditions. Well-circulated regions, subject to regular water-mass replacement will have far greater environmental capacity than adjacent areas exposed to poor levels of circulation (Ward 1999).

Traditionally industry has made use of marine environmental capacity by discharging effluents directly into the sea. Consent to discharge has often been given closely tied in with careful monitoring which ensures that environmental capacity is not exceeded. The difficulty with this approach is that monitoring will often show an impact only once the finite environmental capacity has been exceeded. This means that efforts to reduce discharges of potentially damaging effluents may only be implemented once environmental damage has occurred. Industry can find themselves in a poorly informed and ultimately expensive cycle of cheap waste disposal followed by expensive remedial action and in some cases compensation in line with the “polluter pays” principle.

More precautionary policies and management regimes are now being advocated to address these criticisms (Stebbing 1992; Gray 1998). These will require that action be taken to limit pollution well before environmental capacity is reached. Such an approach requires either much more accurate modelling of environmental capacity and effective planning and regulation to maintain activity well within limits, and/or identification of suitable precautionary indicators which signal problems well in advance, and allow for efficient and timely response.

Monitoring, modelling and the use of indicators

Another paper by the present authors discusses the role of mathematical modelling in establishing in advance the environmental capacity of a defined area or system (Southall, Telfer & Hambrey 2002). Available models vary greatly in complexity and in the levels of assumptions upon which they are based. Most of the models contain detailed hydrodynamic components and seek to model the impacts of one particular component of the fish farm effluent, such as the rate of sedimentary deposition and the impact that this has on benthic communities.

These models must be based around actual environmental information and must also be verified through subsequent monitoring to ensure that initial model predictions are correct. A good model should enable iterative refinements as monitoring data becomes available (Southall, Telfer & Hambrey 2002).

Indicators are essential to facilitate the development and refinement of these models and to serve as key components in any monitoring system. Ideally a range of indicators should be available which indicate the extent to which environmental capacity is being taken up; the suitability of the water for aquaculture production itself; and the degree of impact of aquaculture activity on the wider environment. Indicators should be practical, easily understood and easily recorded. The wide variety of indicators which may be used to determine when environmental capacity of an aquatic system is being approached or exceeded, are discussed in this paper.

2 Indicator Options

What Are Indicators of Environmental Change?

Physical

Physical indicators of environmental state and change are often easiest to measure and can provide clear indication of environmental condition. When plotted over time, they can provide a valuable indication of rate and direction of environmental change.

Oxygen is routinely monitored as an indicator of environmental health and, when plotted as a time series, provides useful indication of the trophic state of the environment. Oxygen levels can provide an indication of the organic loading in a system which may be particularly appropriate given that much concern relates to aquaculture's organic discharges. The fact that oxygen levels have a direct impact on the health of the cultured fish species means it is particularly sensible to monitor oxygen as an early warning to environmental capacity.

There are some difficulties in monitoring oxygen levels however, which means it is not always an ideal indicator. Oxygen levels vary greatly according to the time and location of the testing. Temperature, salinity, and time of day are the most predictable causes of oxygen variation, but the amount of circulation or even the amount of fish activity will also impact on levels. For example oxygen readings taken within a culture cage during the first meal on a calm day will differ greatly from oxygen levels taken outside the cage, in between meals on a windy day. In fertile pond and canal systems, especially in tropical countries, oxygen level varies substantially throughout the day/night, peaking around mid-day, and reaching minimal levels at dawn. Monitoring the levels of oxygen within low energy environments and static waters, such as lakes and reservoirs may be useful early indicator of environmental change especially in temperate regions.

It could also be argued that oxygen levels are driven by other factors of change and that looking at the causes of oxygen depletion would provide more useful early indication of an environmental threat. Identifying indicators as far as possible up the cause and effect chain of environmental impact is desirable.

The level of suspended solids or the rates of sedimentation will provide a direct estimation of the amount of sedimentary matter which is passing into the system. This provides a clear indication of the level of environmental change or degradation that is likely to result. This may also correlate directly with the activities of the fish farm and other anthropogenic activities and is an important factor in determining the overall impact of activities on the aquatic environment. Sedimentation is easy to measure and verify, providing incontrovertible evidence of impact. However, as with the use of dissolved oxygen, the amount of suspended material within the system is variable depending on the time of the year, or the prevailing climatic conditions and season. For example, higher levels of suspended material will occur during the winter months, in the northern hemisphere when wind driven turbulent mixing will re-suspend sedimentary fine particulate materials. Effective use of this factor as an indicator of environmental change or impact must take into account prevailing environmental conditions.

Nutrients and chemicals

Nutrients are often released in the form of wastes from aquaculture systems along with those from other sources. These can take soluble or particulate form and have various environmental fates. Soluble nutrients tend to remain in the water column and undergo dispersion and effective dilution in dynamic water bodies, and accumulation or build -up in low energy systems. Build-up of dissolved nutrients can lead to degraded conditions and changes in trophic conditions within the environment, thus changes in levels may be an early warning of hyper- or eutrophication. The nutrients that are most important in controlling the change in trophic levels are the so called “limiting nutrients”. Though this may change with individual water body, a general rule is that phosphorus is most limiting in freshwaters and nitrogen in marine conditions. Considerable amounts of both of these nutrients are released in aquaculture effluents and thus affect the trophic balance of the systems.

Nitrogen enrichment can lead to substantial increases in primary productivity in marine systems which can lead to a chain of environmentally degrading events. For this reason direct measurement of nitrogen over time is a strong indicator of environmental loading and may forewarn of a eutrophic episode in marine waters (Silvert & Sowles 1996). Likewise in freshwaters, phosphorus measurement over time will give forewarning of change in trophic status and thus approaching carrying capacity.

There are more sophisticated nutrient measuring techniques which may also be used as indicators. One simple technique is to place nitrogen starved algal tissue in mesh bags, at strategic points in various locations away from any suspected nutrient source. After a fixed period of time in the location the nitrogen uptake can be calculated from the percentage change in tissue nitrogen content. The difficulty with this technique is that the algal tissue used must be carefully prepared and measured in advance and the species used must be appropriate to the location and season of the test (Fong *et al* 1998). The advantage is that the indicator measures cumulative nutrient loadings and overcomes the problems of sampling in a dynamic and variable environment.

Many potentially toxic chemicals are routinely used in aquaculture production, particularly in more intensive culture systems. Some of these chemicals are persistent and can be of harm to local or even remote ecosystems when released from the culture location. The amount of residue required to have a biological impact can be very small. For this reason testing for chemical residues in the vicinity of the fish farm, and/or in the flesh of cultured organisms, is regarded as a desirable step toward limiting the environmental impact of aquaculture and safeguarding human health. However, these levels may be lower than the detection limits of the analytical techniques. It may therefore be desirable to investigate environmental concentrations of these chemicals using other methods. For example, it may be possible to carry out bioassay tests to determine the level of bio-accumulation in sedentary species which inhabit the area. Accumulation may also take place in sediments. Testing for residues in both sediments and biological tissue has been successfully carried out for organophosphates and organochlorines. In some cases the ratio of isotopes present may even provide indication of not only the level but also the timing of chemical discharges into the environment. This is potentially a very powerful tool to indicate the levels of chemical residues as a result of specific activity (Zitco 2001, Fillmann *et al* 2002).

Chemical testing of algae can also indicate the levels of heavy metals that are being discharged into the environment. Heavy metals perhaps do not necessarily have direct or proportionate links to organic enrichment and ecosystem impact, but they can be linked to a single effluent source and so act as an indicator of toxic discharge levels. The capacity of algae to bioaccumulate heavy metals can be used to enable chemical environmental monitoring. One difficulty of this technique is that the relationship between environmental levels and the rate of take up is not necessarily linear and the algae may reach saturation point even as environmental loadings continue to increase. The rate of take up may also be influenced by other factors such as salinity and pH (MacFarlane 2002). This combination of factors mean that testing for biochemical markers in the leaf or tissue of algae is not an ideal indicator of environmental pressure and is of little pre-emptive value in determining environmental capacity of a region.

The nutrient balance is another way of measuring the impact of aquaculture and is particularly appropriate as the levels of nutrient inputs into a system directly relate to impact. The amount of nutrients removed from the system at harvest time is subtracted from the total nutrients added to the system in the form of feed or fertilizers. This provides an indication of the efficacy of the production system and the likely levels of nutrient loading. Plotted over time this can provide valuable insight into the levels of environmental pressure and the degree to which steps to reduce these pressures are succeeding. These nutrient balance calculations are perhaps most useful for nitrogen and phosphorous as these are two of the most significant causes of hypernutrification and potential cause for eutrophication. Many countries including Finland (Finnish Ministry of Environment 2001) have already adopted nutrient balance as one of a suite of indicators of the countries sustainable development.

Testing for chemicals can be expensive, requiring technical skill, strict procedural protocols and detailed knowledge of the production system and its likely effluents. Appropriate chemical monitoring and interpretation is also dependent upon a clear understanding of the reactions that occur within the ecosystem as a result of the chemical discharge. Often these reactions are highly complex and impact many of the local environmental systems. For this reason chemical indicators are not ideal as indicators of environmental capacity for extensive rural aquaculture production, although in the future it is likely that further developments will add to the feasibility and practicality of chemical testing (Zitko 2001).

Biological

It may be considered that with the many possible physical and chemical indicators there is little need for further indicators for the biological constituents of the environment. Certainly carefully chosen and well monitored physical and chemical indicators will describe the change in the environmental loadings of the key agents of detrimental environmental change. Biological indicators are still required however, in order to provide an indication of the impact these changes on the ecosystem. They also provide an indication of longer term and cumulative impacts. For example, an oligotrophic system may be able to cope with a significant level of nutrient inputs before an impact is felt or conversely, in eutrophic systems low levels of nutrients may be found in the water as much is often locked up in the sediments (Cognetti 2001). Biological indicators are therefore vital aspects of any suite of indicators as these are the only way to obtain clear information on the reaction of the ecosystem to exterior pressures.

Sub-lethal responses in plants may be used to record changes in the levels of nutrients present in the ecosystem, for example their growth rate will directly correlate to the amount of nutrients that are present in the system. In addition, their high tolerance to many forms of pollution and sessile nature make them ideal indicators of effect in a particular location over a range of environmental conditions. However, care must be taken in interpretation of any sub-lethal response in that it too may vary with natural environmental conditions, such as season and climate.

Increase in phytoplankton growth is a simple indication of early signs of nutrient enrichment and eutrophication. In particular, the level of chlorphyll in the system is used as a direct measure of phytoplankton biomass and thus productivity, which in turn is dependent in the nutrient loading of the system. The indicator is effective in that it is easy to measure accurately and requires no sophisticated analytical equipment. This is perhaps preferable as an indicator to simple nutrient levels as this provides greater clue to the level of impact in terms of phytoplankton growth as a response to eutrophication (Gowen 1994).

Plants are poor indicators of toxins. By contrast many animals are useful as indicators of both toxicity and organic loading. There are two principle measures of impact, either a change in community structure or a presence of toxins in the flesh of animals, particularly sedentary animals, which remain within the region of impact.

Biological indicators may provide indication of impact due to several factors acting in combination, which may otherwise go undetected by simple physical or chemical monitoring. Changes in biological community composition can also provide an early sign of both adverse change and positive indication of progress following restoration processes (Borja *et al* 2000).

The behaviour or stress signals of biological components of the ecosystem can provide valuable indication of the current level of environmental stress or loading. If sensitive enough this may even provide an early warning of harmful increases in toxins or nutrients. The culture species themselves may act as an indicator of environmental stress, although one study determined that the number of moribund salmon at the cage surface bore little correlation the level of disease lower in the water column (Stephen & Ribble 1994).

Most behavioural indicators, although pre-emptive, leave little or no time for any changes in practise to take effect before more serious environmental impact occurs. Using behavioral testing may still be a useful indicator of approaching environmental capacity but there should be a greater degree of sensitivity. At times of environmental pollution mussels exhibit a “stress syndrome” which causes a physiological change measurable along “stress indices”. In the past this technique has been shown to be an effective indicator of both coastal contamination and the biological effects of pollutants (Viarengo & Canesi 1991).

It may be argued with some legitimacy that at the point of observable biological impact environmental capacity has already been exceeded. This same line of argument would conclude that relying on biological monitoring inevitably leads to delayed response to negative impacts. This has been the traditional failing of the use of environmental or assimilative capacity in environmental policy and has been the principle driver of a shift toward more precautionary environmental policies (Stebbing 1992). Therefore in order for biological indicators to be of any use at all in new policy frameworks built around the management of environmental capacity, the indicators must be sensitive and provide as much early warning of negative impacts as is possible.

The composition of the ecosystem can provide a clear indication of the health or environmental stress of the local marine community. The natural variations in species tolerances to environmental impact mean that even small changes in community composition can be indicative of early signs of environmental stress. Species can be classified according to their levels of tolerance or opportunism. Perhaps the best known and most widely used system of classification for this is the r-selected, k-selected and T groupings (Pearson & Rosenburg 1978). The k-selected species have low levels of tolerance and long lifespans, making up the climax community. Their presence indicates a healthy ecosystem characterised by high levels of species diversity, richness and abundance. The r-selected species have shorter life spans, fast growth and early sexual maturation and their presence indicates an impacted community characterised by low species diversity, richness and abundance. The presence of only stress tolerant T species is an indication of heavily impacted ecosystems.

This system enables the biological community structure to provide indication of the level of environmental stress at any point along the range from stable background community through until a heavily impacted azoic community. The combination of monitoring the species composition and statistical analysis of species richness, diversity and abundance, carried out over a time series can serve as a powerful indicator of ecosystem health (Borja *et al* 2000). There is also evidence to suggest that changes in the benthic bacterial composition caused by sedimentation may occur at relatively early stages of environmental impact, even before physical and chemical changes can been detected. This has been observed in sediments below mussel farms and provides potential early indication of impact through biological means. The use of aerobic or heterotrophic bacterial communities gives early indication of changes within the sediments, but is less use in the water column where physical characteristics have greater influence (La Rosa *et al* 2001).

3 Selection of Appropriate Indicators

The difficulty in selecting a suitable indicator for environmental capacity assessment is that it must do more than simply indicate the state of the environment for a single component or group. Ideally it should give some indication of the capacity of the environment to assimilate waste. More particularly it should indicate when capacity is being stretched but not yet exceeded, so that mitigating action can be taken prior to any negative impacts.

The presence of a large number of indicators at levels high enough to be reliably detected often indicates that environmental capacity has already been exceeded, and only remedial, rather than pre-emptive action can be taken. This failing, more than anything else, has led the drive toward increased precaution in development planning strategies. This in turn requires the identification of "pre-emptive indicators" which give appropriate early warning of possible future problems.

A single pre-emptive indicator may prove elusive, particularly within the confines of what is practically and economically feasible in many developing countries. A reliance on expensive and complicated monitoring equipment to detect minute traces of an appropriate indicator will mean that widespread use will be difficult for many countries. It may therefore be necessary to cast the net more widely in the hunt for a suitable indicator.

A wider range of less sensitive, but none the less reliable indicators, may, in combination, provide as reliable and as pre-emptive a warning of environmental capacity as a single highly sensitive measurement. In addition a wide range of coarser indicators designed to act in combination may enable easier and less expensive monitoring to be done as part of a clear and widely adopted protocol stipulated at the start of a project's life.

Such a suite of indicators, used in combination with modelling of nutrient loadings and environmental capacity, should provide both farmer and planner/manager the combination of tools necessary for effective environmental planning and management system based on both prediction and monitoring.

Constraints to use.

Economic

Many of the areas that have already exceeded, or are close to environmental capacity due to poorly managed aquaculture development are areas with poor local economies. It is for such areas that there is the greatest need for clear indicators of environmental pressure. Much of the production systems in these areas are low input extensive systems requiring minimal capital expenditure. As the majority of production is for local markets and consumption there are unlikely to be the scales of revenue generated in order to fund the monitoring of key indicators of environmental change.

Many widely used techniques such chemical and geochemical analysis, toxicity testing, ecological surveys, biomonitoring or even histopathological testing are expensive and require technologically advanced equipment and highly trained personnel. These tests may be suitable to the regulatory strict and economically strong areas of global production but are unlikely to be appropriate or affordable in many developing country situations (Wells *et al* 2001).

It is imperative that any indicators or suites of indicators of environmental quality that are proposed will mean that routine monitoring is locally economically viable.

Monitoring will typically also generate huge volumes of data. In some instances sampling less often or in fewer locations where possible may reduce this burden. Another solution is for the data to be submitted to principle component analysis (PCA) to determine which of the measured parameters are present in most significant quantities and most closely correlate with the observed environmental impacts. Once determined, these principle components may be combined into indices that enable complex technical data to be aggregated into more readily understandable and manageable forms. This may particularly useful for informing managers and decision or policy makers, who may not possess the detailed technical knowledge for interpreting the monitoring results (Shin & Lam 2001).

Comparability

The selection of an appropriate indicator must be determined by the nature of the effluent and the nature and state of the local environment. In addition, it is clear from the constraints to use that local variations in technical ability, finances and manpower will also influence the indicator selection. It is therefore inevitable that indicators which are suitable for one location may not be ideal for another. Indeed it should be encouraged for indicator selection to be influenced by local considerations as opposed to following the example of other areas.

This locally specific nature of indicators is both a strength and a weakness. The great drawback is the loss of comparability that means that effluent consent levels in one location are based upon different calculations from elsewhere. This may also increase the level of sampling, modelling and monitoring work that may have to be carried out by any regulatory body. This workload factor may also be taken into consideration at the time of indicator selection and again a trade off solution may be required. In this case it is preferable for there to be a slight loss of locally specific action, rather than setting unrealistic demands which lead to a failure to visit certain locations.

Simplified Suite Solution

The above constraints highlight the need for a simplified yet effective set of environmental indicators, that allow clear objective environmental assessment, enabling pre-emptive indication of approaching environmental capacity limits. The indicators must be both pragmatic and cheap yet still be sufficiently robust to provide applicable and comparable environmental information. Success in the search for and application of suitable indicators is likely to be one of the key factors in determining the long term health and sustainability of use of aquatic resources, in particular in less developed coastal countries. The continued deterioration of coastal margins of many of these countries, in particular those with established or growing aquaculture industries, is a clear indication of a current failure to implement practical procedures for environmental monitoring and response. (Wells *et al* 2001).

4 Conclusion

Management Considerations

Inevitable subjectivity

The use of indicators is an attempt to provide an objective picture of environmental health and quality. To a large extent this goal can be achieved through the selection of a range of physical, chemical and biological indicators which, in combination, can clearly identify the patterns of environmental pressure at ecosystem level. However some subjectivity inevitably influences this choice.

The constraints of use of each indicator, the preferences or experience of the research team and the choice of principle focus of the study are all non-objective factors which may influence the selection of indicator and interpretation of indicator data. For example, if the principle concern for of the research team is the health of local fish populations emphasis may be given to indicators such as dissolved oxygen, whereas if the principle concern is to protect the local tourism value suspended solids may be given increased consideration.

A team from Edith Cowan University, Australia (Wood and Larvey 2000), offer a good example of inevitable subjectivity of experts in indicator selection. This study asked experts to determine the health of ecosystems and identify the key indicators upon which the conclusion was based. The study concluded the health was " a respondent-dependent concept" and the indicator selection was often influenced by personal perspective or scientific preconditioning.

A suite of indicators should therefore address the widest possible range of impacts of relevance to sustained resource use and biodiversity conservation.

Need for protocol

Aquaculture monitoring of indicators of environmental change is often poorly carried out and there is evidence that standard scientific procedural requirements are regularly ignored (Zitko 2001). This is particularly the case for the monitoring for traces of chemicals used in the production process. This can be partially explained by the wide variety and rapid development of chemicals in use and the inevitable habit of producers to change treatments according to changes in husbandry needs or other external factors. It is imperative therefore that those carrying out monitoring for selected indicators are fully aware of the up to date chemicals and the critical procedural aspects such as sampling, duplication, storing and processing (Zitco 2001).

It should be considered that the number and treatment of samples must satisfy the statistical requirements of the study. This is particularly the case when monitoring fish farm sediments due the highly heterogenous nature of the deposition which impacts upon the local sediments. Careful selection of indicators and subsequent collection and testing of samples may be rendered worthless through inadequate storage and treatment of samples. For example, thawing of samples, opening of containers, sub-sampling and re-freezing may all effect the eventual conclusion of the monitoring. Without a full appreciation of the analytical requirements and equipment which is appropriate for testing contemporary farm practices there is unlikely to be adequate response time and environmental capacity may continue to be exceeded on a wide scale (Zitco 2001).

Once the appropriate indicator or suite of indicators and suitable monitoring regime has been decided there remain aspects which will determine the value of the results. The vast majority of data relating to environmental capacity indicators will be of little use unless is comes as part of a trend series.

