A NITROGEN-BASED ASSESSMENT OF AQUACULTURE:
Shrimp Farming In Northwest Mexico

by

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ABSTRACT

Understanding the interactions between aquaculture and the environment requires the consideration of production inputs and outputs, ecological processes, as well as the identification and application of quantitative approaches. A conceptual and analytical framework to assess shrimp farming systems was developed based on nitrogen (N); nitrogen is an important element appearing in those interactions both as a nutrient and as a toxicant. The assessment relied on Nitrogen Productivity (NP), an indicator developed by merging Nitrogen Efficiency (NE) and Ammonia Assimilative Capacity (AAC). These indicators, calculated through model simulations, were used in the screening for optimal operating conditions. The simulation model was based on N fluxes, the processes influencing shrimp growth, and the production and loss of ammonia. The model simulated changes in stocking density (*Do*) and water exchange rate *(W)*, two of the main management variables associated with the level of intensification in aquaculture. The model predictions were compared to a typical, semi-intensive shrimp farming pond in Northwest Mexico.

The analysis indicated that optimal operating conditions for the Mexican case were attained by increasing *Do* towards 30 shrimp$\cdot$m $^{-2}$, if survival was maximized, and by using no water exchange. This management scheme conveyed a significant reduction (>90%) in water usage, but increased N deposition in sediments. Simulated shrimp farming also performed well only at $W > 0.7 \cdot d^{-1}$; such high rates may have significant economic and biophysical costs that merit analysis. Moderate increments in W (to 0.12 – $0.13 \cdot d^{-1}$) resulted in a significant level (95% or higher) of effluent N in dissolved form, with a higher potential for its reutilization, but increased ammonia levels in the pond. Formulated feeds represented more than 75% of total N inputs for *Do* higher than 15 shrimp·m $^{-2}$, while N from fixation and fertilizer contributed importantly only on the extensive side (*Do* < 5 shrimp·m $^{-2}$) of the farming spectrum. Phytoplankton growth and sedimentation rates represented important processes influencing ammonia levels in the pond.

This nitrogen-based analysis identified dominant processes and the effects of management practices associated with the intensification of shrimp farming. Nitrogen-based tools appear to have great promise for analyzing the biophysical and economic aspects of aquaculture.

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ABBREVIATIONS

AAC = ammonia assimilative capacity

ADF = air-dry feed

CP = crude protein

DMF = dry matter in feed

DN = digested nitrogen

Do = stocking density

EN = nitrogen excreted as ammonia

FCR = food conversion ratio

fN = nitrogen in faeces

FN = nitrogen in feed

FP = protein in feed

FR = feeding rate

IN = ingested nitrogen

ISB = individual shrimp biomass

m = mortality rate

N = nitrogen

n =nitrification rate

NE = nitrogen efficiency

NEBiol = biological approach to NE

NEEcon = economic approach to NE

NO = nitrite and nitrate

NP = nitrogen productivity

ODF = oven-dry feed

PD = protein digestibility

PN = nitrogen in protein

Q = aggregate outputs

RN = nitrogen retained in biomass

s =sedimentation rate

SB = shrimp biomass

SD = surviving shrimp density

SN = nitrogen in shrimp biomass

TAN = total ammonia nitrogen

TFP = total factor productivity

UFN = uneaten feed nitrogen

UN = undigested nitrogen

v = volatilization rate

W = water exchange rate

X = aggregate inputs

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DEDICATION

In memory of my father, Carlos Gomez Solorzano, to whom I owe so much

CHAPTER 1. INTRODUCTION

1.1 Thesis rationale

Evidence in the literature supports the need for an alternative, quantitative assessment of the performance of aquacultural systems; more integrated and comprehensive, within the broad context of sustainability (Barton and Staniford 1998; Beveridge, Ross, and Kelly 1994; Carvalho and Clarke 1998; Lightfoot et al. 1993a; Neiland, Soley, and Baron 1997). This need has been expressed strongly for an approach that considers the importance of ecological processes, not solely on increasing yields, but with a focus on the environmental interactions from the use of inputs and the generation of wastes (Folke and Kautsky 1989; Odum 1989).

Food production systems such as aquaculture[1] are commonly assessed by focusing solely on the production of useful outputs, whether based on biophysical (e.g., kg) or financial indicators (e.g., gross profits). Thus, official statistics are often reported as output per unit area, volume or enterprise e.g., kg per ha or kg per farm. Productivity is the basic measure of efficiency and is a conventional indicator of system performance. It is based on the calculation of a ratio of food produced per area or yield, and the evaluation of changes of this ratio under different production technologies, whether in biophysical or monetary units (Giampietro, Cerretelli, and Pimentel 1992).

Conventional comparison between and among aquacultural systems is based on yield,

[1] Aquaculture is the farming of aquatic plants and animals.

suggesting that more intensive production practices (i.e., those with higher yields) are better (Davies and Afshar 1993). More intensive practices are also assumed to maximize financial returns per hectare. Although the total output can be increased through intensification, overall profitability does not always result (Lee and Wickins 1992). There is a high correlation between intensification of aquaculture and a higher degree of environmental interactions, some of which may lead to negative environmental consequences (Beveridge, Phillips, and Macintosh 1997; Folke and Kautsky 1989). Increasing concern about those consequences questions the effectiveness of the production-led approach to aquaculture assessment within the broad context of sustainability.

Aquaculture has recently followed the trend of older economic systems, such as agriculture, of intensifying management practices to increase yields (Bailey 1997). The trend towards more intensive management, and the capability of farmers to develop it in response to different factors, although not well studied in aquaculture, is well documented in agriculture (Lynam and Herdt 1989).

The intensification of aquaculture has allowed the sector in part to maintain a continuous growth and to be considered the fastest growing food production sector in the world in the last decade (FAO 1999). Aquaculture's annual growth rate of approximately 10.9% (excluding aquatic plant production) has surpassed chicken (5.3%), pig (3.4%), capture fisheries (1.6%), mutton and lamb (1.4%) and beef and veal (0.9%) production (Tacon and Grainger 1999). Aquacultural production increased from about 1 million tonnes in the late 1970`s to 36 million tonnes in 1997 (Fig. 1-1), including aquatic plants, with a value of US$50.36 billion. The volume produced in 1997 represented around 25% of global fish supplies and 29% of fish for human consumption (FAO 1999). Among the aquaculture subsectors shrimp farming is considered the fastest growing, contributing now 26% of the total shrimp supply in contrast to a contribution of 1% a decade ago (FAO 1999).