Ideally this trend series should begin with background data of the site in an unimpacted state, although this will frequently not be possible in areas of high levels of aquaculture development.

Appropriate level of complexity

It his highly inappropriate to advocate the use of complicated and sophisticated suites of indicators for environmental capacity determination if this is beyond the financial means or technical capabilities of the responsible body. The level of detail required should therefore be commensurate with the means, capabilities and understanding of any local bodies. The level of complexity could also aim to be commensurate with the level of risk and the potential threat of exceeding of environmental capacity.

In well-flushed regions with very small levels of aquaculture or other anthropogenic activity it may be unjustified to require frequent employment of highly sensitive techniques which repeatedly reveal 'no impact'. By contrast in areas with high levels of organic and chemical discharges the level and urgency of monitoring should reflect the increased threat of exceeding environmental capacity.

It is apparent therefore that a single indicator or single predetermined suite of indicators may be impossible to apply universally. The choice of indicator should first and foremost be informed by the specific local or regional requirements. It should be remembered that it is better to employ a simple technique which will be reliably and routinely put into use than a sophisticated technique which is infrequently and poorly carried out. If monthly secchi disc readings are all that can be reliably carried out then this should be encouraged, perhaps introducing further indicators once the routine of monitoring is established and the importance of monitoring recognised.

Decision Matrix.

The selection of environmental indicators or suites of indicators for an aquaculture production region is complicated and many parameters must be considered. It is likely that different indicators should be chosen for different situations. In situations where there are no financial constraints or technical limitations sophisticated indicators will be proposed which may be considered inappropriate in other developing country situations.

The indicator should also reflect the nature of the production systems in the region. Insisting upon detailed chemical testing is pointless in regions where the production systems are extensive. It is appropriate to choose an indicator which reflects the nature of the principle environmental threats posed by the production system. In areas of intensive culture which places high organic loads into the water, such as some forms of shrimp pond production, the indicators should be geared toward early indication of nutrient enrichment or oxygen depletion.

It may be possible to draw up a decision matrix to assist in the selection of appropriate indicators. For example certain indicators may immediately be ruled out if there are serious financial constraints. The list of potential indicators may be whittled down further if there is little or no chemical use, and again still further if it is an extremely energetic well flushed environment.

A decision matrix could be a universally adopted tool which ensures that environmental capacity modelling and monitoring is given serious regulatory strength, whilst at the same time allowing for locally specific solutions which are practical, affordable and appropriate to the local ecosystem and production system.

Only in this way can environmental capacity be given a strong and useful position in the regulatory tools of aquaculture producing nations and regions. Where employed to full effect this will allow a pre-emptive calculation of environmental capacity to be set, enabling exploitation levels to be limited within a precautionary margin of this limit. The indicators will enable routine tests to be carried out that may serve to inform of time series changes in environmental condition and aquaculture impacts.

The emphasis must be on small steps in the right direction. It is inappropriate to demand great leaps forward in monitoring, modelling and regulatory time, expertise and investment. A suitable decision matrix may provide the incentive for regions to employ some efforts to increase awareness of the environmental capacity and the impact that activities can have on the environment. This may ultimately lead to great improvements in the levels of aquaculture environmental impact and protect the coastal resource for other local users.

Appendices

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TROPECA

Environmental capacity: its application for the environmental management of aquaculture in tropical developing countries

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Contents

1. Introduction

1.1 The need for environmental management of aquaculture

The aquaculture industry is increasingly important to the economies of many developing countries and an important contributor to dietary protein needs. Aquaculture exemplifies the intrinsic link between the environment and the economy: an economic activity that directly depends on a high quality natural environment for success; yet which can itself negatively impact that environment. Aquaculture development, and especially brackishwater production of shrimp, has developed rapidly but erratically in recent years. In many locations boom has been followed by bust as over-rapid and concentrated development has resulted in habitat destruction, poor water quality and increased disease incidence. These issues have been widely documented and reviewed (see, for example,

Rapid development has both positive and negative social impacts, which vary according to local circumstance. The economic success of aquaculture may cause changes in ownership and access patterns; and aquaculture may negatively impact the well being of other resource users.

There is widespread recognition that aquaculture development must therefore be managed, and its negative impacts limited in some way. But there is little agreement as to the basis or mechanism for such limitation. On the other hand there are powerful arguments to promote and facilitate an activity with the potential to substantially benefit the livelihoods of poor people.

A precautionary approach applied too fastidiously could unnecessarily deprive poor people of the opportunity to use the environmental goods and services available to them. An approach is required that is as objective as possible, which does not unnecessarily hinder development, but which takes full account of the productive capacity of the environment and the needs and values of all stakeholders in both short and long term. The concept of environmental capacity (EC) is fundamental to such an approach.

1.2 Applying the concept of environmental capacity to aquaculture management

The concept of environmental capacity and carrying capacity lies at the heart of most definitions of sustainable development. It recognises that there are limits to ecosystem productivity, or the capacity of the environment to assimilate wastes. The environment can only support a certain population, or level of activity, beyond which its functions and services may be impaired.

Article 9.1.2 of the FAO Code of Conduct for Responsible Fisheries requires that "States should promote the responsible development and management of aquaculture, including an advance evaluation of the effects of aquaculture development on genetic diversity and ecosystem integrity, based on the best available scientific information". Such an evaluation has little meaning unless the effects can be measured, acceptable limits to change agreed, and management strategies to prevent breach of these limits put in place.

Environmental capacity is defined by GESAMP[1] as " a property of the environment and its ability to accommodate a particular activity or rate of an activity…without unacceptable impact". The concept is therefore central to the promotion of sustainable aquaculture development, and the implementation of the FAO Code. Importantly, it requires us to address the cumulative impacts of the whole sector (and in its most comprehensive form all economic activity) on the ecosystem within a specified area. To date these wider cumulative and dispersed impacts have not been effectively addressed by conventional regulatory approaches such as EIA or individual farm consents (Hambrey 2000).

The application of the concept of environmental capacity to the planning and management of aquaculture development relies on two aspects:

1. The ability to measure the rate of environmental change and relate this to activities such as aquaculture
2. The determination of what amount of environmental change is acceptable, i.e. developing an environmental quality standard (EQS)

A recent GESAMP report[2] suggests environmental capacity in relation to aquaculture may be interpreted as:

- The rate at which nutrients are added without triggering eutrophication; or
- The rate of organic flux to the benthos without major disruption to natural benthic processes

More practical interpretations are of particular relevance to the situation in developing countries. These might include for example:

- The rate of organic (or nutrient) flux into a defined aquatic system without reducing aquaculture productivity
- The rate or organic (or nutrient) flux into a defined aquatic system without negatively affecting the interests of other resource users

The concept of environmental capacity can, however, be expanded to include impacts that are more difficult to quantify such as reductions of natural habitat or even loss of scenic value due to visual impacts.

Environmental capacity addresses changes to the environment rather than focusing on production, and relates to all possible causes of change. It requires an understanding of the processes involved in an ecosystem and the monitoring of changes to that system. The determination of *carrying capacity* on the other hand requires that we understand the contribution of a particular activity or set of activities to environmental change – for example the rate of production of nutrients per unit production of aquaculture. The two are intimately related. Understanding environmental capacity is a pre-condition for the estimation of carrying capacity of a particular activity.

Although environmental capacity and carrying capacity are scientific concepts, they incorporate a strongly subjective dimension. The definition of 'acceptable change to the environment' and the definition of environmental quality standards, although informed by science, must rest on subjective judgement.

Various criteria may be applied to determine *what is acceptable*. In order to implement our commitment to maintain ecosystem integrity we need sub-objectives and/or indicators of ecosystem integrity. These sub-objectives might include, for example, water quality (as required for a range of economic and leisure activities including aquaculture); natural productivity; biodiversity; assimilative capacity. Since these objectives may be competing rather than mutually reinforcing (e.g. maximising productivity may not equate with maximising biodiversity or maintaining water quality for a particular activity) we need a decision making process which deals with multiple objectives bounded by absolute limits (beyond which ecosystems and most of their productive and or assimilative functions break down). While it is for scientists to inform on the rate processes, dynamics, indicators, and absolute limits; it is for the various interest groups or their representatives to make the trade-offs between objectives within these acceptable boundaries, and to set socially and economically appropriate targets for environmental indicators. These targets will ultimately be informed through an understanding of the nature and consequences of environmental change on the one hand, and the nature and distribution of resulting costs and benefits between interest groups over time.

Although this process must ultimately be subjective, there are some formal concepts for resolving the conflicts that may arise in any attempt to define limits to what is acceptable. These are discussed further by Silvert[3] who errs towards 'fuzzy logic' as an approach that may be suitable for aquaculture. An example is the consideration of the extent of benthic impact: zero impact is assumed to be 100% acceptable while complete anoxia of the seabed within a specified area may be 0% acceptable. There exists a spectrum of levels of acceptability between these, likely to be dependant upon the stakeholder (farmer, fisherman, environmental NGO, member of local community). Quantifying levels of acceptability, although still subjective, facilitates the negotiation necessary to reach a consensus on what is or is not acceptable change.

A range of tools are now available, ranging from highly formal to highly informal and flexible, to facilitate decision making in the face of uncertainty and multiple objectives (refs****). While these are used increasingly in planning, and especially in respect of major controversial development projects, they have so far been applied very little to address the more subtle but equally important issues associated smaller scale but cumulative developments.

In practice quality standards have been traditionally set by scientists: internationally (European Directives), for example, List I and List II and Red List substances (Dangerous Substances Directives); or they may be locally derived from available data (e.g. sediment quality) to provide operational guidelines. Inherent within the EQO/EQS approach is the concept of a mixing zone or Allowable Zone of Effect (AZE) where it is accepted that standards may not apply or be less stringent. This approach requires a definition of the extent of this zone and the criteria to be achieved within it[4].

In the following sections we review the environmental problems which aquaculture development faces, and the approaches to improved environmental management which have been applied in different parts of the world, including developing countries. We explore the extent to which environmental capacity issues are currently addressed, or could be addressed within existing and possible future management frameworks. We also briefly describe current DFID funded research (*Tropeca*) in Bangladesh and Vietnam which explores the application of environmental capacity as the conceptual framework for improved environmental management of aquaculture in developing countries.

2. The problem

Environmental and social impacts of aquaculture development

The negative environmental impacts associated with aquaculture are well documented (refs *Maraqua reviews, GESAMP, 1991*[5] ****, WWF scottish stuff etc, Blacks review). Box 2.1 lists the variety of potential impacts that aquaculture can have on the environment.

Box 2.1 Potential Environmental Impacts of Aquaculture

> Enrichment (from food, faecal or pseudo-faecal wastes)
> Interaction with the food web (i.e. removal of phytoplankton)
> Oxygen consumption
- Disturbance of wildlife and habitat destruction
- Physical changes to land and water resources
> Interaction between escaped farmed stock and wild species
> Introductions and transfers
> Bioactive compounds (including pesticides and antibiotics)
  - Longevity of inhibitory compounds in animal tissues
  - Discharge of inhibitory compounds in the aquatic environment
  - Development of antibiotic resistant microbial communities
> Chemicals introduced via construction materials
> Hormones and growth promoters

In tropical developing countries the economic benefits of aquaculture development have in the past generally over-ridden environmental concerns. It is only now that the cumulative impacts of many developments (both small and large scale) have become evident that governments, and farmers themselves, are becoming concerned. Below we briefly review the main categories of environmental impact listed in box 2.1

Nutrient enrichment and oxygen depletion

Intensive finfish and crustacean aquaculture generates substantial amounts of waste in the form of soluble metabolic products (in particular ammonia), uneaten food and faeces. Typically, for every tonne of salmon produced, we also generate *kg of nitrogen, *kg phosphorus, and * x kg of suspended solids. Equivalent figures for shrimp aquaculture are ******. Although feeds and food conversion efficiency are constantly improving[6] nutrients from aquaculture now contribute substantially to total coastal nutrient loads in some countries and locations[7].

Nor is this problem associated only with intensive finfish. Although shellfish cultivation involves a net removal of nutrients from coastal habitats, at a local level they concentrate nutrients, in the form of faeces, pseudofaeces and dissolved metabolic products[8].

More extensive pond systems (for example of carps) on the other hand may be net nutrient sinks[9] although much depends on the time scale taken.

Interaction with the foodweb

The nutrients generated by aquaculture may lead to changes in plankton communities with indirect effects throughout the food web. Equally, increased organic matter loadings on sediments in some sheltered coastal habitats may lead to wider and less predictable changes in ecosystem composition. To date few countries have faced up to these wider issues – either in respect of aquaculture or other nutrient sources. It is arguable that the precautionary principle should be invoked in respect of these impacts: they are poorly understood and unpredictable, but potentially significant with possible repercussions on other sectors such as capture fisheries. How strongly the precautionary principle should be applied in these circumstances is however not clear underr any existing guidelines.

Disturbance of wildlife, habitat destruction, and loss of common property resources

Aquaculture, along with felling for charcoal and conversion for agriculture, has contributed to the destruction of large areas of mangrove, which not only provides a variety of environmental goods and services to local communities (firewood, fisheries) and beyond (protection against erosion and storm damage), but also provides important environmental services to the aquaculturists themselves (nutrient buffers, food supply). Globally the proportion of mangrove destruction attributable to aquaculture is not high – perhaps 5-10%[10] but locally it has been significant. For example, it is estimated that 99 percent of the Indus Delta mangrove has been deforested, a reduction of 34 percent has occurred in Indian mangrove areas and 60 percent of the Chakoria Sundarban mangroves have been lost to conversion to shrimp ponds[11]. Other coastal and estuarine habitats such as salt marsh and flats and lagoon systems have also been affected.

These coastal habitats have often served as common property resources in the past. The extreme poor are often land-less and more dependent upon these resources than farmers. They are therefore amongst the first to suffer as a result of environmental degradation[12].

Social conflict associated with aquaculture development has occurred in S and SE Asia and S and Central America. In India fishermen suffered reduced access to their fisheries as a result of broad swathes of aquaculture development (ref**). In Thailand there has been some conflict between shrimp farmers and rice farmers, as the former require (at least some) brackishwater, and the latter require fresh water (ref***). In Indonesia there have been significant conflicts between partner farmers, local agriculturalists and large companies engaged in aquaculture. Increasing social conflict has been observed in coastal areas of Brazil. In contrast to the initial objectives of establishing a shrimp culture industry as an option to poor rural villagers, entrepreneurs from outside the communities now own most of the Brazilian shrimp farms[13].

Comment: Need a good list of references here. India (stuff in Infofish); Phillipines – primavera; Thailand – FW v salt water; Indonesia; Malaysia etc

Impacts on soil and water resources and the spread of disease

In some areas pumping of groundwater and/or construction or modification of canal systems has resulted in saltwater intrusion into what were previously freshwater areas. Disturbance and release of acid from acid sulphate soils (common in mangrove areas) as a result of pond digging has also had negative impacts on resource productivity, including aquaculture yields. Excessive acidity is likely to reduce disease resistance.

Overall, the rapid and over-concentrated development of aquaculture, and particularly coastal aquaculture, has resulted in declines in soil and water quality, increased disease incidence, and more rapid disease spread. This has directly and negatively impacted fish farmers themselves, in both inland and coastal environments. The knock-on effects of disease outbreaks are substantial for local communities and national economies. Some recent examples of the economic impacts are listed below:

- The total negative economic impact of two shrimp viruses, white spot syndrome virus (WSSV) and yellow head disease (YHD), has averaged about $1 billion (U.S.) annually since 1994 in major shrimp-growing countries, including China, Thailand, India, Indonesia, Bangladesh, Malaysia, Taiwan, Vietnam, and Japan[14].
- In Asia, WSSV has caused severe economic losses to the shrimp industry; due to the effects of the virus, China reported losses of US$1 billion in 1993, while in 1995, Thailand lost over US$500 million. At the end of January 1999, WSSV was detected in Nicaragua, Guatemala and Honduras. Later, the virus was also reported in Panama, Ecuador and on Colombia's Pacific coast.
- Marine finfish disease losses in Japan in 1992 were reported at US$114.4 million[15] (Arthur and Ogawa 1996).

- Between 1995-1997, "red spot disease" of grass carp affected 4000 of 5000 cages in Northern Vietnam, with losses estimated at US$ 500 000[16] .

Financial losses are not restricted to the tropics or to developing countries as the bankruptcies in Scotland due to ISA outbreaks and Norwegian losses due to Hitra disease in farmed salmon testify. Australia benefits from low production levels in relation to coastal area, and low levels of water pollution generally, permit a high standard of aquaculture products, but there have already been significant production losses due to adverse environmental impacts on and from aquaculture[17].

Interaction between escaped farmed stock and wild species

There has been much concern and debate over the possible interaction between farmed salmonids and the capture fisheries for wild salmon. These interactions include possible competition for breeding and food resources, genetic exchange, and disease spread. The high incidence of sealice in farmed and wild salmon has become an issue of particular concern. In practice clear relationships and impacts are – like the food web effects noted above – extremely difficult to assign and quantify, but nonetheless are potentially highly significant, with obvious potential effects on other resource users. The relationships between farmed and wild shrimp (*Penaeus*) are also widely debated, but remain uncertain.

Introductions and transfers

The main reported impact of introduced aquaculture species has been in terms of disease introduction to stocks which may have limited resistance. The classic example of this was the introduction of Bonamia? To European shellfish stocks, following the introduction of the Pacific oyster. Direct competive impacts have occurred following the introduction of Tilapia to some parts of the world – in particular Australia. The impact, if any, of the recent introduction and widespread culture of Western White Shrimp *Penaeus vannemei* in East and SE Asia remains to be seen.

Chemicals

Chemicals are used widely in the aquaculture industry for both water treatment and disease prevention/treatment. In some countries the use of antibiotics has been particularly widespread and unregulated, with potentially serious impacts in terms of the development of resistant strains of bacteria – with potentially serious implications for both human and cultured organism health.

Other chemicals, and especially pesticides used to treat protozoan and crustacean parasites, can have significant impacts on the wider environment, including the larval stages of commercially important crustacean and shellfish.

The use of chemicals may incur penalties to the aquaculture industry itself including: (1) international trade difficulties arising from drug residue monitoring and enforcement programmes; (2) the potential for loss of efficacy of prophylactic antibacterial agents; and (3) increased demand for and complexity of effluent treatment (GESAMP, 1997[18].)

The cumulative nature of most environmental and social problems – implications for management

The short review presented above illustrates the cumulative nature of most environmental impacts from aquaculture – insignificant in relation to most small developments, but increasingly serious as the sector grows. At some point the capacity of the environment to cope with the pressure – from nutrients, from chemicals, from habitat destruction, from alien species etc - is exceeded, and water quality declines; disease becomes endemic; other interests are compromised. Environmental capacity has been reached or exceeded. Efforts to define and estimate environmental capacity, however difficult, are therefore fundamental for the environmental management of the sector. We need a more strategic approach based on the understanding that there should be overall limits to the pressures generated by the *sector as a whole* within a particular aquatic system or area. Traditional one off EIA in relation to a particular farm is totally inadequate to address cumulative issues.

Although the concept of environmental capacity is not strictly applicable to social problems, the management issues are the same: while a few small developments have insignificant social impact, a large number or large scale of development does. There comes a point at which cumulative development exceeds the level that is socially acceptable. We need to estimate or agree what that level is in advance, so that aquaculture can be managed in such a way as to avoid social conflict. As for the environmental impacts, the problems cannot be approached piecemeal, through farm level consents or EIA; rather a development strategy must be formulated, based on an estimation of acceptable levels of development or limits to change, and a management system must be put in place to meet the overall strategy.

2.2 Reactions to the problems

In practice, while there have been substantial government, agency and industry responses to the social and environmental issues associated with aquaculture development, few have effectively addressed the cumulative problems described above. Where some attempt has been made to address them, this has most often been in the form of an arbitrary ban or limit on activity in response to pressure from specific interest groups. Only in recent years is there some evidence of more strategic approach developing.

Environmental policy in industrialised nations increasingly reflects the public perception of the environmental impacts of aquaculture, which are often exaggerated relative to the impacts of other economic activities. It has been argued that this has led to simplistic or overly severe marine policy with reduced reliance on science[19].

Canada has an indefinite moratorium on licences for new salmon production operations. In New Zealand there remains a moratorium on site licences for new aquaculture production, while Aquaculture Management Areas (AMAs) are established by local councils. Environmental groups in many other countries are applying increasing pressure on their governments to do the same.

The governments of some developing countries have also introduced highly restrictive legislation in relation to further aquaculture development, although these restrictions are rarely fully enforced. In 1991 the Thai government banned the conversion of mangrove for shrimp farming, and five years later in 1996 it produced a 20 year sustainable development plan that included an environmental plan for shrimp farming. In the same year mangroves in Ecuador were designated protected woodland to prevent further loss of mangrove to shrimp farming.[20]

In India, following a cost-benefit analysis that concluded the costs of coastal shrimp farming and its impact on alternative coastal use exceeded benefits by 4 to 1, the Indian Supreme Court ruled that all large intensive shrimp farms within 500m of the high water line should be demolished by 1997[21].

The Vietnamese Fisheries Law (draft, 2002) states that it is prohibited to discharge untreated and disease-infected waters from an aquaculture site into an outlet channel, aquaculture ponds or into a natural watershed that do not meet the criteria as regulated. It also states that the Ministry of Fisheries will formulate a specialised technique for aquaculture sites, national hatchery centres and monitoring stations for environment prediction and fish quarantine stations[22].

Despite these attempts by governments to regulate aquaculture and its impact on the environment in recent years, the problems have persisted and in many cases worsened. Claridge (1996)[23] has identified a number of legal and institutional factors that may contribute to the negative impacts of shrimp farming, which are of relevance to other aquaculture systems in developing countries:

- Inappropriate, ambiguous or a lack of legislation
- Inappropriate or lack of environmental standards
- Lack of effective land use and resource allocation controls
- Conflict between law enforcement and other government functions
- Ineffective or non-existent law enforcement

Article 9 of the FAO Code of Conduct for Responsible Fisheries[24] requires member States to develop an appropriate legal and administrative framework to facilitate the development of responsible aquaculture. The focus is on encouraging states to help the farmers help themselves as it recognises that 'command and control' measures are rarely effective and 'soft laws' such as codes of practice developed primarily by the industry itself more likely to have a positive impact.

Developing legislation by itself is unlikely to solve the environmental, economic and social problems that occur when aquaculture exceeds environmental capacity. Where legislation has been developed it has generally failed to recognise the need to consider the cumulative inputs from aquaculture and the environmental capacity of water bodies to process those inputs. These problems associated with aquaculture regulation in developing countries suggest it will take some time before "state of the art " aquaculture laws will be drafted, but in any case the belief that legal prohibition of unacceptable behaviour will solve environmental concerns is almost certainly erroneous.[25]

International organisations and the producers themselves are also attempting to deal with the environmental problems facing aquaculture. In response to growing concerns over disease the Network of Aquaculture Centres in Asia (NACA) is attempting to increase awareness of disease outbreaks and encouraging the adoption of management practices that reduce the risk of disease. At the request of the Government of Indonesia, NACA has organized an Emergency Disease Control Task Force Team to accurately assess the disease situation and find measures to reduce the risks and further spread of the disease[26].

Actions to counter the risk of disease are often reactive rather than preventative. The FAO has set up Aquatic Animal Pathogen and Quarantine Information Service (AAPQIS) in order to track the development and spread of disease . As a result of devastating disease outbreaks, many South East Asian countries have invested in expensive quarantine and testing regimes. The Thai government and international donors are also funding research into closed system and recirculation technologies to avoid farm to farm transfer of viruses and chemicals.

Increasing numbers of producer groups are developing codes of conduct and best management practice. Farmers increasingly recognise that poor management in neighbouring farms increases the risk to their own crop, with the cumulative effect of all their farms degrading the receiving environment.

Farmer groups, often working with International organisations to provide technical/management guidance, have developed voluntary codes throughout Central and South America (Mexico, Honduras, Ecuador, Chile) to encourage better and consequently more environmentally friendly shrimp farming practices.

Codes of practice have also been developed in Australia (Australian National Aquaculture Council codes for prawn, tuna and silver perch farming), Thailand (government scheme adapted by local farmer groups) and at a local or regional level throughout Asia. Codes of Conduct have also recently been developed in Europe by the Federation of European Aquaculture Producers as an extension of the FAO Codes of Conduct for Responsible Fisheries.

3. Management tools and approaches

In this section we examine in more detail the strengths and weaknesses of different tools and approaches which can be used for the environmental management of aquaculture development.

The environmental impact of aquaculture can be mitigated by influencing different elements in the establishment and production cycle: siting and design; input quality and quantity; production processes; output quantity and quality. These are not mutually exclusive: controls on outputs may for example force changes in production processes. These different elements can in turn be influenced in different ways: through government regulation; through voluntary initiatives and industry self regulation (driven by communication, persuasion, and facilitation); and through the use of economic instruments. A comprehensive management strategy and plan may encompass any or all of these tools applied to the different production elements.

Table 3.1 Summary of management approaches according to activity and tool

APPLICATION OF TOOL	**Regulatory**	**Voluntary**	**Economic**
siting and design	Strategic planning leading to zoning and/or enforced site selection criteria; Standard planning procedures and consent conditions including design criteria; EIA and follow-up siting and design conditions.	Recommended site selection and design criteria in code of conduct; .	Tax incentives, grant or subsidy (could relate to zoning and/or site selection criteria); Infrastructure provision (e.g. water supply and disposal; markets; utilities)
examples	*MOM (Norway); EIA (many countries);* *Settling pond requirements (Thailand)*	*DESTA (Australia); GAA code of conduct*	*Supply and effluent canals in Thailand;* *Differential grant rates in European regions*
applicability in developing countries	Significant where strong government and planning tradition (e.g. Vietnam; China) Limited in countries such as Bangladesh	More easily promoted and applied for larger scale operations; Siting rarely a matter of choice for small scale operations	Locally effective; but limited wider impact where aquaculture activities are highly financially attractive. For the poor, site selection depends on site availability, not economic optimisation
inputs	Disease and stock controls Conditional (on quality; quantity) licensing of seed, feed and chemical production and/or sale; Site limitation/control on feed or seed quantity	Seed and feed producer quality initiatives – better results = more customers	Subsidy or tax breaks related to use of particular ingredients or products; Price premium for use of certain products (organic or eco-label accreditation) Customer requirements (certain supermarkets) Trade controls
examples	*Danish farm feed limits; feed composition limits* *Seed quality initiatives (?)*	*PCR certified seed, Thailand* *Improved fish feeds, generating better returns and better environmental performance*	*Organic labelling of salmon and shrimp* *Export market food quality regulations (eg. EC, US)*
applicability in developing countries	Effective only where motivated and efficient civil service	Highly applicable, dependent upon training/skills/knowledge Only large scale operations able to consider high-tech control.	Food quality regulations already having significant impact in DCs; Price premiums still very limited market share;

Process	Site biomass limits Effluent discharge regulations Animal welfare regulations Food safety regulations	New technology Codes of Conduct/Practice (CoC; CoP)	Quality/organic/environmental labelling relating to production process Environmental Payment Systems linked to beneficial farming practice
examples	*Site biomass limits (Scotland)* *Thai and Vietnamese pond effluent discharge regulations*	*Aquasmart feed management system* *CoC: FAO (global) , FEAP (Europe); GAA (shrimp)* *CoP: Shrimp farmers in Honduras, Ecuador, Thailand; many salmon production companies/associations*	*Organic labelling of salmon and shrimp* *No examples of EPS to date*
applicability in developing countries	Difficult with many small operators. Risk for farmers as limited crop unless other measures imposed	Applicable, but more difficult to promote, implement and monitor/certify with decreasing scale and increasing poverty	Price premiums still very limited market share; Chain of custody difficult to establish with dispersed small scale industry EPS not a priority for DCs
Outputs	Production limits associated with site permit Discharge consents (quality, quantity) Environmental quality objectives and standards (EQS)	CoP ; local user agreements Avoidance of negative impact on own operation (water quality) Marketable by-products (seaweed/urchin for salmon) Polyculture (practice with one impacts on another)	Emission fees (Polluter pays principle); tradeable permits; product quality standards (eg. depuration of shellfish in cat. B etc.)
examples	*Site production limits (eg Scotland, Finland);* *Discharge consents (most developed and developing countries with significant aquaculture industry);* *EQS (Norway; Australia)*	*Local farmer agreements: Thailand; Sumatra, Indonesia*	*Colombian shrimp farm taxation based on nutrient differences in intake and outflow waters*
applicability in developing countries	Typically not enforceable- excessive monitoring and scientific assessment required	Applicable; but requires farmer organisation and co-operation. More complex and difficult with large numbers of small farms.	Requires close monitoring - difficult to apply to small-scale operators

It is clear from Table 1 that many of the tools available are difficult or impossible to apply to small scale farmers in developing countries. It is also questionable as to whether any of these approaches can effectively take account of cumulative impacts or ensure “ecosystem integrity”. In the following sections we discuss the three main tools or approaches (voluntary, regulatory and economic) in more detail, exploring how they can be applied in practice to the different dimensions of aquaculture activity, and their strengths and weaknesses.

Siting and design

The environmental impacts of a fish-farm depends critically on its location and design. The impacts of the sector as a whole, and its vulnerability to disease and poor environmental quality, depend upon more subtle aspects of siting, such as pattern, distribution and density of farm enterprises.

This issue is typically addressed in four ways:

1. Voluntary: through the promotion of site selection criteria (i.e. voluntary and informed);
2. Regulatory: through the imposition of site selection criteria; either standard criteria, or “custom” criteria, based for example on the outcome of EIA.
3. Economic: through the provision of sites, infrastructure, or subsidy (conditional on location/design)

4. **Planned**: two or more of the above, bound together within an overall aquaculture development plan, at minimum addressing where development is to take place, and in more comprehensive plans, how much and how many.

All of these approaches have been used in different parts of the world, in different combinations, and all have had some success.

Voluntary

Voluntary approaches are near universal. Most countries have produced extension materials in an attempt to promote improved site selection and farm design, and this may had had a positive impact in some cases, especially in developed countries where site selection is more clearly subject to entrepreneurial choice . Increasingly also international agencies, governments and industry organisations promote Codes of Conduct (CoC) and Good Management Practices (GMPs) which include appropriate site selection and design criteria.

Most GMPs do address aspects of good site selection for individual operations; but sectoral planning is required for managing numerous operators to avoid problems associated with over-development when multiple operations begin to exceed the carrying capacity of the system. It is clearly not in an individual's or group's interests to encourage new entrants that may contribute to aquaculture exceeding environmental capacity. A self-governing planning regime is therefore likely to:

- prevent new entrants;
- find new development by individuals within a group difficult to adjudicate on.

Participatory planning still therefore requires linkages with responsible authorities and local government to ensure sustainable and equitable development.

Regulatory

Regulatory approaches include broad brush prohibitions on shrimp farming activity in certain national parks or forest land categories (e.g. Indonesia, Thailand, Vietnam); in freshwater areas in Thailand; and within 500? metres of the coast in India (refs***). License permits in many countries are conditional on specific design features, such as the use of settling ponds or predator nets.

Environmental impact assessment also has the capacity to influence siting and design through regulation. Good EIA practice requires alternative sites and technologies to be considered (though this is often not done), and development consent may then be conditional on development of a particular site in a particular way.

Economic

Economic incentives have been successfully introduced in several countries, although to date they have been directed more at encouraging development in suitable locations ("aquaculture parks" or projects) rather than discouraging development in unsuitable areas – which arguably is the greater problem in the case of most aquaculture development.

Thailand now has several major infrastructure projects where large scale canal systems have been designed specifically for aquaculture, ensuring high quality inlet water and effective waste water management. In Europe grant subsidies are available for the development of aquaculture in suitable areas in need of economic development; indeed these subsidies are graduated according to economic need, and can therefore been used to achieve relatively subtle impacts on the location and intensity of development.

Large-scale intensive operators may be persuaded by authorities to establish in certain designated aquaculture development zones by reduced taxation or subsidised start-up costs. This economic approach may be an alternative to or in addition to regulatory instruments such as licencing and *assumes the aquaculturist has a choice of potential sites.* For small scale aquaculturists in developing countries this is often not the case as culture systems are associated with local available land or water resources. These assets may be sub-optimal in relation to suitable conditions for aquaculture, but will still be used for aquaculture if returns exceed those for other uses.

For the landless poor, small scale cage culture has been found to be a successful in poverty reduction strategy[27]. Donor organisations have encouraged development of such culture through training and subsidising the minimal start-up costs. Being low input systems, cumulative environmental impacts are thought to be minimal. Encouraging development is therefore possible in developing countries even for people without many assets. Managing that development once it has been initiated is more difficult, given the large number of operators and limited management resources. Economic management tools therefore present a number of benefits in developing countries as they use the market itself to regulate the industry rather than expensive monitoring and enforcement activities.

Planned

Strategic planning approaches are less common, but are increasing as awareness and understanding of the problems increase. Some examples are:

- New Zealand – Aquaculture Management Areas (AMAs) are to be established following a two year moratorium on new permits. The councils will be required to address adverse effects including cumulative effects of aquaculture (ref)
- Hong Kong SAR has 26 designated "Marine Fish Culture Zones" and modelling is used in certain areas to show impact of various aquaculture species and feed types and levels of activity on prescribed water quality objectives. This has been assisting them in the planning process as well as giving advice to farmers on site selection[28].
- Zambia has conservation planning areas, wherein layout of the land for fish farms can be specified (ref)
- Ecuador has local zoning plans developing agreements between local residents and farmers allowing shrimp farming to continue along with traditional uses and mangrove planting.(ref)
- In Vietnam a major Danida funded project – Support for Brackishwater Aquaculture **-** is underway to promote coastal aquaculture within a planning framework

However, many developing countries do not have the institutional capacity to implement centralised planning and influence aquaculture development. In many areas of South East Asia aquaculture development goes on unchecked, as authorities cannot adequately control the large numbers of small operators involved. Authorities also lack the scientific information on which to base planning decisions.

Strengths and weaknesses

While voluntary and regulatory approaches can deal effectively with issues related to siting and design of an individual farm, they are much weaker than economic and planning approaches in addressing the issue of overall pattern, distribution and density of development. They are therefore likely to be weaker in terms of addressing cumulative sustainability issues, and environmental capacity.

A further problem with the voluntary and (ad-hoc) regulatory approaches, especially for small scale operators in developing countries, is that site "selection" as such rarely takes place. If a farmer sees that he can convert his land to aquaculture, or gain access to a particular piece of land or water to develop aquaculture, and make money, he will do so. Site selection in practice has everything to do with access, availability and feasibility, and very little to do with technical site selection criteria.

Inputs

What comes out of a fish farm depends upon what goes in. Management of inputs (seed, feed, chemicals) therefore has a major influence on environmental impact. The greater the efficiency of food (and specifically nutrient) conversion, the less waste per unit production. Higher quality seed should result in higher survival, less disease, less chemical use, and better food conversion efficiency. Careful use of more effective chemicals should result in less direct pollution, less disease, improved food conversion efficiency, and - for nationally and especially internationally traded products – improved marketability.

Overall, better quality and more effective inputs should allow for higher rates of production within a given environmental capacity, and are therefore essential preconditions for sustained economic growth. Fortunately improved input quality also commonly generates increased short term returns to both feed producer and farmer, and has therefore been afforded substantial attention by governments and different sub-sectors of the industry alike.

In practice high levels of control of input quality and quantity is difficult except for large scale intensive production systems typical of marine finfish farming in developed countries.

About 80% of carp and 65% of tilapia worldwide are farmed without the use of modern compound feeds—feeds formulated from multiple ingredients[33]. These are species where low levels of feed inputs typify production. Although farmers must observe pond conditions carefully in these production systems they are unlikely to be the cause of environmental degradation through increased nutrient loading. Indeed, there is evidence that these systems tend to cause net depletion of nutrients when the harvest is taken into account.

In developing countries there are typically three types of aquaculture production system: fresh or brackishwater pond culture using fertiliser and sometimes supplementary feed; more intensive aquaculture in ponds or cages using trash fish, sometimes mixed with other feed ingredients; and semi-intensive to intensive aquaculture using mainly manufactured pelleted feeds. The environmental impacts of these different systems depend on the quality and quantity of inputs, the nature of the receiving environment and management skills. Use of poor quality or inappropriate feeds and over-feeding can cause environmental problems irrespective of the type of feed used. The opportunities for regional or national level quality control and improved guidance on suitable feeding rates are however greater for manufactured feeds.

The quality of inputs used in aquaculture operations can be influenced through voluntary, regulatory or economic measures or a combination of these within a broader planning and management system. The quantity of inputs used – of equal importance – is dealt with in the section on “process” below.

Voluntary control of inputs

Feed manufacturers in developed countries are constantly striving to produce higher quality feeds that deliver improved financial performance to the fish farmer, through better growth rate and improved food conversion efficiency. These feeds also deliver improved environmental performance, since this is closely associated with food conversion efficiency. The average effluent load (nutrient load / product yield) has significantly reduced in intensive aquaculture due mainly to improvements in formulated feed and improved feeding practices[29]. An improved understanding of the nutritional requirements of cultured species at each stage in a lifecycle along with development of high-energy diets have contributed to less wastage.

Industry sources suggest that the quantity of nitrogen discharged per tonne of salmon production has decreased from almost 180kg/tonne in the late 1970s to less than 40kg/tonne in the mid 1990s. In the case of one land-based salmon farm in Norway reduction of nitrogen content in the feed linked with increased fat content has reduced the ammonia concentration in effluent water by 38%[30]. While these historic improvements have come mainly from improved feed quality, future progress is more likely to come from improved feed management systems.

Increasingly in Western countries, feeds are being designed and marketed specifically to deliver high environmental performance, so farmers can take advantage of various forms of environmental certification and labelling initiatives. These are increasingly tied to codes of conduct (typically applying to the industry as a whole) and codes of practice (typically associated with a particular producer group or marketing initiative. These codes may specify food quality parameters, and/or constrain the sourcing of ingredients to other environmentally friendly industries.

In developing countries the situation is radically different except for the largest producers. Feed types and feed practices are usually far less sophisticated and more difficult to manage. Only a small proportion of farms use formulated feeds (although this is increasing rapidly), and understanding of quality and the trade-off with price is generally poorly understood. Feed manufacture quality control is often low, batch variability high, and storage inadequate, especially in tropical conditions. Farmers are often not able to make informed choices about feed, and the incentive on the producers to maximise quality is therefore limited. Price remains the dominant selection criterion.

Many producers in developing countries use trash fish and/or a variety of household and other wastes to feed their fish. In some cases the wastes associated with these feeds contribute to pond fertility and are therefore essential to maximise production and financial returns. Agreeing standard codes and protocols (whether the objectives are financial return or environmental quality) is therefore problematic.

Seed quality management has become a highly sophisticated business in developed countries, and quality control is often rigorous. There are major commercial incentives to ensure high and increasing seed quality and customer satisfaction. Traceability is hardly an issue; sales are typically direct from hatchery to grower.

In developing countries the situation is again very different, and seed quality may actually be declining in some countries. Until recently seed production was typically undertaken by state run hatcheries. Quality control was usually a matter of scientific pride (although the lack of competition may have worked against this in some cases) although cost was hardly an issue. The private sector is now becoming increasingly important, and is dominant in most countries. Unfortunately, the small scale of enterprise, complex distribution networks and supply chains, and poor handling throughout militates against high quality and traceability. In some cases broodstock is over-used and genetically impoverished. Farmers may not know the hatchery from which their seed was sourced. However, as the scale of enterprise increases, information networks develop, and product traceability improves, a more market-based assurance of quality should become possible. Hatcheries that are found to have supplied disease-ridden seed are unlikely to remain in business for long. However, it is likely to be some time before this process is both effective and fair. Many factors, including genetic quality of the broodstock, rearing conditions in the hatchery, distribution, handling and the farmers own husbandry practices are likely to affect survival and growth. Farmers will always be inclined to blame seed quality rather than their own practices, and objective identification of key points in the production and distribution chain affecting quality will remain difficult. Government may need to intervene, at least in the early stages, to promote higher quality production and distribution and to facilitate exchange of objective information.

Some changes are already taking place. In order to protect their reputation and generate increased demand for seed, some larger and more sophisticated commercial hatcheries are beginning to provide technical assistance to farmers on how to avoid and deal with disease and maximise survival in the early stages.

Regulatory control of inputs

Feed

There are legal limits on phosphorus and nitrogen content of aquaculture feeds in some European countries such as Denmark (ref**). These limits can result in substantial improvements in environmental performance with limited administrative cost. Enforcement is relatively simple in the case of manufactured feed through spot batch analysis at the point of production.

Authorities in some countries (including Denmark and Finland) have also attempted to regulate the quantity of feed inputs, on the basis that this will necessarily limit effluents, and may stimulate improved feed management practices and use of higher quality and less polluting feeds.

Regulatory control of feed input quantity and quality is impractical in most developing countries. A wide range of feeds are used, including locally available vegetable and animal wastes, trash fish, moist pellets made locally from a mix of ingredients, and formulated feed. Control is very difficult except for dry compound feeds, which are not yet the norm in most developing countries.