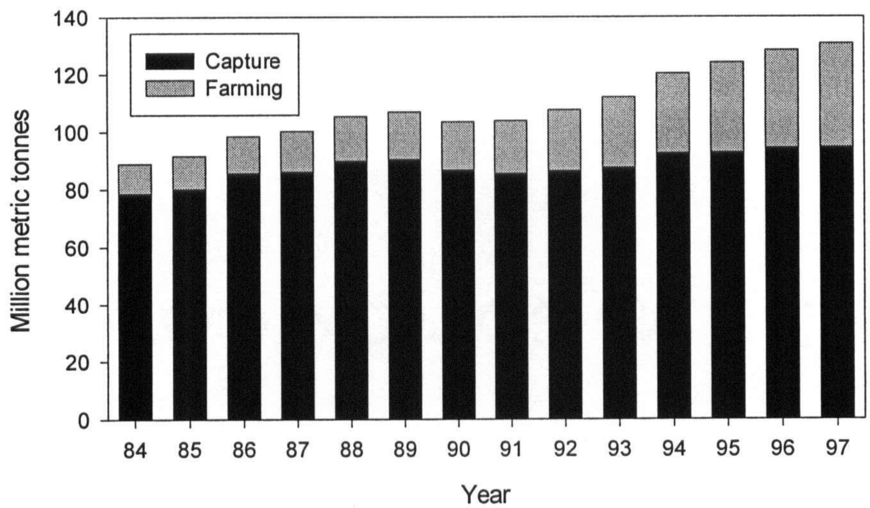

Fig. 1-1. Global fish supply (FAO 1999)

Aquaculture may play a crucial role in the future global supply of fish (Bailey 1997; Barton and Staniford 1998). It is expected that aquaculture growth will continue to compensate, not only for the stagnation in catches in the fisheries sector (Fig. 1-1), but also to supply the rapidly increasing demand for a greater diversity of aquatic products. This perception is well embedded in the political agenda of various countries around the world. The World Commission on Environment and Development report of 1987 (p. 138) stated that " the expansion of aquaculture should be given high priority in developing and developed countries" (WCED 1987). Projections indicate that the demand for fish for human consumption may reach 120 million tonnes by 2010, and that 40% of the volume and 50% of the value, of global fish supply, will come from aquaculture (Barton and Staniford 1998; FAO 1999).

Production from aquaculture is expected to increase mainly through intensification practices (i.e., higher yields) as availability of land and suitable coastal areas is decreasing rapidly and expansion of aquaculture into the open ocean has yet to prove technically and economically feasible (Barton and Staniford 1998). As a comparison, the intensification process in agriculture resulted in increased yields over the last 50 years (Matson et al. 1997), and almost a doubling of global agricultural production in the last 30 years (Cassman 1999). More than 90% of that growth was derived from yield increases and less than 10% from area expansion (Goodland and Daly 1996; Tilman 1999).

Concern has developed however over the environmental costs that tend to increase in parallel with the intensification process of aquaculture (Beveridge, Phillips, and Macintosh 1997; Folke and Kautsky 1989). Although extensive methods currently account for most aquacultural production, it is semi-intensive and intensive production practices for species such as shrimp and salmon that receives more attention, particularly for the associated environmental impacts. The concern focuses mainly on whether intensive practices can be sustained due to environmental costs associated with the exploitation and the potential deterioration of the natural resource base on which aquaculture depends (Naylor et al. 1998; Stewart 1995). Intensive aquacultural production practices, such as those for shrimp and salmon, require a considerable amount of imports mainly in the form of feeds and export a large amount of nutrients to the surrounding environment (Folke and Kautsky 1992; Stewart 1995). Similar concerns have been expressed over the intensification process of agriculture (Geng, Hess, and Auburn 1990; Matson et al. 1997; Vavra 1996). But the intensification of other sectors such as agriculture, tourism and industry may also lead to pollution affecting aquacultural production.

There is increasing awareness that aquacultural systems depend upon ecosystem integrity and have limits, particularly with regard to assimilation of waste products (Beveridge, Phillips, and Macintosh 1997). Pollution from wastes may impair the production process through water quality deterioration, which is regarded as a major problem in the more intensive forms of aquaculture (Talbot and Hole 1994). Stress of the cultured organisms, disease and production collapses have been closely associated with water quality deterioration in semi-intensive and intensive shrimp farming (Bailey 1997). Pollution from aquaculture may also affect the integrity of adjacent ecosystems and create conflicts with other resource users such as agriculture, tourism, and fisheries.

As a result, aquaculture is perceived in many parts of the world as a threat to ecological and socioeconomic systems (Bailey 1997) and the industry has been placed under increasing scrutiny (Clay 1997). Although aquaculture is largely an unregulated activity in most of the world, public opinion and political pressure have led to strict regulations or moratoria placed in certain areas (e.g., shrimp farming in India and salmon farming in

British Columbia). Restrictions imposed may be major constraints on further aquaculture developments due to higher production costs in trying to conform to government regulations concerning, for example, water quality. The exaggeration of the severity of aquaculture impacts has led to politically based measures in some parts that may restrict the potential for aquaculture (Muir 1996). Therefore, regulations need to be based on a clear understanding of the interactions between aquacultural systems and the adjacent environment.

Analytical approaches to assess aquacultural systems within the broad context of sustainability (Davies and Afshar 1993; Stewart 1995) are scarce, and consequently there are few insights into the issue. Most of the work has been descriptive and therefore there is a lack of methodologies and indicators.

Given the strong analogies of aquaculture and agriculture, I considered it important to examine agricultural studies and to translate that knowledge into the aquacultural context. Some authors (Barg and Phillips 1997; Barton and Staniford 1998; Mearns 1997; Shell 1991; Upton 1997) have expressed the potential value of this approach. A review of the larger body of analytical literature on agricultural systems indicated that there is a consensus on productivity, if appropriately measured, as a surrogate of sustainability in food production systems (Herdt and Steiner 1995; Rayner and Welham 1995). That review also indicated that the multiple input-output approaches to productivity measurement, such as Total Factor Productivity (TFP) or Total Resource Productivity (TRP) may represent appropriate measurements. In contrast to traditional productivity assessments, the multiple input-output approach considers more inputs and outputs, whose use and production are the source of the environmental interactions. However, those approaches have relied on money as a currency (i.e., monetary-based) and have had difficulty in assessing the consequences of wastes (i.e., externalities), such as pollution for example. In contrast, biophysical-based analyses have been regarded as more appropriate in the analysis of broad sustainability, resembling more a systems approach, in contrast to the more marginal approach of monetary-based analyses (Rees and Wackernagel 1999; Wackernagel 1999).

Nitrogen has been extensively utilized as a currency in biophysical analyses to assess agricultural production (Aldrich 1980; Brouwer 1998; Dou et al. 1998; van Eerdt and Fong 1998; Watson and Atkinson 1999). The measurement of nitrogen flows in agriculture is regarded as a good indicator of system performance (Magdoff, Lanyon, and Liebhardt 1997). In contrast, nitrogen-based analyses of aquaculture are very limited, in spite of the reality that nitrogen seems of higher relevance in aquacultural assessment as a result of the more important physiological implications on aquatic organisms.

Nitrogen is an important nutrient element and toxicant, appearing in important interactions between aquacultural production and the environment (Handy and Poxton 1993; Kibria et al. 1998). Nitrogen fluxes in aquacultural systems link the farm and the adjacent environment through feed, fertilizer, water inflow and outflow, harvest, fixation and nutrient losses to atmosphere and sediments.

The intensification process of aquaculture is characterized by an increased amount of nitrogen flowing into and from the production system, which gives rise to stronger environmental interactions. Some of those interactions result from the production of various nitrogenous compounds, which influence most importantly the water quality (arguably the most important resource in aquaculture) and hence the aquacultural production processes. Those interactions through nitrogen appear to be more important in coastal areas, which are mostly nitrogen-limited and where most semi-intensive and intensive aquaculture occurs (e.g., shrimp and salmon farming). Besides, these areas often correspond to important agricultural regions (e.g., Gulf of California) from which considerable amounts of excess nitrogen are received (Paez-Osuna, Guerrero-Galvan, and Ruiz-Fernandez 1999)

The most common indicator to assess nitrogen use in agriculture is nitrogen use efficiency (NE), or the ratio of nitrogen retained in harvest over nitrogen inputs. Although this indicator captures the capacity of the organisms to assimilate nitrogen, it does not reflect the capacity of the environment to assimilate nitrogenous wastes. The assimilative processes (i.e., assimilative capacity) of the environment are important as they influence considerably the water quality of aquacultural systems. The more intensive forms of aquaculture may produce a higher proportion of nitrogenous wastes that may affect negatively the production process through the deterioration of water quality. Therefore it appears to be of high relevance to consider the capacity of the environment to assimilate nitrogenous metabolites within a nitrogen-based analysis of aquaculture.

Nitrogen fluxes also have important financial implications in aquacultural systems. Feed represents one of the highest costs of production for semi-intensive and intensive aquaculture (e.g., shrimp and salmon), mainly due to the inclusion of fishmeal and soybean meal, which have a high protein, high nitrogen content (Bowen 1987; Hargreaves 1998).

The consideration of the capacity of the organisms to retain nitrogen, the production of nitrogenous metabolites and their effect on water quality, and the capacity of the environment to assimilate those nitrogenous wastes and thus support aquacultural production would capture the long-term productive potential of aquaculture. Such an approach, in contrast to the traditional productivity assessment, would consider various inputs and outputs, whose use and production are the source of environmental interactions. Therefore, the use of nitrogen as a currency in a multiple input-output approach to productivity measurement may integrate more appropriately the ecological and economic imperatives of aquaculture.

Such an approach may be a useful tool to assess the benefits and costs of, for example, the development and intensification of shrimp farming in Northwest Mexico given the considerable potential for growth and the lack of, and need for, regulatory policies.

1.2 Thesis objectives

The thesis goal is to develop a conceptual and analytical framework for a comprehensive approach to aquacultural activities that integrates and synthesizes the ecological considerations of aquaculture and establishes clear links to the economic and social dimensions. This will allow the assessment of aquacultural practices (production) as an integral activity of longer-term economic development, congruent with the maintenance of environmental integrity (ecological processes) and social imperatives.

The thesis concentrates on the biophysical dimension, as I consider the understanding of ecological processes and the maintenance of ecological integrity as the most relevant aspects to sustain aquacultural production in the long-term. Without maintenance of ecological processes there can be no aquaculture.

The study aims to integrate production and environmental interactions in order to assess aquacultural systems in a more comprehensive manner, drawing on the multiple input-output approach to assess productivity and the use of nitrogen as a currency linking the farm and the environment. Therefore, I want to move from a production focus towards a linkage of aquacultural production and ecological imperatives.

In developing the dissertation I focus on shrimp farming as it incorporates a broad range and intensity of biophysical and socioeconomic interactions. Shrimp farming is arguably the most controversial aquacultural subsector in terms of sustainability. To illustrate the application of my model I use a typical, semi-intensive case of shrimp farming in Northwest Mexico. However, I intend the application of the model to be as generic as possible.