Seed

Following on from the rapid spread of viral disease in shrimp over the last 2 decades, many countries in S and SE Asia imposed import bans on shrimp and fish seed. The tendency to do this is reinforced by the desirability in some cases to protect fledgling domestic hatchery industries.

The use of wild-caught fry has environmental capacity implications as fry-collection may impact on capture fisheries. In Bangladesh a ban has been imposed on shrimp-fry collection in an attempt to reduce perceived environmental damage, although the ban has not been implemented. Alternative livelihoods are being investigated for the large numbers of fry-collectors that will be affected if the ban is effective. Some estimation of sustainable fry collection linked to the (environmental) capacity of a system to sustain a certain level of fry harvesting would allow some within a community to make a living as fry-collectors.

Chemicals

The control of chemical use in aquaculture is mainly realised through licensing of chemicals and monitoring of farms and end products. Chemicals persistent in the environment are being phased out and farmers are learning the disadvantages of improper use. Both pose challenges for developing countries where institutional limitations often prevent adequate enforcement to check for sale and use of banned chemicals.

Some pharmaceutical companies attempt to offload chemicals that are banned elsewhere onto developing country farmers at a discount. These farmers are keen to use any chemical they feel will protect their crop, often using chemicals improperly (wrong type or too much) due to a lack of training.

The impact of a chemical input can go well beyond the confines of a farm unit and co-ordination on a wider scale is necessary. Local management agreements associated with codes of conduct appear to be the answer here. A major step would be the development of local fora and communication channels that would allow notification of such actions and co-ordination of water exchange amongst adjacent farmers.

3.3.2 Economic management of inputs

Economic management of inputs can be achieved through taxing the use of environmentally damaging inputs – low quality feed and seed, certain chemicals and medicines – or subsidising the use of more environmentally friendly alternatives. A key principle in the use of such tools is to balance taxes and subsidies so that there is no net cost to the sector or the government.

There are few if any examples of this approach having been used to date – either in developed or developing countries.

Management practice

Government can only have limited control over husbandry and management practices on farms – except in so far as controls on siting, design, inputs and outputs influence these practices. Typically therefore farm management is subject to more general manipulation through codes of conduct and codes of practice. Although attempts may be made to enforce codes of conduct or practice this is necessarily difficult in both developed and developing countries, and voluntary industry led approaches are therefore the norm. Voluntary approaches are now being reinforced through market demand and the opportunities offered through government or private certification schemes.

Innovative technology may also have a substantial impact on management practice and environmental performance. Adoption of these technologies may be stimulated indirectly through input and output controls or directly through licensing conditions.

Regulation

Regulatory instruments may used to encourage certain farm practices. In agriculture for example there are specific regulations relating to animal welfare. Practices such as stocking density may in future come under this kind of regulation in European countries, but are unlikely to be either introduced or be effectively enforceable on smaller farms in developing countries.

More generally, international and regional commitments and national legislation may call for "Best Available Technology Not Entailing Excessive Cost" (BATNEEC); but this is not easily enforced, even with respect to large companies in developed countries, and is largely irrelevant in developing countries.

Increasingly therefore governments take a less command and control approach, and seek to facilitate or encourage adoption of good practice. They do this through promotional materials, hosting workshops to develop codes of conduct and practice, and encouraging the development of industry standards such as ISO 14001. Although these standards are voluntary at the present time, there is no fundamental reason why they should become legally required at some point.

Voluntary/industry

Improved management practice by its very nature is most suited to voluntary approaches and industry initiatives. There has been a proliferation in recent years of industry led codes of practice (good management practice, GMP; best management practice BMP) throughout the world. Some of these are very broad based, and seek to address the concerns of consumers, environmental groups and the public at large about the sustainability of aquaculture. Others are much more specific and seek to exploit a market opportunity for a distinct product reared in a particular way (such as organic standards).

Unlike most regulatory approaches, codes of practice usually seek to address every aspect of a culture process - siting/design, inputs, process and outputs. They tend to be flexible documents under continual review, since knowledge and technology are constantly improving, offering new opportunities for improved environmental performance, productivity and profitability.

The drawing up of codes of conduct and codes of practice are becoming commonplace in aquaculture. The two can be differentiated by simply defining codes of conduct as 'how you behave' and codes of practice as 'what you do'. As such Codes of Conduct are more general and can be sectoral in scope, while Codes of Practice tend to be more specific and technical.

Aquaculture codes of conduct have recently been developed by international organisations such as the FAO and FEAP[31] while more technical codes of practice have been developed (mainly for shrimp culture) by NACA and the Global Aquaculture Alliance (GAA) and Aquaculture Certification Council (ACC). In addition industry groupings in a number of countries have developed codes of practice, again mainly relating to shrimp culture including Thailand, Mexico, Ecuador, Honduras, Greece, Bangladesh and Australia.

Codes and GMPs can only be developed effectively on a collaborative basis as they are voluntary and therefore require consensus amongst stakeholders to adhere to the practices outlined. Actual practice is found to lag behind generally accepted good practice as some GMPs may be unfeasible, too costly or otherwise unacceptable[32]. A useful exercise is to compare actual practice with GMP and determine where the largest difference occur and why – this helps to develop appropriate strategies to encourage more widespread adoption of GMP, or address specific issues using alternative tools . Overall the more important aquaculture is to the community, the more interested and committed it will likely be in adopting better practices[33].

Environmental capacity is increasingly referred to in codes of conduct or industry objectives, such as the following 'vision' for aquaculture in Sinola, Mexico[34]:

> *"More efficient aquaculture production, developed within the sustainable capacity of the coastal ecosystem, managed through aquaculture parks, with open participation from all sectors, organised as one association with an entrepreneurial vision."*

Environmental capacity is, however, yet to be used within industry codes of practice as the basis for establishing Environmental Quality Standards and local user agreements.

Feed management

Optimising the feeding regime is both beneficial to the farmer and the environment. Feed is generally the largest operational cost and any waste of feed is therefore a waste of money. Achieving the correct feed input in terms of quality and quantity is therefore a key consideration for maximising production within the limits of environmental capacity. Being so obviously in the farmers own interest, feed management appears more suited to a participatory approach than to a regulatory framework. Regulations may limit innovation, set sub-optimal levels and be difficult to enforce. This is particularly true in a developing country situation. Improved feed efficiencies and regimes are therefore an important part of BMPs.

In small scale fin-fish systems in developing countries, feeding is typically adjusted by observation. With the high labour levels often typical of these production systems, feeding may be adjusted very accurately to reflect demand. In these situations, quality rather than quantity of feed may be a better target for improved environmental management.

The cage culture of lobster in South East Asia mainly involves the use of trashfish as feed. Close observation of feeding habits is conducted and uneaten trashfish is collected by hand from the bottom of the cages and the seabed. The success of this culture system in certain coastal areas has nevertheless resulted in a localised reduction in water quality, causing increased mortality rates and greater costs as farmers move cages to the seaward edge of culture areas seeking unpolluted water. Individual farmer control of feed inputs is not enough in this instance as feed inefficiencies and waste outputs from the cages still culminate in nutrients exceeding environmental capacity.

Feed provision in intensive shrimp farming is generally adjusted according to consumption as indicated by residual food (or lack of it) in a feeding tray. In practice all of these systems tend to adjust feeding rate to the point of satiation. This may still be a higher rate than that generating optimal food conversion, growth rates, and minimum pollution. The return to the farmer may, however, justify a certain level of overfeeding, as they may achieve higher growth rate and production, usually resulting in better profitability.

Reducing the environmental impacts of feed inputs will mainly be through training and local management agreements: farmers can be made more aware of the benefits to themselves of avoiding overfeeding and groups of farmers can reach agreement to avoid the cumulative affects of excessive feeding. The use of potentially unsuitable or unsustainable feeds can also be phased out through these two methods.

Technical innovation can have a major impact on improved feed management. Computerised feedback systems, such as 'Aquasmart', are increasingly used by the salmon industry throughout the globe. The supply of feed is shut off as soon as feed in a cage is left uneaten and falls past a sensor at the bottom of the cage. The system is sensitive enough to significantly reduce feed wastage with the economic and environmental benefits this brings. For aquaculture in developing countries, the Aquasmart system is not suitable as it is associated with a particular feed regime, is costly and impractical to install.

Economic tools

Those economic tools which could be applied to inputs can also have an influence on management practice. Clearly the more expensive an input is, the greater the likelihood of efficient utilisation. A tax on any input whether it be water, feed, seed or chemicals is likely to result in more careful and efficient use.

In practice increasing the cost of key inputs typically causes political outcry (although it has been achieved to a very substantial degree with fuel). For developing countries keen to see aquaculture develop however, such taxes are highly unlikely.

Outputs

The most significant outputs from aquaculture in environmental terms are nutrient loads from uneaten feed and excretory products, and chemical and pharmaceutical products.

Voluntary

Most codes of conduct and practice include provision to reduce waste outputs. It is normally in the farmers interests to increase the efficiency of input use, and this in turn will reduce the level of outputs.

It is less obviously in farmers interests to reduce the actual discharge of waste from his farm – indeed most farmers are keen to flush wastes out of their system and/or surrounding waters as quickly as possible. When these wastes begin to build up in the immediate surroundings, and in turn begin to affect the quality of water entering the farm, the incentive to reduce waste outputs may strengthen, and indeed, peer pressure may also increase.

Voluntary management of waste for the good of an individual farm or groups of farms therefore has significant potential, but may need to be encouraged/facilitated, especially where the problem is a group rather than individual farm problem.

Regulatory

Where resources permit, environmental quality objectives and associated standards have been applied to the aquaculture sector. With EQS are in place, frequent and extensive monitoring regimes are required to ensure compliance with standards. The cost of this monitoring is significant and is either borne by government or, as is increasingly the case, imposed on the producer (the 'polluter pays principle'). Despite these efforts to quantify and regulate outputs such as nutrients, the levels allocated are rarely directly related to the capacity of the receiving environment to deal with those outputs.

In Denmark a discharge quota for nitrogen (550t) and phosphorous (54) is allocated to the aquaculture sector, but the quota is not specifically related to a defined carrying capacity of the environment. Despite the recorded discharge amounts not reaching permitted amounts, in 1997 the Danish Ministry of the Environment and Energy enforced a provisional stop on any new permits and the extension of existing permits[35].

In Finland discharge permits are set on a site-specific basis so can take into account several activities in one area. A practical standard used is 8g P and 70g N per Kg of fish produced, but overall there is poor compliance with companies under-reporting the amount of feed used and fish produced.[36]

With the large number of small operators, variety of feeds, lack of capital and weak institutional capacity for monitoring and enforcement, regulatory policies such as those in Europe are not feasible for developing countries in most instances. A major issue in pond based systems is when and how water is exchanged. Some level of treatment prior to discharge may be agreed (such as settlement ponds) or indeed communal waste management may be developed (such as sludge disposal). Improved co-ordination of water exchange in pond systems should improve the quality of receiving waters and reduce the risk of disease transmission – but the value of this to individual farmers must be demonstrated if voluntary approaches are to work..

Product characteristics

Regulatory management of aquaculture product may be undertaken through food safety standards. The environmental quality of the culture environment is implicit in these food standards. Regulators can and frequently do ban seafood products from being imported and governments may chose to punish the individual producers responsible for the certain seafood products being banned. A suitably efficient traceability system is rarely in place for such action, although recent bans are leading to much greater pressure to introduce such systems. The main incentive for producers to comply with food standards is therefore economic – their product will be either completely unmarketable or miss out on high-value markets.

Participatory management of outputs is very much linked to codes of practice and local agreements. Greece for example has developed a Quality Assurance scheme for Greek farmed sea bass and sea bream through the adoption of a code of practice[37].

Economic

Economic incentives in relation to aquaculture outputs often focus on the product itself as economic instruments in relation to other outputs such as waste pose monitoring and enforcement difficulties. The quality requirements for the product dictate that certain chemicals and medicines are not used in the culture process or for a certain period prior to harvest to avoid the possibility of residue. A number of South East Asian countries have recently experienced major economic losses that accompany European and US import bans due to the detection of chemicals or antibiotics in excessive quantities in seafood products.

Beyond the minimum requirements associated with food safety is a quality spectrum that products are increasingly judged on. Producers are hoping that a greater consumer focus on food safety and environmental issues will create a price premium for certified organic produce and produce from environmentally friendly practices. Organic certification of Atlantic salmon (*Salmo salar*) has recently been developed by the Soil Association, an organic certification body in the UK, which include standards relating to animal welfare, including water quality, stocking density and use of chemicals and feed quality.

An interesting dichotomy may develop in the near future as environmental quality standards could potentially diverge from product quality standards. This is particularly true in relation to feed where seafood product quality standards in Norway and the EU currently require certain levels of marine-derived ingredients in the feed for a product to be classed as 'seafood'. Feed R&D is now focusing on replacement of marine-sourced ingredients with vegetable alternatives, which are less expensive and seen by many environmentalists to be preferable to meal and oil derived from reduction fisheries.

Demand-side interventions (eco-labelling) could give farmers a price premium for sustainable farming and increase the likelihood of voluntary adoption. Consumers will have a hard time visually distinguishing which products are farmed with best management practices so certification is necessary. A sustainable product price premium — which may decrease the probability of purchase yet still increase total farmer revenues — is yet to be determined. Clearly this is a long-term approach which will have to consider differences by species, geographical region and consumer group[38].

Waste outputs

As aquaculture is so dependent upon a high quality environment, direct economic benefits are derived from good environmental practice. Farmers using many Best Environmental Practice (BEP) measures should therefore gain direct economic benefit from their actions with lower costs or higher prices if premiums for BEP are available.

One farmer's good practice may not, however, prevent damage caused by a neighbouring farm's poor practice or the cumulative effects of surrounding farms. Group initiatives therefore remain important to economic management tools even if it is only extending to group training on the benefits of BEP.

In industrialised countries, economic incentives to reduce environmental impact are still not deemed to be sufficient to solve the problem. In 1998 French regulators set charges per unit of nutrient added to the water, calculated based on amount of feed input and FCR. The farmer can reduce fees by using more efficient feed or by installing microfilters. French aquaculturists have, however, received subsides to encourage the installation of treatment systems[39].

In Colombia new legislation that levies a tax based on measured differences between intake and discharge water aims to pay for facilities, especially processing plants, with the incentive to further reduce costs by eliminating contaminants from effluent[40]. It is, however, too early to determine how effective this legislation will be.

With incentives to reduce accumulative pollutants[41] researchers found that incentive-based instruments may also be hard to implement due to:

- ⇒ Monitoring and enforcement problems though it may be easier to regulate inputs instead
- ⇒ Society lacks faith in the adequacy of incentive-based policy
- ⇒ High set-up costs for cleaner technology could prevent a competitive economy achieving optimal pollution management.

Chemical use is also controlled by product standards where detection of certain residues results in import bans, which creates a strong incentive for developing countries to deal with the problem through stricter monitoring and control. This message should eventually permeate down the supply chain until farmers realise that incorrect chemical use means that their harvest cannot be sold.

As with other inputs (seed and feed) positive action on chemical use is too often remedial, with import bans from major customers triggering action. An example of this is the 2002 ban by the European Union on Thai shrimp due to the presence of the antibacterial agent nitrofurazone. One potential reason for this chemical residue being detected is thought to be farm water exchange practices. One farmer who may be applying a chemical treatment and not harvesting may discharge water which enters a farm where harvesting is about to occur.

Potential of alternative approaches to keep aquaculture development within environmental capacity

Environmental capacity has been more widely investigated in land-use planning than for aquaculture. Researchers in this field have found that although the capacity concept may seem to lend itself to a regulatory approach, the regulatory policy tool itself is insufficient to deliver development which does not breach identified capacity constraints[42]. It is therefore recommended that capacity should be determined in a participatory manner, preferably at the development planning stage.

Although determining environmental capacity has the potential to be used in a predictive context, it is likely to be applied to situations where aquaculture is already present or where development is suspected of exceeding EC. Participatory management at the planning stage therefore deals with new developments, changes to existing operations or the influx of new operators.

A combination of approaches (a management system) is required to keep aquaculture within the bounds of environmental capacity. Such a system must incorporate at minimum:

- the definition of a management boundary or zone within which environmental capacity can be addressed;
- environmental objectives for the zone, and associated criteria and indicators;
- agreed limits or response levels for these indicators;
- mechanisms to limit or change aquaculture activities so as not to breach agreed limits
- monitoring to verify that limits are not breached and to guide the intensity of management intervention

Many existing approaches do not seek to meet or maintain any particular environmental target or reference point relating to the wider environment or “ecosystem integrity”. Rather they seek merely to reduce the environmental impact of individual developments, or the impact per tonne of product. Influence on the total level of activity (and overall impact on *ecosystem integrity*) is typically weak, and in developing countries a steady increase in activity rather than restraint is understandably favoured for economic development reasons. Most developing countries lack any kind of management system that would allow them to meet the requirements of Article 9.1.2. of the FAO code of conduct for responsible fisheries.)

Where has EC been applied so far in aquaculture?

MOM and Lenka, Norway

The MOM (Modelling On-growing fish farms Monitoring) management system has been developed in Norway in order to regulate Norway’s considerable industry involved in the intensive off-shore cage farming of salmon. The concept is based on integrating the elements of environmental impact assessment, monitoring of impact and environmental quality standards (EQS) into one system[43].

MOM can be applied both to existing farms and planned farms to allocate an EQS for the farm based on an environmental quality objective (EQO). As most environmental impacts are deemed to be benthic, this EQO is to ‘prevent an accumulation of organic matter that will lead to the extinction of the benthic infauna. The carrying capacity has therefore been defined as the maximum fish production that will allow a viable macrofauna in the sediment under the farm. When the sediment becomes azoic, the capacity is judged to have been exceeded.

In addition to setting EQS in relation to capacity, the system also sets the frequency and intensity of monitoring in relation to the sensitivity of a site. The more sensitive the site to eutrophication, the great the level of monitoring

The MOM system is used primarily at site level, but may be linked to more comprehensive management systems at higher geographic levels and considering other activities. LENKA, a system devised in 1991 to estimate the potential for aquaculture in Norway, has been incorporated into the MOM model so that cumulative effects of various aquaculture sites in a region are considered. The main cumulative consideration has been found to be competition for space rather than environmental impact.

Decision Support Tool for Aquaculture (DESTA), Australia[44]

DESTA was a software tool developed in Australia for rating soil and water information for aquaculture. The tool facilitates aquaculture management decisions regarding site selection and design for new aquaculture facilities. The information required to make an assessment is entered into the computer program through a series of interactive dialogue screens. The computer program then processes the information and produces limitation ratings for each environmental parameter supplied by the user. The output from the software is a tabular summary of user responses classed into slight, moderate and severe limitation ratings. If sufficient information has been supplied by the user, then the software program provides an assessment of the potential of the new site for aquaculture.

DESTA has an environmental capacity dimension as the existing impacts of surrounding users are considered in evaluation as well as the contribution of future impacts by the operation in question.

Total Maximum Daily Limit (TMDL), United States

The Total Maximum Daily Loads (TMDL) system applied by the US environment Protection Agency (EPA)[45] is an example where the capacity of the receiving environment to process aquaculture outputs has been considered. A TMDL specifies the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards, and allocates pollutant loadings among point and nonpoint pollutant sources. A TMDL must contain a margin of safety and a consideration of seasonal variations.

The Environmental Quality Standards associated with TMDLs directly relate to the environmental capacity of the waterbody as the trophic state of the waterbody is determined and quantified reductions to nutrient inputs set where necessary. Reduction levels are set that fully support the designated uses of that waterbody.

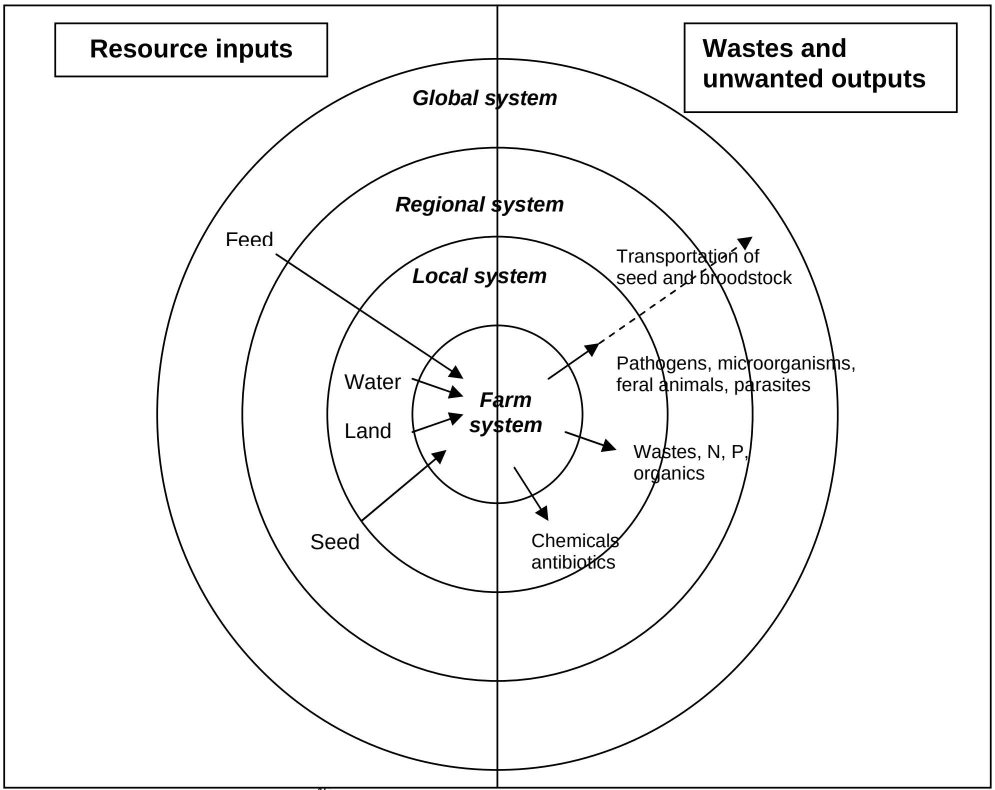
The process of implementing the TMDL system throughout the US has been very slow and onerous. After the system was first established from the Clean Water Act in 1972 it was not until the latter part of the 90's that community pressure resulted in the EPA implementing the scheme through demanding states list waters and set TMDLs.

Appropriate scale for use of environmental capacity

Figure 1 illustrates the environmental impacts of aquaculture in relation to spatial scale. Most of the potential impacts of aquaculture can be considered local and would be expected to decrease with distance away from the source and with time. Other impacts such as interactions between escapees and wild stock leading to genetic alterations and possible reduction or extinction of indigenous species may be more widespread and permanent. These possible impacts are not a result of aquaculture exceeding environmental capacity, but a direct consequence of the presence of a certain type of aquaculture, with limited relationship with the scale or total level of production.

Regulation and management relating to these irreversible impacts, including escapees, introduced species and genetically modified organisms is essential, but in practical terms are beyond the scope of the environmental capacity concept. They should therefore be considered as part of aquaculture planning and management alongside environmental capacity.

Fig. 1 Interactions between aquaculture and the environment at the local, regional and global spatial scales.*



* adapted from Kautsky, N. et al (2000)[46]

As environmental impacts of aquaculture can be global in scale (feed), it follows that environmental capacity can be considered on a global scale. The capacity of the global environment to sustain harvesting of trashfish species or reduction fisheries for conversion to fishmeal and fish oil can be seen to dictate the global capacity for aquaculture.

Aquaculture is one of many feed markets for fishmeal and fish oil and is not currently leading demand for fishmeal and oil or being limited by supply. However, this is likely to change. According to several researchers (see Tacon, 1998[47], Naylor, 1999[48], Barlow, 2000[49]), the global limitations of fishmeal and particularly fish oil in aquaculture feed could become a factor within the next few decades. Assuming that supplies continue to be steady, aquaculture has the theoretical potential to utilise the total annual fish meal supply by 2020 and all of the annual fish oil supply within about five years from now. Chinese aquaculture alone has potential requirements for nearly half the global supply of fish oil by 2015[50]. With current research efforts to develop reduced dependencies on fishmeal and oil, however, the expansion of aquaculture is likely to continue beyond the existing capacity determined by available feed[51].

Although assessment of some dimensions of environmental capacity at global level can therefore be undertaken, the exercise is highly theoretical, and opportunities for associated management measures limited. In practice the most appropriate focus for action in relation to environmental capacity is likely to be at the level of a specific aquatic system (such as a bay, loch or lagoon) or a specific organisational structure (such as a producer group) associated with the sector.

4. The application of environmental capacity in developing countries

5.1 Constraints

The constraints on implementing environmental management in developing countries are considered here to determine what is possible under existing conditions and what changes may be necessary for future implementation. Four key constraints to successful implementation of environmental regulation in developing countries have been identified[52]:

- public sentiment favours economic development over environmental protection
- environmental regulatory institutions are weaker
- fiscal and technical resources for environmental protection are in short supply
- production is often dominated by hard-to-monitor small-scale farms.

The first of these is related partly to different priorities in developing countries, and partly to limited understanding of environmental issues and their links with long term economic success.

Most environmental regulations require a public-sector institution capable of establishing rules of conduct for polluters, monitoring performance with respect to these rules and enforcing compliance. Many developing countries lack the capacity to undertake these responsibilities due to financial and institutional constraints.

There are less formal regulatory instruments that shift some of the burden for monitoring and enforcement onto the private sector. Again developing countries may be poorly placed to introduce greater self-governance as private-sector environmental advocacy is generally less prevalent and less well-organised than in industrialised countries.

The aquaculture sector in developing countries is characterised by large numbers of small operators, which makes self-governance financially attractive to management authorities. This characteristic also makes a reliance on private-sector control extremely risky – large numbers of operators are difficult to monitor. Smaller numbers of large operators simplifies governance, but these operators are often likely to enjoy political influence due to their socio-economic importance to the region.

Codes of conduct have been developed and are in place throughout temperate and tropical aquaculture systems. Local management agreements often form part of such codes or are already in place through the necessities of co-ordinating water exchange, inputs or harvesting in specific culture areas.

5.2 Research

A large EU project to assess the carrying capacity and impact of aquaculture on Chinese bays has recently been completed with the primary output being the development of a GIS package to model and manage aquaculture development in the bays[53]. The three-year project was resource intensive and highly complex, being dependent upon large data sets and extensive modelling conducted by several overseas research institutes. Outputs are intended to both inform further development of the methodologies and technology used as well as informing planning and management of aquaculture in the bays. It is still to be seen whether the stakeholders will use the information as a basis for aquaculture development.

On a smaller scale is an on-going study to estimate the carrying capacity in Kandleru creek in Andhra Pradesh, India[54]. The intention here is to use limited water quality surveys of key parameters (TN, TP, DO, total solids, volatile solids, secchi disc) and link these to land and water use around the creek. The research will attempt to identify factors affecting water quality and set EQOs that can be revised by on-going monitoring. This research links in with development of practical farm-level management measures to reduce effluent load to within the carrying capacity of the creek and combine these into an integrated management of the creek. The research therefore takes the additional step of linking capacity with management measures.

Both studies focus on carrying capacity rather than environmental capacity. This simplifies things as the impacts and needs of other users are not tackled directly. The difficult final steps of agreeing levels of acceptable environmental change between stakeholders and the allocation of capacity amongst the different uses and users are beyond the scope of the research, but these decisions must be made if environmental capacity is to be fully incorporated into aquaculture development.

5.3 Determining Environmental Capacity

It is unclear whether either the large scale, high-tech research of the Chinese Bays project or the smaller scale, low-tech project in Kandleru Creek will result in environmental management being guided by the environmental capacity of the ecosystems involved. For such a situation to occur requires either regulatory or stakeholder confidence in the determination of environmental capacity. Without acceptance of the results of capacity research or of the iterative process to establish environmental capacity, the subsequent setting of more objective EQS is unlikely.

The determination of environmental capacity can lend itself to the particular issues of environmental monitoring in developing countries – a lack of scientific information and a lack of resources to record and collect that information.

5.4 Setting standards

Environmental capacity requires consideration of the cumulative effects of operations on the environment, which is increasingly a feature of standard EIA procedure as well as being inherent to codes of practice and in local management agreements. Ultimately some quantification of impact should be made to define environmental quality standards for either the pond/cage environment or the ambient environment. An EQS, although relating to a measurable parameter, need not be based on an indicator requiring scientific monitoring.

Acceptable changes to the environment partly or wholly as a result of aquaculture can vary enormously depending on the assessors perspective. A fish farmer is likely to set EQS based on effects to his operation and local communities that are highly dependent on aquaculture would be more supportive of the aquaculturalists perspective than less dependent communities. As aquaculturalists are highly dependent upon a healthy environment, stakeholder discussions may well show that EQOs of various parties are similar.

5.5 Monitoring

Monitoring systems are more effective and more cost-effective when government monitoring activities are supplemented by well-designed self-monitoring and information from the local community. Whereas large facilities might be monitored continuously and for a wide range of pollutants, this is unlikely to be feasible (or cost-effective) for many small emission sources. For smaller sources, the standard-setting and the monitoring program perhaps should be based on process or equipment requirements rather than point-of-discharge measurements.[55]

It is recognised that the non-point source nature of the effluent problem associated with aquaculture development suggests direct effluent regulation is unfeasible and voluntary adoption of best management practices is the current approach favoured by industry and international organisations[56]. This approach also lends itself to the situation often found in developing countries where regulatory processes are often ineffective.

5.6 Opportunities

The direct application of environmental capacity in the management of aquaculture is thus far associated with intensive, temperate systems in industrialised nations. Tropical aquaculture systems have only recently become the focus of capacity research and this is yet to translate into capacity-based management measures.

The Environmental Quality Standards or Objectives set are generally determined through large-scale, long-term scientific sampling regimes, which are then supported by active and well-resourced monitoring programmes. These descriptive terms are rarely applied to situations in developing countries, but a highly technical (and high cost) approach is not the only way to introduce environmental capacity into the environmental planning and management of aquaculture.

local agreements and community-based resource management – eg. Thai farmers and experience of community management in Sumatra (see Tobey, J. ref.34)

Stakeholder monitoring programmes – being tried out by Utteran in Bangladesh

Improving linkages between various institutions (as we are doing in Tropeca),

(set up of producer/trade organisations, training initiatives, suitable economic and market incentives – see Van Houtte 96 and Bailly & Willmann 2001)

5. Conclusions

1. There are significant environmental and resource use problems associated with the rapid development of aquaculture in developing countries, and these are associated with substantial social and economic costs, especially to poor producers

2. Existing reactive and regulatory approaches to environmental management are largely unworkable, and inadequate to address cumulative and wider aquatic system issues in tropical aquaculture

3. The concept of environmental capacity offers a possible framework for raising awareness amongst producers of the need to cooperate and work together to address wider environmental issues

4. A range of models and adaptive management approaches have been developed by scientists and policy makers which seek to manage aquaculture within environmental capacity. There is an opportunity to test some of these out in practice in a rapidly changing developing country context

5. In order to stay within carrying capacity, there will be, in many cases the need to manage and possibly limit the scale, density and intensity of aquaculture and other uses of shared water resources. This has implications for the allocation of access and use rights, and cuts across the key issues of equity and poverty.

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TROPECA

Status of aquaculture and associated environmental management issues in Vietnam

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Contents

1 Introduction

Aquaculture has been practiced for millennia together with the establishment and development of the Vietnam country. However, aquaculture with the application of the scientific method to aquacultural production has developed since the 1960s in Vietnam. The development of aquaculture underwent different stages related to the political and economic circumstances of the country.

Many freshwater and some brackish-water fish species such as mullet, milkfish and Asian seabass has been cultured in ponds since the 1960s (Son, 1996[b]). Freshwater cage culture in the Mekong delta has originated from Cambodia also in the beginning of 1960s (Phuong, 1998). Marine cage culture began to develop in the early 1980s when the Department of Fisheries (DoFI) in Binh Tri Thien province (Thua Thien-Hue Province now) initiated shrimp culture (*Metapenaeus ensis*) in pens in Tam Giang lagoon. However, actual marine cage culture developed in the form of lobster culture in Khanh Hoa province based on the study conducted by Ho Thu Cuc (UoF) in collaboration with Khanh Hoa DoFI in the early of 1990s (Tuan et al., 2000).

Aquaculture has great potential to develop in Vietnam. It has a 3,260km coastline, 12 lagoons, straits and bays, 112 estuaries, canals and thousands of small and big islands scattering along the coast. In the inland area, an interlacing network of rivers, canals, irrigation and hydro electric reservoirs has created a great potential of water surface with an area of about 1,700,000 ha, in which:

- 120,000 ha are small ponds, lakes, canals, gardens;
- 340,000 ha are large water surface reservoirs;
- 580,000 ha are paddy fields which can be used for aquaculture; and
- 660,000 ha are tidal zones.

The above figures are not included the water surface of rivers and about 300,000 - 400,000 ha of straits, bays and lagoons along the coast, which can be used for aquaculture activities but have not been planned yet (MoFI, 1994[a]). In addition, Vietnam has various species of which many are high valued and suitable for culture. The number of freshwater fish species, brackish- and salt-water fish species are 544 and 186, respectively. Sixteen major shrimp species of high economic value have been recorded, and the following species have been cultured*: P.monodon, P.merguiensis, P.indicus, Metapenaeus ensis, P.orientalis, Panulirus ornatus, Macrobrachium rosenbergii*.: Some major molluscs species include pearl oyster, oyster, scallop, clam, cockle, snail, of these pearl oyster, clam and cockle have been put under culture. Among 90 high valued seawead species, *Gracilaria spp* (11 species), *Sargassum , Kappapsycus alvaresii* are the most noticeable. In the future when Vietnam can produce hatchery seed, aquaculture is predicted to develop strongly.

Regarding the management structure operating in the country, the Central Government is the highest executive organ. It is responsible for the issues related to politics, economy, culture, society, national defense, and foreign policy. The government administers all affairs of the national fishery development mainly via the Ministry of Fisheries (MoFI). The MoFI controls all institutions in its sector in order to fulfil annual, five-year, and long-term plans, which are handed by the national assembly or the central government. It has the right to issue legal documents in order to implement decisions made by the national assembly and the central government. The MoFI's stipulations affect all ministries, People's committees, institutions, and citizens in the whole country (National Institute for Administration, 1995). The Central Government in general, the MoFI in particular has paid much attention on aquaculture (Tuan et al., 2000).

The Central Government or the MoFI manages all big projects (normally more than US$ 5 million) related to the development of aquaculture/fisheries of the country. Other institutions such as the Ministry of Planning and Investment (MPI), the Ministry of Agriculture and Rural Development (MOARD), the Ministry of Education and Trainning (MOET), the Ministry of Science, Technology and Environment (MOSTE) or its National Center for Natural Science and Technology (NCNST) play a role as consultant units for the government or collaborating partners of the MoFI. Institutions in the fisheries sector such as research institutes for aquaculture (RIA) No I, II, and III and in other sector such as universities, oceanographic institutes, etc are implementation units through scientific contracts/research projects (fig. 1).

Based on the MoFI's direction, provincial Departments of Fisheries (DoFI) were making plans for the development of aquaculture in each province in order to fulfil their tasks at lower levels. The management system at the provincial and district levels is similar to that at the national level. However, provinces and districts normally manage projects at lower values.

The DoFIs have its agencies such as Agency for Resources Management (ARM), Center for Aquaculture Extension (CAE). Those are implementation units at the local levels.

The management structure looks well organized. However, at local government levels, there were some constraints as follows (Tuan et al., 2000):

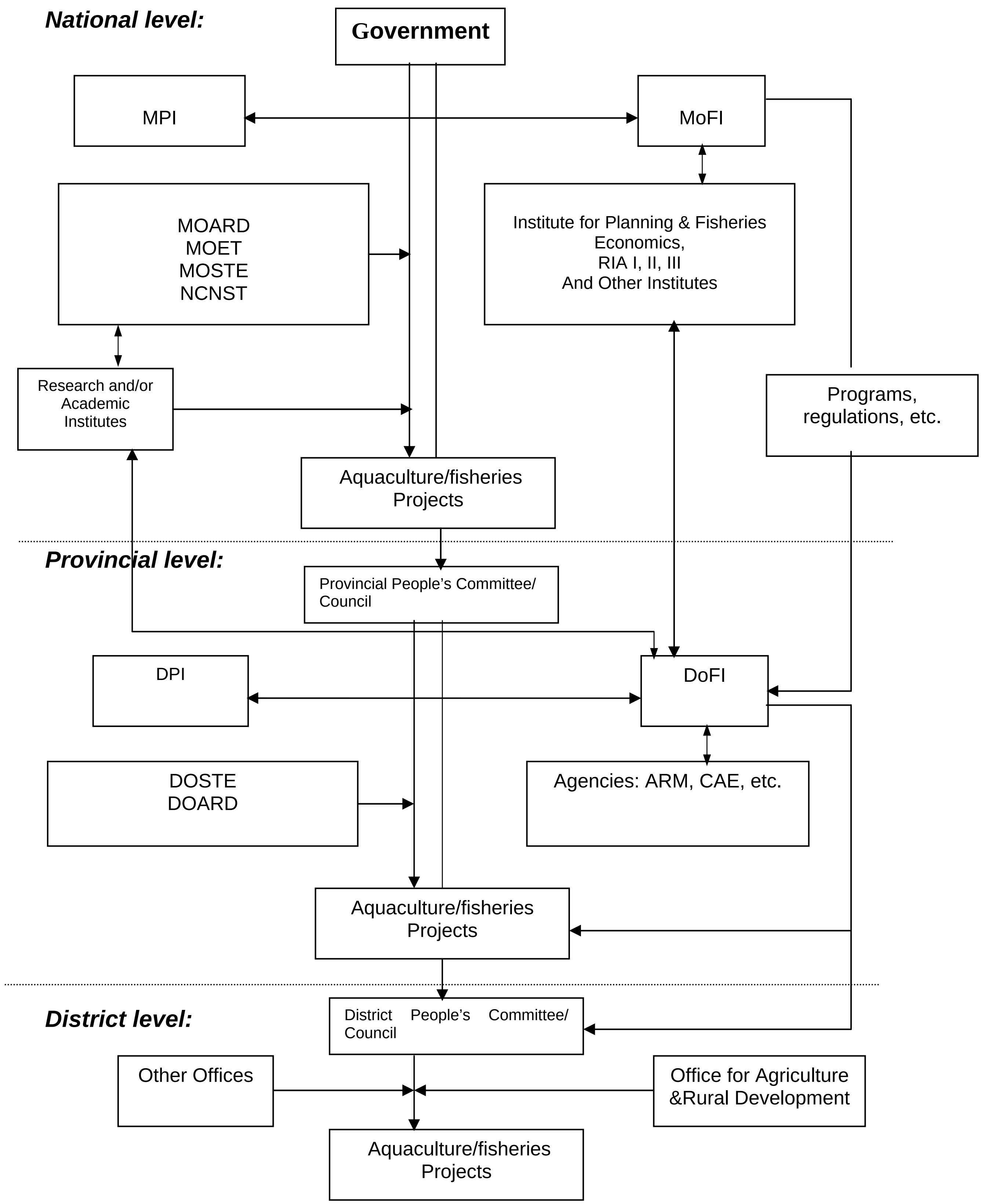

Figure 1. Management framework for aquaculture/fisheries development in Vietnam.

- DoFI has not enough power to manage all issues related to the development of aquaculture and fisheries in its province. The DoFI is not the final decision-maker. It works as a consultancy unit for the Provincial People's Committee. Its documents sent to districts are not mandatory.

- At district level, there is no fishery office. Only one person in agriculture office is responsible for fisheries. Therefore, there is a shortage of manpower to implement DoFI's plans at district and commune levels.

2 Current Aquaculture Activities

2.1 Aquaculture production

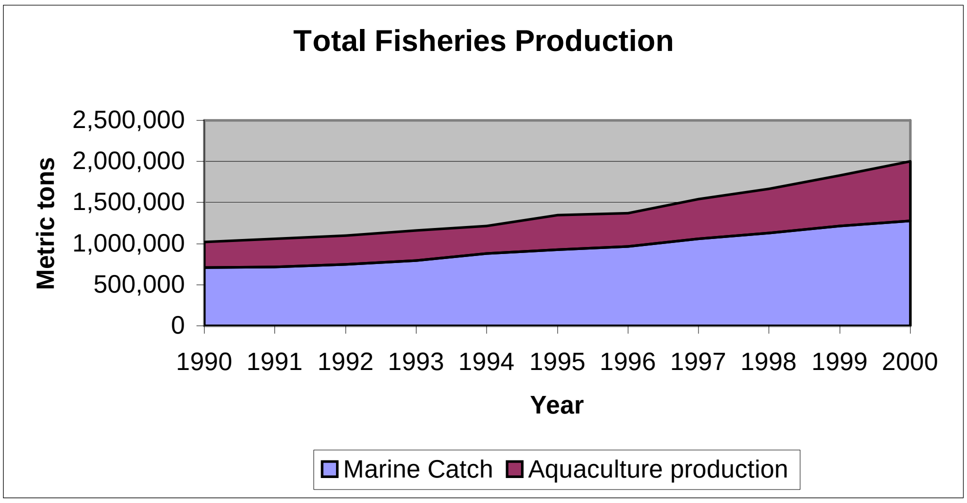

Figure 2. Total fisheries production of Vietnam
(Source: MoFI's annual reports)

The aquaculture production of Vietnam has increased more than twice for ten years since 1990 (fig.2). The production tends to increase in the future.