To pursue the goal the thesis has the following objectives:

1. To review the concept of sustainability in food production systems with a focus on aquaculture and the efficacy of current approaches to its assessment;

2. To review the main processes of aquacultural systems and the environmental interactions associated with the intensification process of aquaculture;

3. To develop a conceptual and analytical, proactive framework to assess the long-term productive potential of shrimp farming systems based on the importance of nitrogen in ecological processes and socioeconomic considerations;

4. To understand, through the model, the origin and fate of nitrogen that enters shrimp farming systems at various levels of intensification and influences shrimp production, and the water quality in the pond as well as the potential impact of the effluent;

5. To illustrate the application of my approach using a typical, semi-intensive case of shrimp farming in Northwest Mexico, and to analyze through model predictions the implications of its potential intensification; and

6. To draw conclusions and discuss the suitability of implementing the approach in a generic sense.

1.3 Thesis organization.

Chapter 1 contains the rationale, purpose, and significance of the thesis. Chapter 2 reviews the concept of sustainability in food production systems, the need to view aquaculture within the broad context of sustainability, and the complexity of the issue. This Chapter contains also the conceptual framework of the thesis. In Chapter 3, I introduce the case study and I review the intensification trend and environmental interactions of aquaculture, with a main focus on the interactions among nitrogen fluxes, aquacultural production and water quality in shrimp farming. In Chapter 4, I present the analytical framework and the tools utilized for the analysis. Chapter 5 contains the results pertinent to the purpose of the work, their discussion and the main contributions of the study. Chapter 6 includes a summary of the main conclusions I can draw from this research.

1.4 Thesis significance

I expect my thesis to contribute importantly in the form of a novel, more effective, and alternative tool for analyzing and managing the biophysical and economic aspects of aquaculture within the broad context of sustainability.

The value of such an approach would be the operational application to the assessment of the long-term productive potential of aquacultural systems in the form of a Nitrogen Productivity indicator. A nitrogen-based assessment of productivity would allow: (1) utilizing nitrogen-rich resources more effectively at the farm level, (2) protecting aquaculture from adverse environmental problems caused by nitrogenous wastes from other sectors and from aquaculture itself, and (3) protecting the environment from the potential adverse effects of nitrogenous metabolites from aquaculture. I expect this approach to contribute importantly to management decisions and regulatory policies, at both private and public levels, that can support an appropriate development of shrimp farming, and other forms of aquaculture in Mexico and elsewhere.

CHAPTER 2. ASSESSMENT OF AQUACULTURE

In Chapter 2, I review the concept and complexity of sustainability in food production systems, and the importance for aquaculture of translating knowledge from agriculture, an analogous sector with a longer experience. Also in this Chapter I propose to transfer the concept of multiple input-output productivity from agricultural systems to the assessment of aquaculture, and the use of nitrogen as an appropriate currency.

2.1 Sustainability: the concept and the complexity

Sustainability as a major goal for development was recognized by the Brundtland Commission (WCED 1987) and the United Nations Commission for Environment and Development (UNCED 1993). This resulted in a series of commitments from various countries to view developments such as aquaculture within the sustainability context. Therefore, sustainability has become recently an important focus for management and development in aquaculture, and there is increasing interest to examine aquacultural systems accordingly (Beveridge, Phillips, and Macintosh 1997; Stewart 1995).

The interest in more sustainable production practices arises mainly from the concern over the perceived degradation of the environment associated with the intensification of aquaculture (Kelly 1996). Intensive farming of shrimp, for example, is considered inherently unsustainable due to the strong environmental interactions through input utilization and undesired output production (i.e., wastes) which has resulted in some environmental degradation (Greenpeace 1997; Gujja and Finger-Stich 1996; Lynam and

Herdt 1989; Naylor et al. 1998). Similarly, it has been suggested that intensive agricultural production is not sustainable on a long term, neither on economic or environmental grounds (Neher 1992; Vavra 1996). Therefore, there is the need to assess the performance of aquacultural systems within the context of sustainability.

However, the concept of sustainability in food production systems is very complex. Ecosystems are hierarchical levels of organization where processes operate at different scales, and those at lower levels are constrained by those at higher levels (Odum 1989). Systems are embedded within others and their interconnectedness makes it difficult to specify sustainability as a property of a particular system (Herdt and Steiner 1995). Food production systems such as aquaculture are linked to natural systems at different levels, exchanging energy, matter and information with the environment and characterized by non-linearity, disorder and unpredictability. (Conway 1987; Fresco and Kroonenberg 1992; Muir 1996). The lack of agreement on the definition and meaning of sustainability, and of methodologies and indicators to measure it, reflects the complexity of the concept (Farrell and Hart 1998; Vavra 1996).

In spite of the complexity, sustainability is perceived as increasingly important in the social and economic management of production systems, particularly those that depend on environmental resources such as aquaculture (Muir 1996; Spedding 1995). Therefore it is important to develop certain criteria and indicators against which the performance of an aquacultural system can be assessed (Barg and Phillips 1997; CGIAR 1991).

There is general consensus that there are three overlapping dimensions of sustainability in aquacultural systems. Aquaculture is an *economic* activity that serves a *social* purpose, and its production depends on *biological* processes that result in *physical* changes. Therefore, three main dimensions are represented: the biophysical, the economic and the social, all of them highly interrelated. Looking at them independently, although recognizing that they are interrelated, each dimension may be affected by different factors and to varying degrees of impact. The biophysical dimension may be affected by the quantity of output, which depends on the physical quantity of inputs and the biological growth process. The economic dimension may be affected by the relevant prices for inputs and outputs, which determine the profitability of an enterprise (i.e., the difference between revenues and costs). The social dimension may be affected by the capacity of aquacultural systems to support the hierarchy of surrounding human communities. This appears to be the most difficult parameter to measure as there are few, if any, accepted methods to assess social impact.

2.2 Need to look to agriculture.

Most studies and analyses of sustainability in food production systems have been performed within agriculture. This is understandable given the higher importance of agriculture by the amount of food supplied, the area under cultivation and the longevity of the sector when compared to aquaculture. Although aquaculture has been practiced for a long time, it is considered a “recent” or new industry (OECD 1989). Consequently, analyses of sustainability of aquacultural systems are scarce and in general the literature addresses the issue in a descriptive manner. Only a few have attempted a more analytical approach (Davies and Afshar 1993; Stewart 1995).

Aquaculture has a strong similarity to agriculture and it is often difficult to separate them (Barton and Staniford 1998; Edeson 1996; Howarth 1990; Leach 1976). The dependence on natural resources and services of the aquacultural sector, its intensification process and associated environmental interactions are remarkably similar to the agricultural sector. Shrimp farming, for example, has environmental, economic and social interactions similar to the production of other monocrops, such as sugar, cotton or cattle (Barraclough and Finger-Stitch 1996; Neiland, Soley, and Baron 1997).