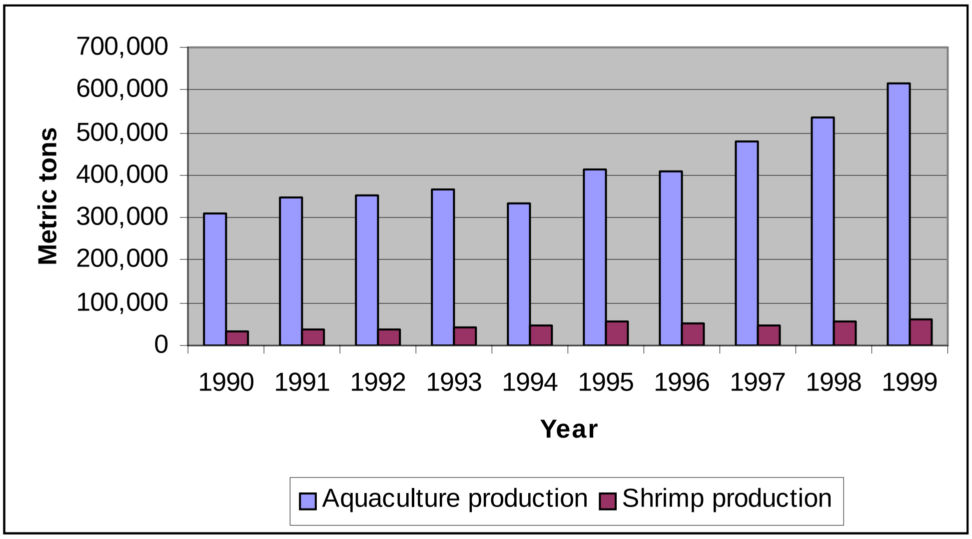

Figure 3. Shrimp production compared with aquaculture production
(Source: MoFI, ????)

Although shrimp production accounted for 10% of the total aquaculture production (fig. 3) and about 3% of the total fisheries production, it contributed to the exports approximately 30% and 50% of the volume and the value, respectively (figs. 4&5).

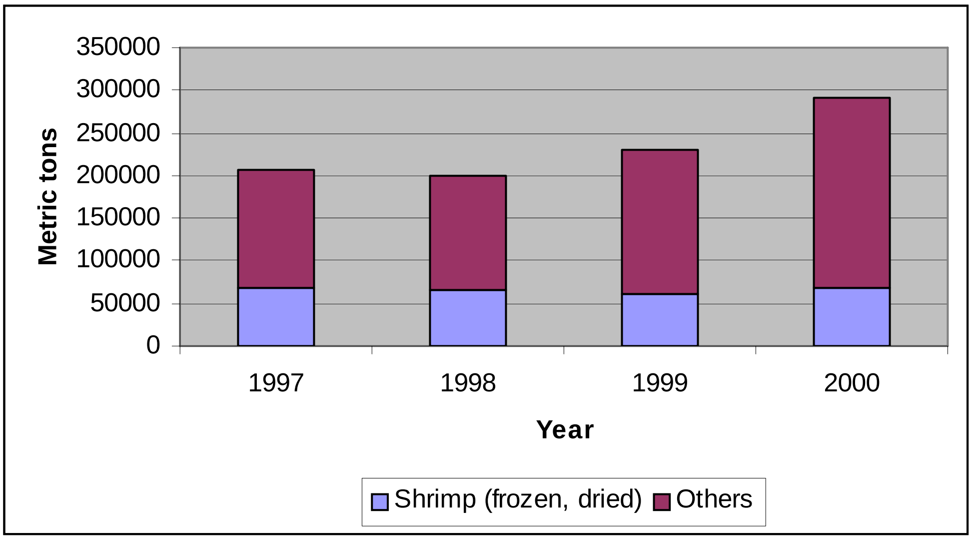

Figure 4. Shrimp and other seafood exports by year

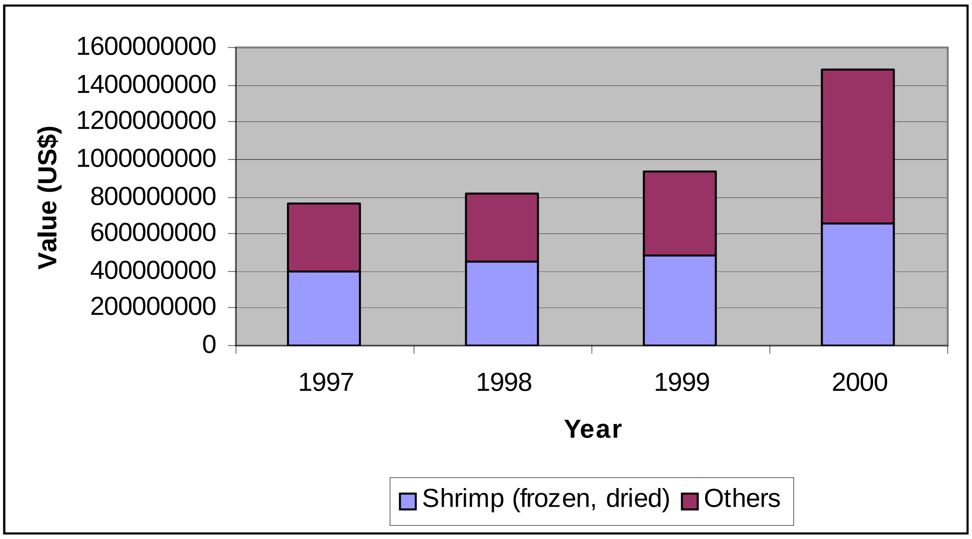

Figure 5. Export value of shrimp and other seafood by year

Because of high profit, shrimp farming continued to be expanded, though it faced with the disease outbreak in the years of 1996 and 1997 that caused 10% of the total production lost.

2.2 Culture areas and species cultured

The main culture types and areas as follows (fig. 6):

- Inland aquaculture:

- ✓ Pond culture: common in the delta of the North and the Middle. The main culture species are common carp *(Cyprinus carpio)*, bighead carp *(Aristichthys nobilis)*, silver carp *(Hypophthalmichthys molitrix)*, grass carp *(Ctenopharyngodon idellus)*, and tilapia *(Tilapia mossambica, T. nilotica)*.
- ✓ Cage culture: in reservoirs in the mountainous areas and along the Mekong river in the South. Most reservoirs were impounded after 1954 for various purposes such as irrigation, hydro-electricity, flood control and water supply. The reservoirs may be classified into large (more than 10,000 ha of water surface), medium (1000 – 10,000 ha), and small ones (less than 1000 ha). There are very few large reservoirs including Hoa Binh (19,000 ha), Thac Ba (18,000 ha), Tri An (32,400 ha), Thac Mo (10, 600 ha), and Dau Tieng (18,000 ha). Most are medium (460 reservoirs) and small. In some medium and large reservoirs, cage culture is commonly practiced. Grass carp (*Ctenopharyngodon idellus*), sand goby (?), snakehead (*Ophiocephalus spp*), common carp (*cyprinus carpio*), and catfish (*Clarias spp*) are the major culture species (Van and Luu, 2001). While Catfishes (*Pangasius bocourti* Sauvage 1880, *P. hypophthalmus* Sauvage 1878, *P. micronemus* Bleeker 1862, *Clarias macrocephalus* (Gunther 1864) and *C. gariepinus* (Burchell 1822) are commonly cultured in the Mekong delta. (Phuong, 1998).

- Coastal aquaculture:
  - ✓ Pond culture: tiger shrimp *Penaeus monodon* is the major species cultured along the coast of Vietnam, especially in the Mekong delta.
  - ✓ Cage culture: the major culture areas are Quang Ninh, Phu Yen, and Khanh Hoa provinces. The main species cultured are lobster (*Panulirus ornatus, P. hormarus, P. timpsoni,* and *P. longipes*), groupers (*Epinephelus bleekeri, E. akaara, sexfasciatus, E. malabaricus, E. coioides, E. merra* and *Cephalopholis miniata).* In addition, some other species such as Seabass *Lates calcarifer*, Yellowtail *Seriola dumerilli*, Sea bream *Parargyrops edita*, Snapper *Lutjanus spp*., Sea-horse *Hippocampus*, Pearl oyster (*Pinctada maxima*, and *P. martensii*), and ornamental fishes were also cultured in cages (An 1993; Tuan 1998).

Figure 6. Major Aquaculture Areas in Vietnam

Table 1. Summary of the commonly cultured species in Vietnam

Scientific name & synonyms	Common names	Distribution	Habitat	Seed supply	Culture form	Culture Area
Epinephelus akaara	Hong Kong grouper, Red spotted grouper, Red grouper	Ton Kin gulf, Southern Central Sea	Marine, demersal; common in rocky areas	wild	Cage, pond	Quang Ninh, Phu Yen, Khanh Hoa
E. malabaricus	Malabar grouper, Estuarine grouper	Ton Kin gulf, Southern Central Sea	Marine and brackishwater. Coral, rocky reefs, sandy and muddy bottoms, tidepools, estuaries, mangrove; juveniles occur in shallow coastal waters and estuaries	wild	Cage, pond	Quang Ninh, Phu Yen, Khanh Hoa
E. merra	Honeycomb grouper	Ton Kin gulf, Southern Central Sea	Shallow-water coral reefs in lagoons and bays	wild	Cage, pond	Quang Ninh, Phu Yen, Khanh Hoa
E. coioides	Orange-spotted grouper	Ton Kin gulf, Southern Central Sea	Marine and brackishwater	wild	Cage, pond	Quang Ninh, Phu Yen, Khanh Hoa
E. sexfasciatus	Sixbar grouper	Ton Kin gulf, Southern Central Sea	Silty sand or muddy bottoms	wild	Cage, pond	Quang Ninh, Phu Yen, Khanh Hoa
E.bleekeri	Duskytail grouper, Yellow spotted grouper	Ton Kin gulf, Southern Central Sea	Shallow rocky banks.	wild	Cage, pond	Quang Ninh, Phu Yen, Khanh Hoa
E. fuscoguttatus	Brown marble grouper, Flowery cod	Ton Kin gulf	Marine, shallow coral reefs and rocky bottoms, clear water. Juvenile found in sea-grass	wild	Cage	Quang Ninh
E. tauvina	Greasy grouper, Green grouper	Ton Kin gulf	Marine, clear oceanic water and coral reefs, hard bottom; juveniles in reef flats and tidal pools	wild	Cage	Quang Ninh
Cephalopholis miniata	Coral hind	Southern Central Sea	Well-developed exposed coral reefs, clear water	wild	Cage	Khanh Hoa
Seriola dumerili	Amberjack, Dumeril's amberjack	Southern Central Sea Lat. 14°34N to 17°32N; Long. 109°30E to 108°22E	Intersecting water with floating seaweed	wild	Cage	Da Nang
S. nigrofasciata	Black-banded kingfish, Black-banded travelly	Southern Central Sea Lat. 14°34N to 17°32N; Long. 109°30E to 108°22E	Intersecting water with floating seaweeds	wild	Cage	Da Nang
Rachycentron canadum	Black kingfish			Hatchery	Cage	Quang Ninh
Lates calcarifer	Seabass, Baramundi	Ton Kin gulf, Central Sea, South Sea		Wild, Hatchery	Cage	Khanh Hoa
Hipppocampus kuda	Black sea-horse	Southern Central Sea		Wild, Hatchery	Cage	Khanh Hoa
H. trimaculatus	Three-dotted sea-horse	Southern Central Sea		Wild, Hatchery	Cage	Khanh Hoa
H.histrix	Thorn sea-horse	Southern Central Sea		Wild, Hatchery	Cage	Khanh Hoa
H. spinosissimus	Short-mouth sea-horse	Southern Central Sea		wild	Cage	Phu Yen, Khanh Hoa
Panulirus ornatus	Yellow ring spiny lobster	Southern Central Sea	Rocks	wild	Cage	Phu Yen, Khanh Hoa
P. hormarus	Spiny lobster	Southern Central Sea	Rocks, coral reefs	wild	Cage	Phu Yen, Khanh Hoa
P. stimpsoni	Hair spiny lobster	Northern Central Sea	Rocks, coral reefs	wild	Cage	Phu Yen, Khanh Hoa

Table 1. Summary of the commonly cultured species in Vietnam (cont.)

Scientific name & synonyms	Common names	Distribution	Habitat	Seed supply	Culture form	Culture Area
Ctenopharyngodon idellus	Grass carp			Hatchery	Cage, pond	The whole Country
Oxyeleotris marmoratus	sand goby			Hatchery	Cage, pond	In the South
Ophiocephalus spp	snakehead			Wild	Cage, pond	South
Channa micropeltis	Spot snakehead			Hatchery	Pond, canal, rice-field	Mekong delta
Channa striata	Snakehead			Hatchery	Pond, canal, rice-field	Mekong delta
Leptobarbus hoeveni	Hoeven's carp			Wild, Hatchery	Pond, canal, rice-field	Mekong delta
Cyprinus carpio	common carp			Hatchery	Cage, pond	The whole Country
Pangasius bocourti	Catfish			Hatchery	Cage	Mekong delta
P. hypophthalmus	(River) Catfish			Hatchery	Cage, pond	Mekong delta
P. micronemus	Catfish			Hatchery	Cage	Mekong delta
Clarias macrocephalus	(Walking) Catfish			Hatchery	Cage, pond	South
C. gariepinus	Catfish			Hatchery	Cage, pond	South
Trichogaster pectoralis	Snakeskin gouramy			Hatchery	Pond, rice-field	Mekong delta
Barbodes gonionotus	Silver barb			Hatchery	Pond, canal, rice-field	Mekong delta
Barbodes altus	Tin foil barb			Hatchery	Pond, canal, rice-field	Mekong delta
Anabas testudineus	Climbing perch			Hatchery	Pond, canal, rice-field	Mekong delta
Notopterus notopterus	Grey feather back			Wild, hatchery	Pond, canal, rice-field	Mekong delta
Notopterus chitala	Feather back			Hatchery	Pond, canal, rice-field	Mekong delta
Hypophthalmichthys molitrix	Silver carp			Hatchery	Cage, pond	The whole Country
Aristichthys nobilis	Bighead carp			Hatchery	Cage, pond	The whole Country
Labeo rohita	Rohu, Indian carp			Hatchery	Cage, pond	South
Cirrhinus mrigala	Mrigal			Hatchery	Cage, pond	South
Tilapia mossambica	Tilapia			Hatchery	Cage, pond	South
Tilapia nilotica	Tilapia			Hatchery	Cage, pond	South

(*Source*: Son,1996 [a,b]; Hambrey, 1997; Trai, 1997; Phuong, 1998 Phuong, 1998; Tuan, 1998; Khanh et al., 2000; Van and Luu, 2001; Viet, T.T., et al., 2001)

Seed supply is one of the major constraints to the development of the aquaculture in Vietnam.

2.3 Production systems

The two main aquaculture systems in Vietnam were pond and cage culture. Ponds were used commonly to raise shrimp along the coastal area and freshwater fish species in the inland area. Ponds normally had a rectangular shape and various sizes from 100 square meters to few hectares. Cages were designed in various ways depending on cultured species and characteristics of culture areas (Table 2).

Table 2. Summary of commonly used cages in Vietnam

Candidates cultured	Cage				Culture Area
	Type	Shape & size	Frame	Bag	
Freshwater Fish as food: Grass carp, sand goby, snakehead common carp, catfishes, etc.	Floating	Bottom: rectangular; Various size: 2*4*1.5, 3*4*2m, etc.	Wood and buoys	Net	Sites with various depths in reservoirs in mountainous areas and in Mekong river
Marine Fish as food: Grouper, Yellowtail, Black kingfish, Seabass, Snapper, Seabream, etc.	Floating	Bottom: rectangular; Various size: 2*1.5*2, 2*2*2, 4*2*2, 10*5*2, 3*3*3m, etc.	Salt-resistant wood and buoys	Net	Sites with depth of 20-50m in bays (Quang Ninh, Da Nang, Phu Yen, Khanh Hoa)
Ornamental fish: Butterfly fish, anemone fish, etc.	Floating	Bottom: rectangular; Various size: 2*1.5*2, 2*2*2, 4*2*2	Salt-resistant wood and buoys	Net	Sites with depth of 20-50m in bays (Khanh Hoa)
Fish as medicine: Sea-horse	Fixed	Bottom: rectangular; Size: 3*6*1m	Salt-resistant wood	Net with mesh size 1mm	Estuaries, lagoon (Khanh Hoa)
Crustacean: Lobster	Fixed	Bottom: rectangular, square Various size: 2*1.5*2, 2*2*2, 4*2*2, 10*5*2, 3*3*3m, etc	Salt-resistant wood	Net	Shallow sites in bays (Phu Yen, Khanh Hoa)
	Submerged	Bottom: rectangular, square; Bottom area 20-50m, height 1-1.5m	Iron	Net	Shallow sites in bays (Phu Yen, Khanh Hoa, and Ninh Thuan).
Shrimp	Fixed	Cylinder shape; bottom diameter 2.5-2.8m, height 1.5m	Bamboo	Bamboo	Lagoon (Thua Thien-Hue)

(Source: An, 1994; Son, 1996 a,b; FEC of Thua Thien-Hue province, 1998; Luong, 1998; Phuong, 1998; Van and Luu, 2001*)*

In general, the aquaculture industry in Vietnam developed spontaneously (fig. 7). That caused pollution, and disease problems. Many shrimp farms and recently, lobster farms have failed.

Figure 7. Lobster farms spontaneously bloomed out in Khanh Hoa

2.4 Feed supply

Feed and feeding applied in culture of commercially important species in Vietnam are summarized (RIMP, unpubl; Ky, 1994; FEC, Ninh Thuan province, 1996; Trai, 1997; Deng, 1998; Phuong, 1998; Hoa, 1999; Thuy, 2000) in table 3 below.

More than 42 kinds of formulated feed such as CP.Group, Betagro Kiladum, Classic, Grobest... (Thailand), Woosung (Korea), Seahorse (Taiwan), KP 90, Thanh Toan, Nam O (Da Nang, Vietnam), etc self-made feeds and trash fish were applied in shrimp farming in Vietnam. Many formulated feeds were applied without confirmation/testing of their quality.

Lobsters are fed exclusively with fresh whole or chopped fish and shellfish (fig. 8). The most commonly used species/groups for feeding lobster are Lizardfish (*Saurida* spp); red big-eye (*Priacanthus* spp); Pony fish (*Leiognathus* spp); pomfret; snails, oyster and cockles; small swimming crab, other crabs and shrimps. Finfish comprizes about 70% of the diet, with 30% shellfish. The preferred fish (comprizing 38% of fishes in diet) was lizardfish (Tuan et al., 2000).

Table 3. Feed and feeding in aquaculture in Vietnam

Cultured species	Feed and feeding
Tiger shrimp	Most intensive and semi-intensive farms used pelleted feeds. The self-made feeds and trash fish were commonly used in the third month just before harvest to reduce the production cost in those systems or in extensive farms. Feeding and FCRs changed depending on the kind of feed used.
Groupers	Trash fish; FCR=4-17; feeding 3-5% body weight;
Asian Asian sea bass	Trash fish
Black kingfish	Trash fish
Yellowtail	Trash fish and crustacean.
Sea-horses	- Fry: zooplankton, mainly on *Copepods*; feeding 10-15% body weight, twice/day: 8am and 4pm. - juvenile (>30mm body length): small crustacean such as *Mysidacea, Palaemonidae, Amphipods*, *Lucifer*, etc.; feeding 5-8% body weight; twice/day: 8am and 4pm. - Sea horses only feed on live food.

Lobsters	Preferred feeds were shellfishes such as mollusks, crustacean. Among trash fishes, lobsters prefer Red Big-eye, Pony fish, Lizardfish. FCR= 28-29. Feeding: small-sized lobster: 3-4 times/day. Feed amount was increased in the evening. Trash fish was chopped into small pieces, and mollusks'shells were excluded; Large-sized lobster (>400g/pc): 2 times/day. Feeding intensity of lobster increased strongly just before melting. In last few months of a culture cycle, shellfish amount (mollusks, crustacean) was increased while trash fish decreased
Pangasius fishes	Self-made feeds or moisture feeds made of trash fish and rice bran was used. FCR=1.35-6. Feeding frequency = 1-5 times/day.

Figure 8. Preparing trash fish for lobster

Previous studies (Trai 1997) have shown that only whole fresh trash fish are used, and that food conversion ratio in cage culture, averaging 5.9 (fresh weight) is significantly higher than that for pond culture of grouper where average FCR was found to be 4.3. Feed costs comprise around 18% of the farm gate price of grouper.

Using low quality formulated feeds and trash fish may cause polluted waters.

2.5 Seed supply

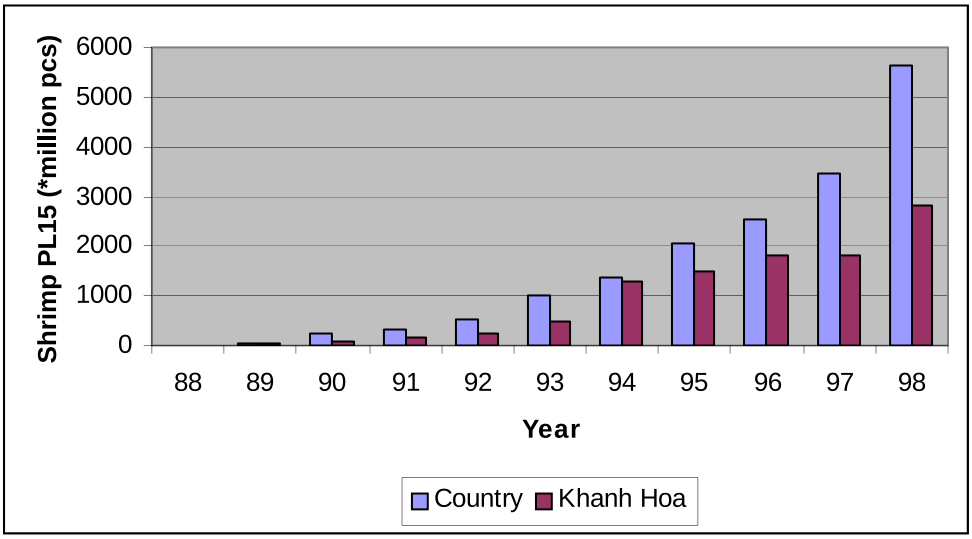

Figure 9. Shrimp seed production by year

The shrimp seed production has increased significantly since the year of 1988. Khanh Hoa was the first and biggest producer of shrimp PLs (Fig. 9). Recently, many provinces countrywide have produced the seed to meet their own demand. However, there is still a shortage of the seed in the country. Some enterprises imported shrimp brood-stock from the Philippines, Singapore, and Malaysia. There have been disease outbreaks due to the imported brood-stock in the country.

Total grouper seed came from the wild and recently from hatchery. One hatchery in Khanh Hoa province produced approximately 1,000,000-2,000,000 juveniles a year enough to meet the local demand and sell elsewhere. In addition, some companies imported grouper seed from Taiwan and sell locally without testing the quality. There was a disease problem in Khanh Hoa last year (fig. 10).

The potential supply of lobster seed is also being assessed roughly from first principles, using area of suitable habitat and natural productivity as indicators of potential seed production. Total lobster seed production (mainly *Panulirus ornatus*) of the country was around 1,000,000-2,000,000 pieces per year in recent years. The price was between VND 20,000 and 120,000 (ca US$1.4-8.6). It tended to increase by size and by year.

Figure 10. Disease Outbreak in grouper cultured in Khanh Hoa

Although the central government had a decree on seed management (MoFI, 1996[a]), the implementation seemed not good in practice.

2.6 Ownership

Ownership of aquaculture farms changed over time and by region. Before 1975 all famrs in the North belonged to the state-owned or co-operative-owned enterprises while in the South those were private-owned ones. Between 1975 and 1986 Most aquaculture farms were co-operatives. Since 1986, especially after “the open door policy” was initiated, aquaculture farms have belonged mainly to the private sector since 1990. Although most of them were active and creative, they were still too small to compete with foreign companies, and too spontaneous to plan. Currently, there have been five major types of enterprises as follows:

- Private;
- Improved co-operative;
- State-owned;
- Joint-venture between Vietnam partner(s) and overseas partner(s); and
- 100% foreigner-owned

Planning and environmental management were implemented well in large-scale enterprises.

3 Environmental issues

Table 4. Summary of Environmental status associated with aquaculture

Env. issue Type of aquaculture activity	Water quality (nutrient loading)	Water manag ement	Disease	Habitat damage	Soil quality	Feed	Owner-ship / access rights	Soil salination
Aquaculture in the North (Dr Dung?): 1. Shrimp culture 2.								
Aquaculture in the Middle: 1. Cage culture of lobster, sweet snail, seabass in Xuan Tu water. 2. Shrimp culture and lobster nursing in Nha Phu lagoon 3. Farming of shrimp, grouper, and lobster in Cam Ranh bay.	 *** *** **	 * * *	 ** *** ***	 ** *** **	 ** ** ***	 *** ** ***	 *** *** **	 * *** ***
Aquaculture in the South (Mr Tu?): 1. Shrimp culture 2.								

**** is high; ** is medium, * is low*

In the Middle from Da Nang province to Binh Thuan province, Khanh Hoa was the biggest producer of aquaculture products, especially sea products. There are approximately 12,000 lobster cages producing ca 1,000mt a year currently. There have been disease problems for two years. This may result from exceeding of the environmental capacity, especially in the Xuan Tu water where there were about 2,000 lobster cages and 100 snail cages and few seabass cages. The same problems also occurred in the Nha Phu lagoon and in the Cam Ranh bay.

4 Environmental policy and implementation at national or regional level

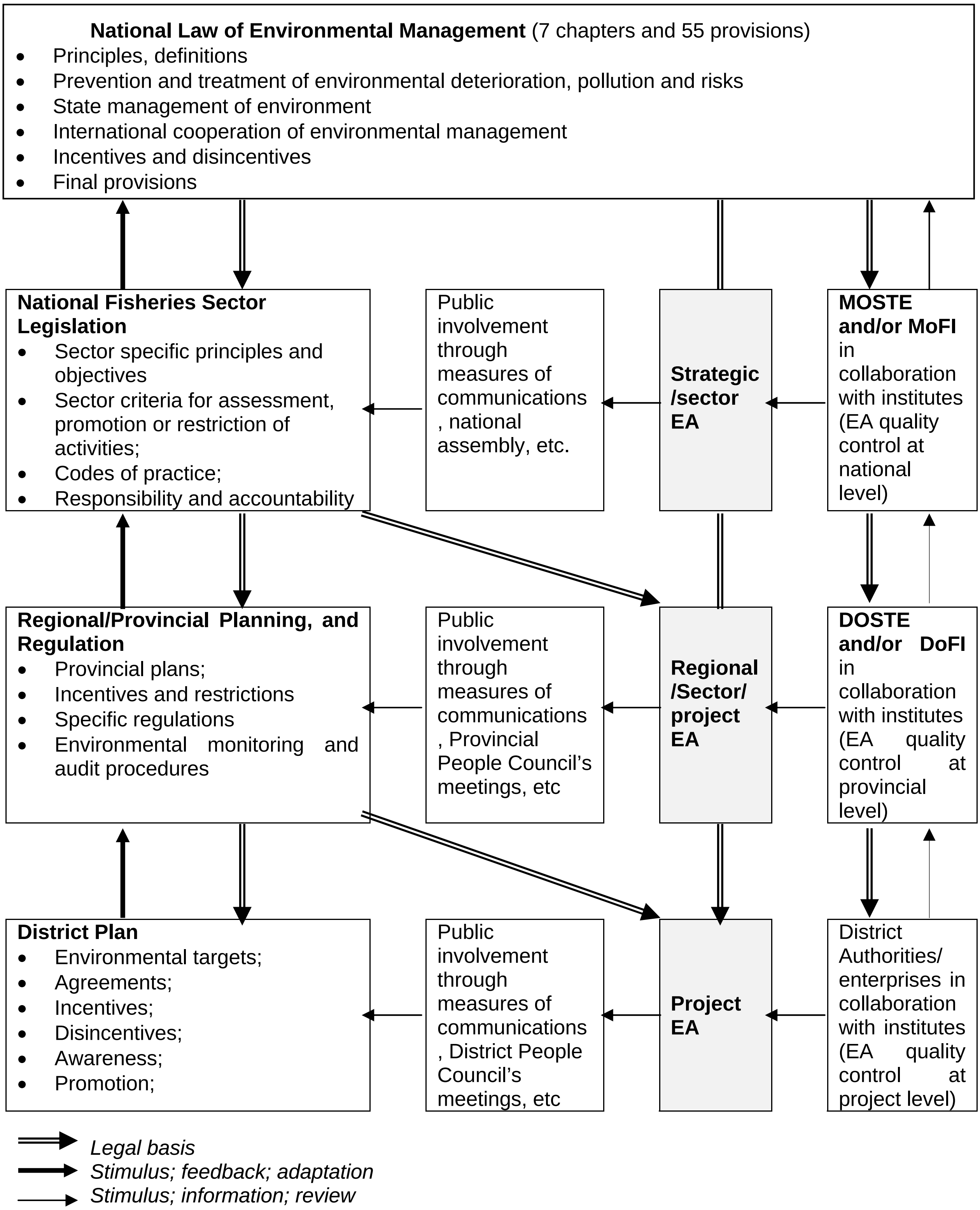

Figure 11. The legal and institutional Framework for Environmental Assessment (EA) of Aquaculture in Vietnam

The environmental policy and implementation at national or regional level are summarized in the figure 11 (Modified from Hambrey's model, 1998). Vietnam Assembly issued the Law of Environmental Management in 1994. The Law 's objectives were to protect the salubrious environment for people's health and living, to ensure the sustainable development in harmony with the whole environment. At lower levels including regional/provincial/sector and district levels, there were plans, regulations, and criteria for the environmental management (table 5). In general, each province or each sector had its own regulations based on the Law and its specific circumstances.

Table 5. National environmental quality standards of coastal water

Factor	Unit	For swimming	For Aquaculture	Other places
Temperature	^oC	30	-	-
Smell		no	no	no
pH		6.5 – 8.5	6.5 – 8.5	6.5 – 8.5
DO	mg/L	≥ 4	≥ 5	≥ 4
BOD (5 day period)	mg/L	< 20	< 10	< 20
Suspended solids	mg/L	25	50	200
Asenic	mg/L	0.05	0.01	0.05
NH_3 (based on N)	mg/L	0.1	0.5	0.5
Cd	mg/L	0.005	0.005	0.01
Pb	mg/L	0.1	0.05	0.1
Cr (VI)	mg/L	0.05	0.05	0.05
Cr (III)	mg/L	0.1	0.1	0.2
Cloride	mg/L	-	0.01	-
Copper	mg/L	0.02	0.01	0.02
Floride	mg/L	1.5	1.5	1.5
Zinc	mg/L	0.1	0.01	0.1
Mn	mg/L	0.1	0.1	0.1
Ferric	mg/L	0.1	0.1	0.3
Hg	mg/L	0.005	0.005	0.01
Sulfide	mg/L	0.01	0.005	0.01
Cyanide	mg/L	0.01	0.01	0.02
Total phenol	mg/L	0.001	0.001	0.002
Oil foam/skin	mg/L	no	no	0.3
Oil suspension	mg/L	2	1	5
Chemicals used as pesticides	mg/L	0.05	0.01	0.05
Coliform	MPN/100mL	1000	1000	1000

The implementation of the Law and regulations has not been good yet, especially at local levels because there was a shortage in human resource as well as facilities for environmental assessment, mitigation and management of impacts, etc.

5 Current research and implementation and opportunities for collaboration

There have been 121 fisheries sector projects in Vietnam since 1982. About one-tenth of the projects are occurring of relevance to the TROPECA project (Table 6). Danida-SUMA and Hon Mun MPA projects are among potential collaborators.

Table 6. List of fisheries sector projects that may be relevant to TROPECA

Donor	Project Title	Project date	Commitment (US$' 000)	Grant/ Loan	Executing Agency (EA)	Implementing Agency (IA)	Co-implementing	Project Objectives	Location
ADB (RETA 5552)	Coastal and marine environmental management	.	.	.	.	MOSTE	.	Regional technical assistance involving Cambodia, Viet Nam and China. In Viet Nam the RETA is preparing investment projects . The Minh Hai component involves the preparation of an integrated environmental management plan, including mangrove reforestation and aquaculture.	Ha Long Bay, Minh Hai, Ho Chi Minh city
DANIDA	Fisheries Sector Programme Support (Fisheries SPS) (104.Vie.41)	TBA	41	Grant	MOFI	Provincial Fisheries Dept.	.	Environmentally and socially sustainable growth in the fisheries sector in line with the international standards	Nghe An; Ha Tinh; Bac Kan; Khanh Hoa; Quang Ninh
DANIDA	Coastal Aquaculture	TBA	1,000	Grant	.	.	.	Environmentally sustainable coastal aquaculture development in the North-Central Coastal region of Vietnam	Thanh Hoa; Nghe An; Thua Thien-Hue
DANIDA/WB	Viet Nam coastal wetlands protection and management development.	2000-ongoing	65,600	31,100 loan and 11,300 grant	DANIDA	MOSTE	.	Restore mangrove forests along the 470 km Mekong coast, giving a boost to aquaculture and improve the quality of live.	Tra Vinh, Soc Trang, Bac Lieu, Ca Mau

DANIDA	Support for brackish water and Marine Aquaculture (SUMA) (a component of the Fisheries SPS)	2000-2005	6,460	grant	DANIDA	MOFI	.	To strengthen the administration and management practices as required to supply marine aquactic products through environmentally and socially sustainable aquaculture development. Topics: Legislation, Aquaculture planning, Technology development, pilot community projects, Credit, HRD capacity building, Information collection and dissemination	On national level and 5 provinces, Quang Ninh, Ha Tinh, Nghe An, Khanh Hoa and Ca Mau
IUCN/GEF/ DANIDA	Hon Mun Marine Protected area Pilot Project	2001-2005	2,123	Grant	IUCN	MOFI	.	To conserve a representative example of internationally significant and threatened marine bio-diversity. To enable local island communities to improve their livelihoods and in partnership with other stakeholders to effectively protect and manage the marine bio-diversity at Hon Mun as a model for collaborative MPA management in Vietnam.	Khanh Hoa
MRC	Assessment Mekong Fisheries: Migration and Spawning and impact of Water Management	1997-2003	5,213	grant	MRC	MOFI, (RIA2)	.	covers 4 countries	Mekong Delta in Viet Nam
MRC	Management of the Reservoir Fisheries in the Mekong Basin, Phase II	2000 - 2004	4,455	grant	MRC	MOFI (RIA3)	.	Sustaninable co-management models for optimal fish production in reservoirs develop, implemented and disseminated in the Lower Mekong Basin (4 countries)	Central hihgland of Viet Nam (Daklak)
Netherlands	Vietnam-Netherlands Integrated Coastal Zone Management Project	2000-2003	2,000	grant	SNV	MOSTE	Doste	Assist in the establishment of the required institutional structures at national and provincial level for ICZM, expand institutional and professional capacity to apply ICZM, develop long term strategy and action plans; and initiate short term application of ICZM in three provinces	Thua Thien Hue, Nam Dinh, Baria-Vung

								through practical problem solving approaches.	Tau
PEMSEA	National Demonstration Site for Integrated Coastal Management at Danang	2000-2004	.	grant	PEMSEA	DOSTE, Danang	PPC Danang	Workshop on Integrated Coastal Management. PEMSEA's project design is based on two management frameworks, namely:integrated coastal management and risk assessment/risk management. Demonstration sites will be set up throughout the region to implement these two mechanisms	coastal lands and waters of Danang Municipal ity including Son Tra Peninsul a, Danang Bay, Son Tra coastal waters and their adjacent lands
SEAFDEC	Pilot project for semi-intensive culture of shrimp (to include conservation of mangrove friendly aquaculture) (SD/AQ99-CM03)	1999-2002	.	.	Seafdec	RIMP	MOFI	Project operated by RIMP staff to make a model for semi-intensive culture of shrimp to increase production and thus profitability, and at the sam time teach the fishfarmers how to conserve the resources, e.g mangroves	SEAFDE C
UNDP	Environmental Management in Coastal Aquaculture	2000	375	Grant	MOFI	Ria No.1	.	Environmentally sustainable coastal aquaculture development in the North-Central Coastal region of Vietnam	Thanh Hoa; Nghe An; Thua Thien-Hue

(Source : FAO Hanoi, Vietnam, http:www.fistenet.gov.vn)

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40.

TROPECA

Nutrient cycling in flooded production systems

Tropeca Working Paper No. 5

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May 2005

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Summary of key findings and conclusions

1. Aqueous N is more affected by management than sediment N

2. All management additions of N lead to slowly increasing sediment NO_3^-, which is vulnerable to leaching and might "spill over" downstream

3. Addition of mineral N fertilizer (NH_4^+, NO_3^-)
    - leads to a surface NO_2^- spike

4. Addition of active organic residue
    - depletes surface NO_3^-
    - leads to a surface NO_2^- spike

5. Exchange of "dirty" topwater (containing active organic residue)
    - leads to a surface NO_2^- spike

6. Exchange of "clean" topwater (containing no organic residue)
    - depletes surface NO_2^-

7. Aqueous P is highly buffered by equilibration with insoluble forms

8. Management additions of P are likely to have only short-term impact

9. P availability is largely determined by pH
    - management operations rarely attempt to influence pH

10. Rice plant roots create an oxic rhizosphere within a largely anoxic sediment
    - this enhances their ability to exploit sediment N (as NO_3^- instead of NH_4^+)

11. Rice plant roots acidify their rhizosphere by oxidising Fe^{2+}
    - this enhances their ability to exploit sediment P

Management implications

1. Use a holding tank to "clean" topwater exchange water so as to prevent a surface water NO_2^- spike

2. Use (appropriately-fertilized) rice plants to "mop up" excess N and P added to the sediment

Introduction

Nitrogen (N) and phosphorous (P) are essential components of all biomass. Limitations in the supply of these nutrients can determine the productivity of extensive biological production systems such as shrimp farms and rice fields. Oversupply, on the other hand, might lead to irreversible shifts in system function or – more likely – unwanted effects downstream.

In order to understand N and P cycling in flooded systems we need first to understand oxygen (O_2) supply and demand, and the changes in acidity (pH) dependent on them.

Oxygen

Oxidation of organic matter (here represented simplistically as CH_2O) is essential to life. It requires an electron-acceptor (oxidant) capable of reduction. The energetically-preferred sequence of electron-acceptors commonly found in agro-ecosystems is shown in Table 1.

Table 1 Oxidation of organic matter (CH_2O)

oxidant	**reaction**	**approximate stoichiometry**
oxygen (O_2)	respiration	$CH_2O + O_2 \rightarrow CO_2 + H_2O$
nitrate (NO_3^-)	denitrification	$5\ CH_2O + 4\ HNO_3 \rightarrow 2\ N_2 + 5\ CO_2 + 7\ H_2O$
iron^{III} ($Fe(OH)_3$)	iron reduction	$CH_2O + 4\ Fe(OH)_3 \rightarrow 4\ Fe(OH)_2 + CO_2 + 2\ H_2O$
sulphur^{IV} (H_2SO_4)	sulphate reduction	$2\ CH_2O + H_2SO_4 \rightarrow H_2S + 2\ CO_2 + 2\ H_2O$
OM (CH_2O)	methanogenesis	$CH_2O + CH_2O \rightarrow CH_4 + CO_2$

Where O_2 is available, NO_3^- is not generally oxidised; where NO_3^- is available, Fe^{III} is not generally oxidised; and so forth down the list. O_2, ubiquitous in the atmosphere at some 21% by volume, diffuses into the flooded system from the surface (z = 0) to depth (z) and is consumed at rate Q as it does so:

$$\frac{\partial}{\partial t}[O_2] = \frac{\partial}{\partial z}\left(D\frac{\partial}{\partial z}[O_2]\right) - Q \qquad [1]$$

where $[O_2]$ represents the local O_2 concentration and t time. Similar equations apply to the other oxidants (*cf* Arah & Kirk, 2000). The result is a stratified system with an oxic zone (in which O_2 is the primary oxidant) overlying a suboxic zone (in which NO_x is the primary oxidant) overlying an anoxic zone (in which reduction of Fe^{III} and SO_4^{2-} and methanogenesis predominate) (Ponnamperuma, 1972).

Nitrogen

N transformations in the oxic and suboxic zones are summarised in Table 2, in which the denitrification reaction is broken down into its component steps. N fixation in the oxic zone produces organic N at the expense of the oxidation of considerable amounts of CH_2O. Ammonium (NH_4^+) produced by mineralization of organic matter is oxidised in the oxic zone to nitrate (NO_3^-) which diffuses into the anoxic zone where it is sequentially denitrified to nitrite (NO_2^-), nitrous oxide (N_2O) and nitrogen gas (N_2). Soluble products diffuse back into the oxic zone, while the gases (N_2O and N_2) largely bypass that zone *via* ebullition to the atmosphere (Patrick & Tusneem, 1972).