Given the strong analogies, I considered it important to review more deeply agricultural analyses in order to extrapolate and apply appropriate concepts and approaches to aquaculture. The literature acknowledges the importance for aquaculture of learning from the older agricultural sector (Barg and Phillips 1997; Barton and Staniford 1998; Mearns 1997; Shell 1991; Upton 1997).

2.2.1 Consensus on productivity as an indicator.

There is no agreement on the definition of sustainability within the agricultural sector and consequently there are no accepted indicators or methodologies to assess the sustainability of agricultural enterprises. However, there is a broad consensus that the productivity of agricultural systems should be maintained in the long term. Many of the sustainability definitions within agriculture converge by considering directly or indirectly productivity and the importance of maintaining it over time.

Examples of some definitions are given below:

"Sustainability is the ability of a system to maintain productivity in spite of a major disturbance such as caused by intensive stress or a large perturbation"(Conway 1985). "Sustainability is the capacity to maintain output at a level approximately equal to or greater than its historical average, with the approximation determined by its historical level of variability" (Lynam and Herdt 1989). "A sustainable system is that which achieves production combined with conservation of the resources on which that production depends, thereby permitting the maintenance of productivity" (Young 1989). "Sustainability refers to the ability of an agroecosystem to maintain production through time in the face of long-term ecological constraints and socioeconomic pressures" (Altieri 1987).

It is implied therefore that if productivity is assessed adequately, it may constitute an appropriate indicator of the behaviour of agricultural systems within the context of sustainability. Due to the mentioned similarities with agriculture, these concepts may be extrapolated to aquacultural systems. Still, like all analogies, those between agriculture and aquaculture "break" given the diversity of systems and production methods utilized. Important biophysical differences between agriculture and aquaculture with regard to animal production are as follows (From Shell 1991):

1. The aquatic environment is in general more restrictive and chemically more variable than the terrestrial;

2. Although most agricultural and aquacultural production relies on herbivores and omnivores, intensive aquaculture involves more the production of carnivores requiring a higher proportion of animal protein (i.e., fishmeal) with a high nitrogen content;

3. With some exceptions, uneaten feed in aquaculture cannot be recovered and decomposes; farmed organisms have to compete for oxygen with microorganisms that decompose wastes; and

4. Although metabolic waste decomposes in the ground in agriculture and there is little problem of waste toxicity, wastes are released into the water in aquaculture. A high proportion of nitrogenous waste enters the water column as ammonia, which may be highly toxic to aquatic organisms.

The continuous increase in inputs utilized in most semi-intensive and intensive aquacultural systems normally increases yields, but it may offset reductions in the natural or environmental productive capacity. Productivity gains that occur at the cost of degradation of the natural environment, for example water quality, may lead to a fall in the productivity of aquacultural systems. This will imply that incremental production responses to input may be more difficult in the future. Therefore, a system which requires increasing inputs to maintain constant outputs over time may be regarded as not sustainable, and it will collapse in the long term on biophysical, economic or social grounds (Herdt and Steiner 1995). An important issue therefore is not the assessment of productivity change *per se*, but to determine whether productivity gains occur at the cost of ecosystem degradation.

2.2.2 Productivity assessment: partial and total.

Partial productivity, or the ratio of output to a single input, is the most common measure of productivity within agricultural systems. Such is the case of yield, which most often is the partial productivity of land input. Similarly, aquacultural productivity is measured by yield, which is commonly expressed as biomass production per hectare. The problem with partial productivity is that it considers only a subset of inputs, such as land, for which it is considered a misleading measurement of productive efficiency (Geng, Hess, and Auburn 1990).

An alternative measurement of productivity is the multiple input-output approach[2], such as total factor productivity (TFP), or total resource productivity (TRP) which consider multiple inputs and outputs of the production process (Gollop and Swinand 1998; Rayner and Welham 1995). Such approaches measure an index of total outputs relative to an index of total inputs in a production period. For example, TFP is a ratio of an index of aggregate output (Q) to aggregate inputs (X), where TFP = Q/X.

In comparison to partial productivity (i.e., yield), multiple input-output productivity recognizes the use of multiple inputs. However, as usually measured, it does not

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[2] Multiple input-output productivity is regarded as "the ratio of an output quantity index to an input quantity index" This renders productivity to be a relative concept, not an absolute one. This definition is in contrast to biological productivity, defined as the dry weight of biomass in an area per unit of time.

consider inputs and outputs that are external to the decision-making process (i.e., it ignores externalities). Output is generally regarded as the result of the production process that is useful; but very often production results in undesired outputs (i.e., wastes). Those external outputs of one system may become external inputs to another, or to itself.

By comparing the performance of the system to a base period a change in productivity is obtained, which renders it a relative measurement. This is in contrast to the absolute measurement, concerned with obtaining the maximum amount of output in period t given an amount of input in period t.

2.2.3 The problem of monetary-based analyses

Relevant market prices are commonly used as weights to calculate the output and input indexes, through various arithmetic and geometric functional forms (Diewert 1992). Therefore, the assessment of multiple input-output productivity in both agricultural and aquacultural systems has relied on money as the dominant currency. However, assessments that rely solely on money-based analyses are increasingly considered as inadequate within the broad context of sustainability (Herendeen 1999; Lange 1999; Rees 1999). Such economic evaluations do not reflect adequately important processes (e.g., waste assimilation) which are important to the integrity of ecosystems (Rees and Wackernagel 1999) and support production processes such as aquaculture.

TFP has been applied to the assessment of aquacultural systems such as shrimp farming. The behaviour of extensive, semi-intensive and intensive shrimp farming systems in Asia was compared using total factor productivity (Gunaratne 1997). That analysis however concentrated on useful outputs (i.e., shrimp) using monetary units, and the implications of farm wastes were not considered.

The increasing awareness about the structural and functional importance of ecosystems in sustaining productivity has raised the need for meaningful biophysical analyses (Odum 1997; Rees 1999). Biophysical-based analyses are regarded as more comprehensive, resembling more an overall systems approach when compared to more marginal, money-based analyses (Rees and Wackernagel 1999; Wackernagel 1999). However, the aim should be not to rely on either one, but to properly integrate biophysical and economic instruments, which is an important goal of the recent Ecological Economics discipline (Lange 1999).

2.3 Biophysical-based assessment: the use of nitrogen

An obvious problem in using biophysical units in the assessment of multiple input-output productivity is that if a steady state is assumed the ratio would be always equal one, from the basic concept of the conservation of mass. However, the consideration of other outputs in the productivity calculation, besides the conventional harvest, is more comprehensive and may render a more valuable assessment of the behaviour of food production systems. For example, semi-intensive and intensive aquacultural systems may affect themselves negatively through their outputs (i.e., wastes).

Energy is the most extensively utilized biophysical currency in the assessment of costs and benefits in food production systems. Energy input-output analyses have been conducted in agriculture (Fluck 1992; Giampietro, Bukkens, and Pimentel 1994; Pimentel 1974; Schahczenski 1984) as well as in aquaculture (Bardach 1980; Pimentel, Shanks, and Rylander 1996; Pitcher 1977). Carbon can be measured by energy, whereas carbon and nitrogen are related by a coupling of the two elements in living and non-living processes (Avnimelech 1999; Redfield 1958; Schimel, Braswell, and Parton 1997; Socolow 1999).

Given the various linkages through nitrogen between the farm and the adjacent environment and the importance of nitrogen in the aquacultural production process and its associated environmental interactions (Chapter 3), I propose to utilize nitrogen units in the assessment of aquacultural productivity.

Nitrogen has been utilized extensively to assess the efficiency of inputs such as feed and fertilizer in terrestrial animal production (Dou et al. 1998) but its use in aquaculture has been limited. As discussed in Sec. 2.2.1, the physiological implications are arguably more important in aquatic organisms that excrete nitrogenous compounds in the medium in which they have close contact and in fact grow. Waste products such as ammonia are excreted directly into the water column where they may become toxic. In contrast, excretory products do not normally affect land animals directly (Shell 1991). Therefore it may be of higher relevance to use nitrogen in the assessment of aquacultural production.

There are various important reasons to justify nitrogen utilization:

1. Nitrogen is the main building block of protein, whose production is arguably the main objective of aquaculture;

2. Nitrogen has important physiological implications on the health and growth of cultured organisms. Excess nitrogen as ammonia (NH_3), ammonium ($NH4^+$) or nitrite (NO_2^-) may be toxic to aquatic organisms (Aldrich 1980; Handy and Poxton 1993; Hargreaves 1998; Wedemeyer 1996);

3. Nitrogen is accumulating rapidly in coastal areas and has been considered as the greatest threat to the integrity of coastal ecosystems (Howarth 1998; Vitousek et al. 1997). Primary production and eutrophication in coastal areas are mainly controlled by nitrogen (Ryther and Dunstan 1971);

4. Excess nitrogen in aquatic ecosystems may have important implications on human health by contributing to toxic blooms of dinoflagellates whose toxins may affect humans through the consumption of fish and shellfish (Burkholder and Glasgow 1997; Hodgkiss and Ho 1997);

5. Formulated feeds for intensive and semi-intensive aquaculture rely importantly on fishmeal and soybean meal as protein sources, which have high nitrogen content. Those ingredients represent a high cost in formulated feeds which in turn represent the single highest cost in aquacultural production (Hargreaves 1998);

6. Nitrogenous compounds importantly influence water quality in aquaculture. The cost of maintaining appropriate water quality by reducing those nitrogenous compounds is high;

7. Food production systems represent the major flow and disruptor of the global nitrogen cycle (Bleken and Bakken 1997; Socolow 1999); and

8. Nitrogen is considered an existential imperative, whose need compared with other nutrients, can not be substituted or substantially reduced (Smil 1991).

As a result of the above reasons, nitrogen may be considered an important integrator between water quality, the aquacultural production process and the surrounding biophysical, economic and social environments. Hence, the assessment of productivity based on nitrogen may capture better the ecological and economic imperatives of aquacultural production.

Derived from the conceptual model (Fig. 2-1), the main criteria for aquacultural systems to become unsustainable would be a falling or decreasing production per unit of nitrogen input (i.e., decreasing nitrogen efficiency), and/or a decreasing capacity of the environment to assimilate nitrogenous wastes.

2.3.1 Conceptual model

Fig. 2-1. Criteria for more sustainable aquacultural systems

Efficient use of nitrogen (minimize nitrogen loss per unit of production)

↓ *Leading to:*

Less vigorous interactions with the environment (less nitrogenous waste)

↓ *Resulting in:*

Reduction in impact

↓ *Permitting:*

The maintenance of ecological integrity

↓ *Conserving:*

The capacity of an aquatic ecosystem to assimilate nitrogenous wastes

↓ *Allowing:*

The provision of adequate water quality for aquaculture

↓ *Supporting:*

The capacity of aquacultural systems to produce continuously

↓

Contributing towards:

2.4 Use of nitrogen (N) as a currency in food production systems.

2.4.1 Nitrogen Efficiency

The most common indicator to assess N use in agriculture is Nitrogen Efficiency (NE), or the ratio of N retained in harvest over the N inputs:

(2-1) NE = N in harvest / N inputs

The relationship between N inputs and N in harvest may also be expressed as Nitrogen Conversion Index which is the N input / output ratio (Gomiero et al. 1997) or as a Nitrogen Cost, which is also the inverse of NE (Bleken and Bakken 1997).

Efficiency can be defined as the ratio of output to input, and within the biological realm it represents the efficiency of a biological process (Spedding, Walsingham, and Hoxey 1981). In order to calculate efficiency it is necessary to specify 1) inputs and outputs of interest; 2) the process or system under consideration; 3) the context or environment under which the system is operating; and 4) the period of time to which the ratio refers.

Higher NE corresponds normally to environmental and economic benefits. However, important aspects of efficiency calculations are that values of a process apply only to specific situations, and that if efficiency assessment is to be used as a basis towards improvement, the system has to be appropriately described as to assess its sensitivity to important variables (Spedding, Walsingham, and Hoxey 1981).