Table 2 Major N transformations in flooded systems

reaction	**approximate stoichiometry**	**symbol**	**inhibitors**
oxic			
nitrogen fixation	$3\ CH_2O + 2\ N_2 + 7\ H_2O \rightarrow 4\ NH_4OH + 3\ CO_2$	J_{fix}	
nitrification	$2\ NH_4OH + 4\ O_2 \rightarrow 2\ HNO_3 + 4\ H_2O$	J_{nit}	NO_3^-
suboxic			
denitrification [1]	$CH_2O + 2\ HNO_3 \rightarrow 2\ HNO_2 + CO_2 + H_2O$	J^1_{denit}	O_2
denitrification [2]	$CH_2O + 4\ HNO_2 \rightarrow 2\ N_2O + CO_2 + 5\ H_2O$	J^2_{denit}	O_2, NO_3^-
denitrification [3]	$CH_2O + 2\ N_2O \rightarrow 2\ N_2 + CO_2 + H_2O$	J^3_{denit}	O_2, NO_3^-, N_2O

A simple (two-compartment) model of these processes is illustrated in Fig.1. In a typical shrimp-production system the oxic zone might represent the water and the suboxic zone the underlying sediment. The actual boundary between the two zones is fluid, but for our current purposes let it stand: *oxic* denotes water; *suboxic* denotes surface sediment. Transformations J (solid lines) are represented by dual-substrate kinetics subject to concentration-dependent inhibition where indicated in Table 2. For example:

$$J^2_{denit} = k^2_{denit}[CH_2O][NO_2^-]\left(\frac{I_{O2}}{I_{O2}+[O_2]}\right)\left(\frac{I_{NO3}}{I_{NO3}+[NO_3^-]}\right) \qquad [2]$$

where I_X is an inhibition constant for species X. Transfers F (dashed lines) obey Fick's Law:

$$F_{ij} = k_D([X]_i - [X]_j) \qquad [3]$$

where k_D is a transfer coefficient, i represents the source compartment and j the destination. Oxygen transformations (Table 1) and transfers (not shown in Fig.1) are handled similarly.

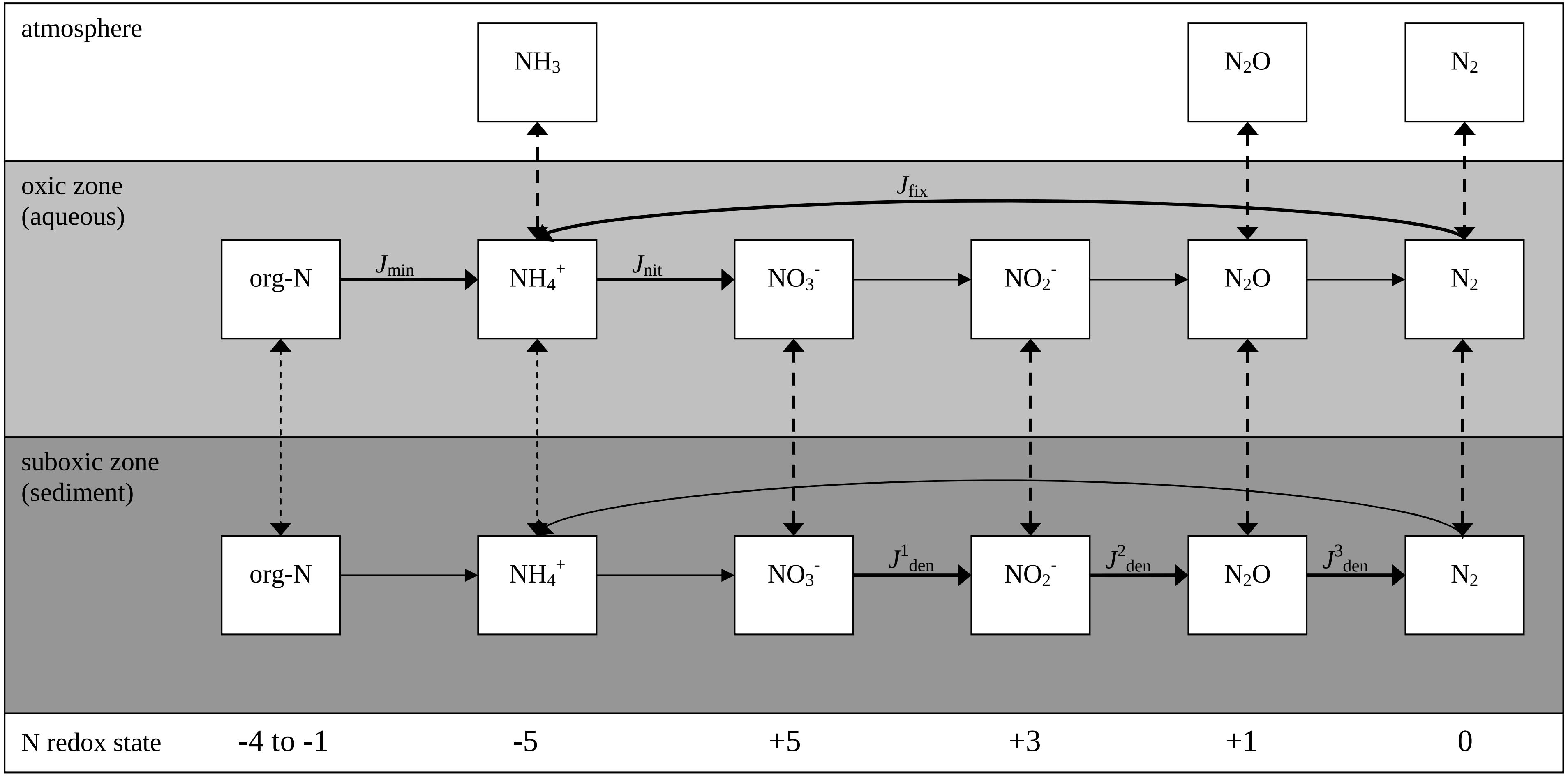

Fig.1 Nitrogen cycling in flooded soils and sediments. Boxes represent pools, solid lines transformations (fixation, mineralisation, nitrification, denitrification), dashed lines transfers (diffusion, ebullition, settling); bold arrows represent major fluxes.

Typical model output is illustrated in Fig.2. The dual-compartment system is subjected to three management operations, at times *t* = 25, 50 and 75 d. These are:

(i) *fertilizer* addition (equal amounts of NH_4^+ and NO_3^- added to the oxic zone);
(ii) topwater *exchange* (reoxygenation of the oxic zone coupled with addition of organic matter and inorganic N;
(iii) *residue* addition (active organic C and N added to the oxic zone).

The actual *numbers* illustrated in Fig.2 count for little – model parameters were not optimised against data. What matters instead is the *patterns* they show.

Oxygen and Active Organic C

Active organic C in the system declines continuously until restored by surface water exchange (which is assumed to bring with it a flush of organic matter) or direct input of residue, after which the process begins again. Where active C is abundant, O_2 consumption exceeds supply from the surface and the O_2 concentration declines; where active C is scarce, O_2 concentration rises. Topwater exchange restores the O_2 concentration of the surface zone; addition of mineral N fertilizer has little effect; addition of organic residue enhances O_2 consumption.

Ammonium and Nitrate

Mineral N fertilizer addition leads to spikes in the concentrations of NH_4^+ and NO_3^- in the oxic zone, the former declining more rapidly than the latter (as a consequence of volatilization and, more importantly, nitrification). Addition of organic residue leads to a bell-shaped NH_4^+ spike (mineralization) accompanied by a drop in surface zone NO_3^- (nitrification, diffusion to depth, denitrification). Topwater exchange leads to a mixture of the above two patterns. There is a gradual increase in sediment NO_3^- throughout.

Nitrite and Nitrous Oxide

All management operations lead to a marked spike in surface zone NO_2^-, followed after a few days by a lesser increase in N_2O. In every case this is due to denitrification in the sediment (of NO_3^- diffusing down from above) followed by upward diffusion of its products. Nitrite (NO_2^-) is toxic to shrimp (Chen & Chen, 1992). Simulations (not shown) in which topwater exchange does *not* bring with it an addition of active organic matter show no such spikes; instead there is a marked decline in surface zone NO_2^- and no subsequent increase in N_2O.

Nitrogen Uptake

Nitrogen uptake is here taken to be a simple function of the availability of NO_3^- and NH_4^+, weighted in the sediment by the relative immobility of NH_4^+ (which is adsorbed on clay particle surfaces). It is higher in the surface water than in the underlying sediment, and also more responsive there to management effects. In general, *all* management effects are more marked in the aqueous zone than in the sediment.

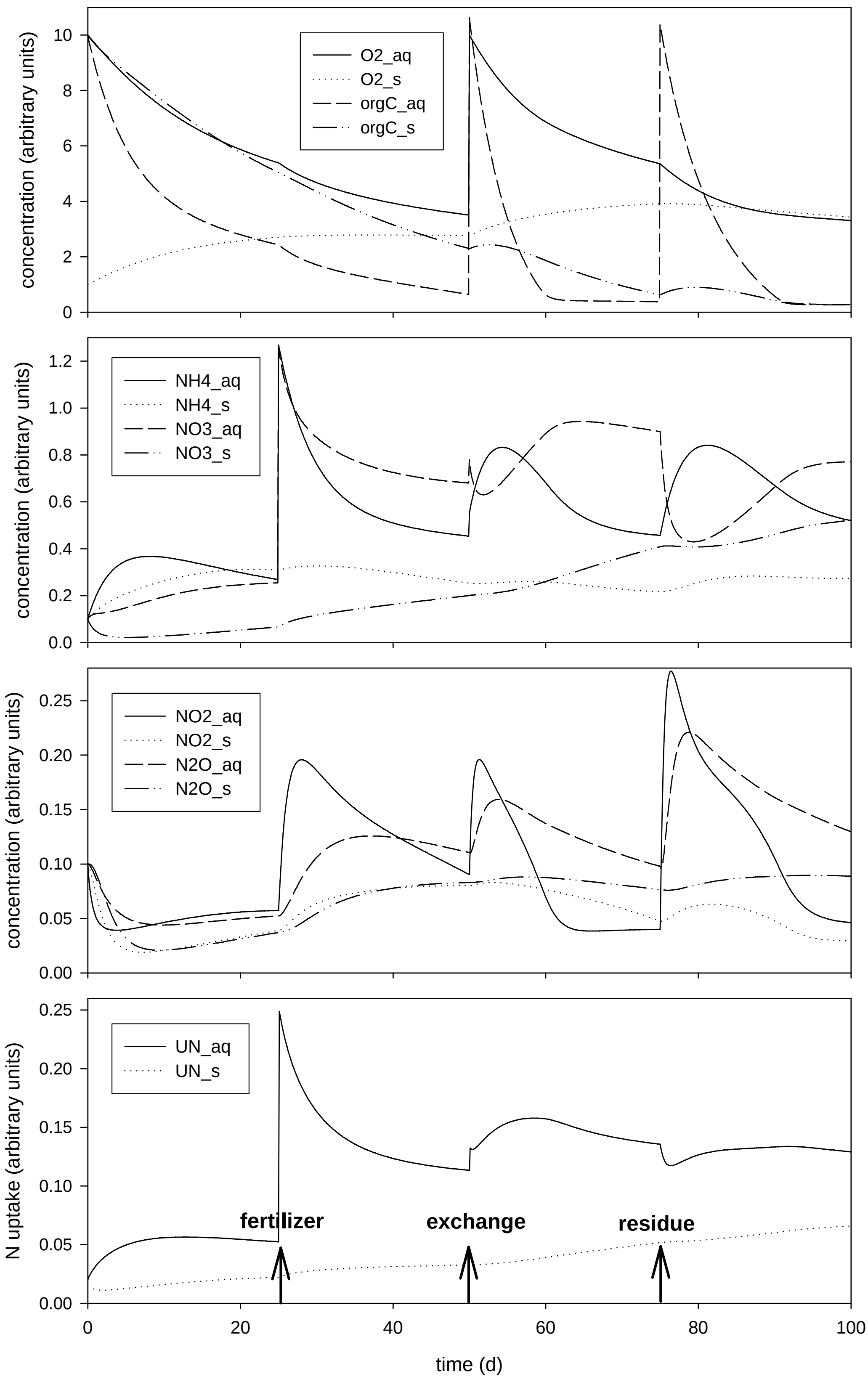

Fig.2 Output of dual-compartment N transformation model. Surface water (_aq) and sediment (_s) concentrations and potential N uptake rates; orgC denotes *active*, not *total* organic C, other symbols should be self-explanatory.

Rice roots

The simple model described and discussed above takes no account of a very important feature of rice roots: they have *aerenchyma* (Armstrong *et al*, 1991). These air-filled internal passages conduct O_2 from the surface through the roots to the root tips, where it escapes into the rhizosphere (the root zone).

The consequence is a (small) *oxic* rhizosphere, in which nitrification becomes a possibility (Fig.1). This may enable the plant to take advantage of the greater mobility of NO_3^- than NH_4^+ in a sediment, giving it a greater potential N uptake rate than suggested in Fig.2 (Kirk & Kronzucker, 2000).

Nitrification is not an unmixed blessing, however. It acidifies the rhizosphere (Table 2), further reducing the already limited NH_4^+ mobility as the concentration of HCO_3^- decreases (Kirk *et al*, 1994).

Phosphorous

Phosphorous exists in the biogeosphere overwhelmingly in the +6 redox state, as phosphate PO_4^{2-}. The changes in redox potential E_h which dominate O_2, CH_2O and N processes have no effect on the redox state of P (Schlesinger, 1991).

The availability of P to biomass depends on the balance between precipitation of soluble PO_4^{2-} to insoluble forms and the remobilisation of those forms. The major insoluble form is apatite [$Ca_5(PO_4)_3OH$] but PO_4^{2-} may also be immobilised by adsorption to, substitution in, or occlusion by many other mineral forms. As an example of the complexity this gives rise to, dissolution of Fe-oxyhydroxide minerals in anoxic zones (Table 1) may result in the release of co-precipitated PO_4^{2-} (Bostrom *et al*, 1988).

Nevertheless, the various processes governing precipitation and remobilisation are probably most importantly influenced by acidity (pH). Below ~pH 5.7, increasing pH is likely to increase available P as it inhibits co-precipitation with Al_3^+; above ~pH 5.7, increasing pH reduces available P by promoting precipitation of $Ca_5(PO_4)_3F$ (Lindsay & Vlek, 1977).

A stylised version of the many processes at work is illustrated in Fig.3. The background pH of the (brackish) aqueous zone is taken to be somewhere around pH 8, the range pH 6-9, greatly simplifying the above account: decreasing pH enhances P availability, increasing pH reduces it.

There is no need to create a model of so simple a system: it is obvious that increasing pH (alkalinity) will tend to decrease P availability (*ie* the soluble P pool) in the surface water, while decreasing pH (acidity) will tend to increase it. The (small) labile P pool acts to buffer the system against any short-term change, the (large) stabilised P pool against longer-term shifts. Management additions to the system will generally be of organic or soluble P, which will be converted relatively quickly into the labile forms, and more gradually into the stabilised ones. The consequences of undersupply and oversupply will both take considerable time to be manifest in the background levels of the indicator pools (organic and soluble P).

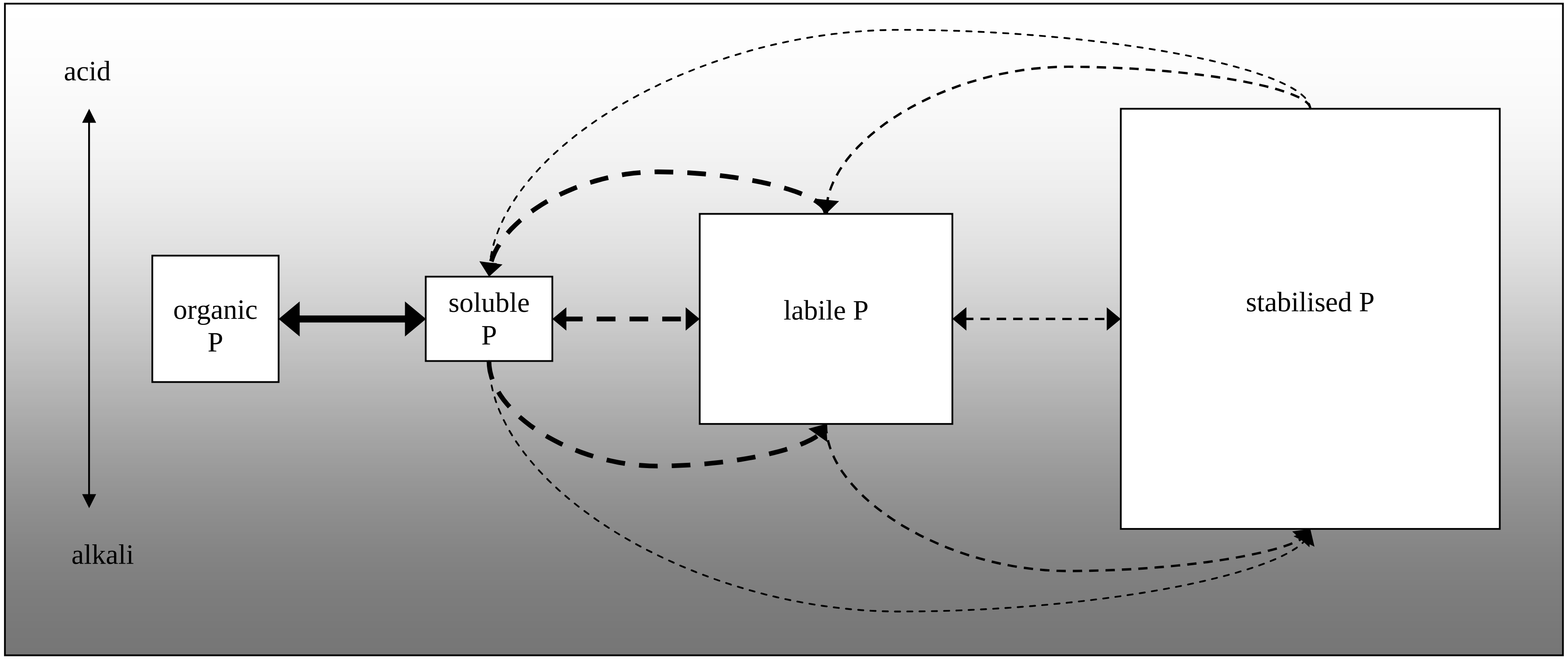

Fig.3 Phosphorous cycling in flooded soils and sediments (simplified). Boxes represent pools; solid lines represent biologically-mediated fluxes; dashed lines represent largely inorganic fluxes. Transformations above the midline are favoured by increasing acidity, those below it by increasing alkalinity (pH).

Acidity (pH)

Nitrification increases acidity (reduces pH) and thereby increases P availability, as does denitrification to a lesser extent (Table 2). Sulphate reduction (Table 1) and N fixation (Table 2) both increase alkalinity (increase pH) and might therefore be expected to have the opposite effect.

One important reaction is missing. Where an oxic rhizosphere meets a surrounding anoxic zone, oxidation of reduced iron Fe^{2+} occurs:

$$O_2 + 4\ Fe^{2+} + 10\ H_2O \rightarrow 4\ Fe(OH)_3 + 8\ H^+ \qquad [4]$$

This greatly acidifies the contact zone and promotes P mobility and uptake (Kirk *et al*, 1994; Saleque & Kirk, 1995).

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Bullet points

- Aqueous N is more affected by management than sediment N
- All management additions of N lead to slowly increasing sediment NO_3^-, which is vulnerable to leaching and might “spill over” downstream

- Addition of mineral N fertilizer (NH_4^+, NO_3^-)
  - leads to a surface NO_2^- spike
- Addition of active organic residue
  - depletes surface NO_3^-
  - leads to a surface NO_2^- spike
- Exchange of “dirty” topwater (containing active organic residue)
  - leads to a surface NO_2^- spike
- Exchange of “clean” topwater (containing no organic residue)
  - depletes surface NO_2^-

- Aqueous P is highly buffered by equilibration with insoluble forms
- Management additions of P are likely to have only short-term impact

- P availability is largely determined by pH
  - management operations rarely attempt to influence pH

- Rice plant roots create an oxic rhizosphere within a largely anoxic sediment
  - this enhances their ability to exploit sediment N (as NO_3^- instead of NH_4^+)
- Rice plant roots acidify their rhizosphere by oxidising Fe^{2+}
  - this enhances their ability to exploit sediment P

Bottom line

- Use a holding tank to “clean” topwater exchange water so as to prevent a surface water NO_2^- spike

- Use (appropriately-fertilized) rice plants to “mop up” excess N and P added to the sediment