The problem of obtaining NE is often the definition of system boundaries. Usually the use of concentrated N such as that in formulated feeds results in a higher NE at the farm level. But if the efficiency of the feeds components is assessed the results may vary considerably (Bleken and Bakken 1997; Van der Hoek 1998).

Assessment of NE has been conducted with the Ecopath model in extensive aquacultural systems, polyculture, and integrated aquaculture-agriculture systems (Dalsgaard and Oficial 1998; Lightfoot et al. 1993b; Wolff 1994). Those systems usually do not result in water quality deterioration affecting the production process, and the environment is assumed to assimilate the nitrogenous wastes. Thus, although Ecopath captures the partitioning of N into useful biomass, it does not capture the partitioning of N into nitrogenous wastes such as ammonia, which is of relevance in the more intensive forms of aquacultural production.

2.4.2 Nitrogen budgets

Nitrogen budgets are a common tool to assess N partitioning in food production systems, and are based on the principle of conservation of mass. Nitrogen budgets have had very limited use in coastal ecosystems, including aquaculture, despite their power as tools in ecosystem analysis and management (Cockcroft and McLachlan 1993).

In aquaculture, N budgets have been mainly used to determine the efficiency of specific resources such as feeds and fertilizers (Avnimelech and Lacher 1979; Schroeder 1987), the fate of nutrients in the systems (Briggs and Funge-Smith 1994; Hopkins et al. 1993), and the pollution potential of effluents (Foy and Rosell 1991).

Nitrogen budgets are usually calculated at the farm gate level (i.e., purchased inputs and outputs) due to the ease of obtaining data. Although N mass balances at the pond level require the total of inputs and outputs, general assessments at higher levels, for example the farm, can be estimated with fewer data, such as the concentration of N in feed, fertilizer and the aquacultural product (Boyd 1990; Boyd and Teichert-Coddington 1995).

Those N budgets that consider internal processes are usually calculated for simple systems. Data for various of these internal processes are lacking in shrimp farming systems (Montoya et al. 1999) and there is the need to generate this data, through simulation modelling for example, in order to understand better the partitioning of N in the system. In general, more complex budgets can give a more appropriate insight into the recycling of N, an important concept from the sustainability point of view (Watson and Atkinson 1999).

2.4.3 Simulation modelling of nitrogen dynamics.

The assessment of N dynamics in shrimp farming has focused on the more intensive forms of production. The assessment has been based on the simulation modelling of the effect of feeding parameters on shrimp growth and water quality (Montoya et al. 1999), or on the fate of ammonia in outdoor ponds (Lorenzen, Struve, and Cowan

1997). The first model focuses on the flow of feed N and shrimp growth, and it only captures the origin of ammonia while the second model assesses only its fate.

Most shrimp farming around the world is conducted as semi-intensive farming, although there are some indications of the industry becoming more intensive, such as in Mexico (Fig. 3-4). It is important then, to explore the N-related implications of the intensification trend, from extensive to intensive.

Therefore there is a need to quantitatively relate the origin and fate of N, including the contribution of natural foods, in both useful outputs (i.e., shrimp) and important wastes (e.g., ammonia), and to evaluate the implications under a wider range of production methods (i.e., from extensive to intensive). Such a comprehensive approach would imply a need to consider both, the capacity of the organisms to retain N (i.e., NE) and the capacity of the environment to assimilate nitrogenous wastes.

CHAPTER 3. SHRIMP FARMING AND NITROGEN

In Chapter 3 I present the case study, a review of aquacultural systems, the aquacultural production methods and their characteristics, and the environmental interactions associated with the intensification of aquaculture. The main focus is on the interactions between shrimp production and water quality through nitrogen (N) flows, and the importance of the assimilative capacity of the environment with regard to nitrogenous wastes.

3.1 Case study

3.1.1 Focus on shrimp farming.

Shrimp farming is concentrated in tropical, developing countries where shrimp[3] have been reared for hundred of years mainly as incidental crops as they enter coastal fish ponds as juveniles. Increasing demand, high profitability and scientific and technological breakthroughs led shrimp farming towards more intensive production methods in the 1970's and 1980's with the support of national governments and international development agencies (Neiland, Soley, and Baron 1997). The intensification process resulted in substantial increments in production (Fig. 3-1). By 1997, shrimp farming accounted for 27-29% of global shrimp production (both wild-caught and farm-raised), in contrast to a contribution of 6% in 1970 (FAO 1997).

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[3] The marine and brackish water crustaceans of the family Penaeidae. The FAO convention is to call marine and brackish water forms, shrimp, and freshwater forms, prawns.

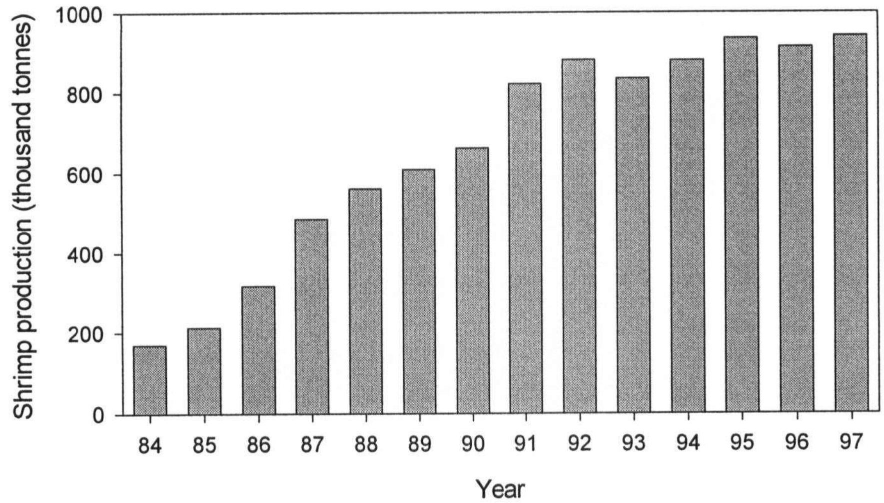

Fig. 3-1. Global shrimp farming growth (FAO 1997)

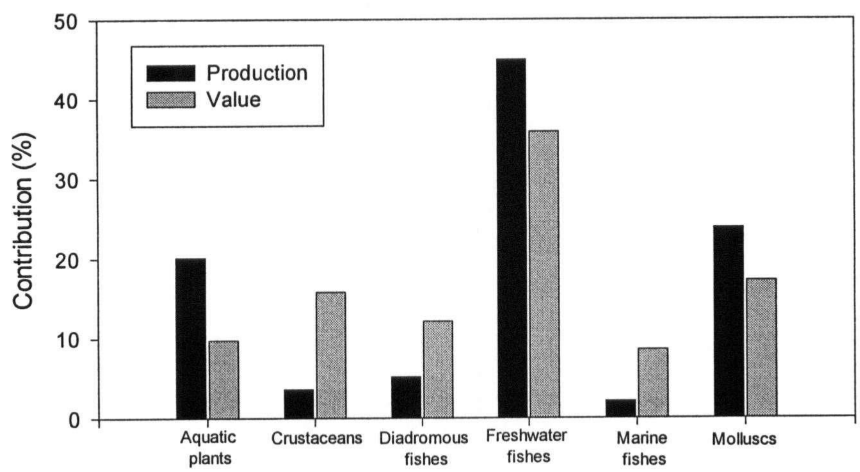

Fig. 3-2. Contribution of major farmed groups to global aquaculture production and value in 1997. Total production = 36.05 million tonnes. Total value = US$ 50.36 billion (FAO 1999)

Although not significant as a source of food, shrimp farming contributes importantly to some Asian and Latin American economies through the generation of income, foreign exchange earnings and jobs. Marine shrimp are exported mainly to developed countries such as the United States, Japan and Europe. Farmed crustaceans (of which 90% were marine shrimp) represented 3% of total aquaculture production by volume, but 16% by value, worth US$6.6 billion in 1997 (Fig. 3-2) (FAO 1999).

However, the shrimp farming industry has since the late 1980's faced serious disease outbreaks, mainly viral, which has resulted in considerable production declines and serious financial losses (Kongkeo 1997). Although there has been recovery and increased production in previously affected areas, these gains have been offset by the presence of disease in new areas. This has been the pattern of production in the last years, which is expected to continue for some time (FAO 1999). It is increasingly recognized that disease and the resulting production collapses are strongly linked to water quality deterioration caused by the intensification practices (Shang, Leung, and Ling 1998).

3.1.2 Shrimp farming in Northwest Mexico

To illustrate the application of my approach, I selected the case of shrimp farming in Northwest Mexico. The Mexican government supports aquaculture, and shrimp farming in particular, as an important activity to create employment and income. Aquaculture was considered in the National Development Plan of Mexico 1995-2000 as a mechanism of economic growth and poverty alleviation (SEMARNAP 1994), and as a way of diversifying Mexico's economy, which has been primarily based on oil. This has happened under a relatively stagnant capture-shrimp fishery.

The economic importance of aquaculture in Mexico is small, contributing less than 1% to agricultural GDP (SAGAR 2000). Of the total fisheries production in 1998, aquaculture represented less than 13%; most aquacultural production (65%) originated from inland waters and the rest was represented by brackish and marine species (SEMARNAP 1998). However, the greatest potential lies with coastal aquaculture given the extensive coastline (approx. 10,000-km), the amount of estuaries and the variety of finfish and shellfish species and favourable year-round temperatures (Merino 1987). For example, the current area under shrimp farming (Fig. 3-4) represents a small fraction of the 335,000 ha estimated with potential for shrimp aquaculture in Mexico (Fig. 3-3) (FIRA 1998). Besides the potential area for development, the proximity to United States represents a strong incentive for the Mexican shrimp farming sector to generate foreign exchange in already well-established marketing channels (Martinez and Pedini 1998).

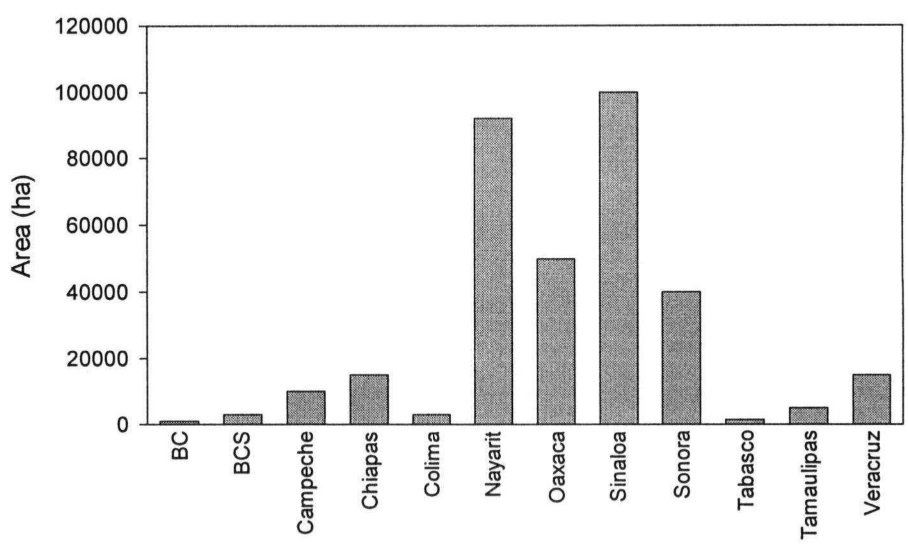

Fig. 3-3. Mexican states and their estimated potential area suitable for shrimp farming (FIRA 1998).

The farming of marine shrimp, which is mainly export-oriented, represents the most important aquacultural subsector in Mexico in economic terms. Recent changes in the Mexican laws that affect shrimp aquaculture advocate its development by allowing the private sector to participate in the farming of those species. The farming of marine shrimp was legally reserved for the social sector before the changes in fisheries laws that started in 1986 (Miller 1990). Those changes resulted in a rapid growth of shrimp aquaculture in the last years (Fig. 3-4).

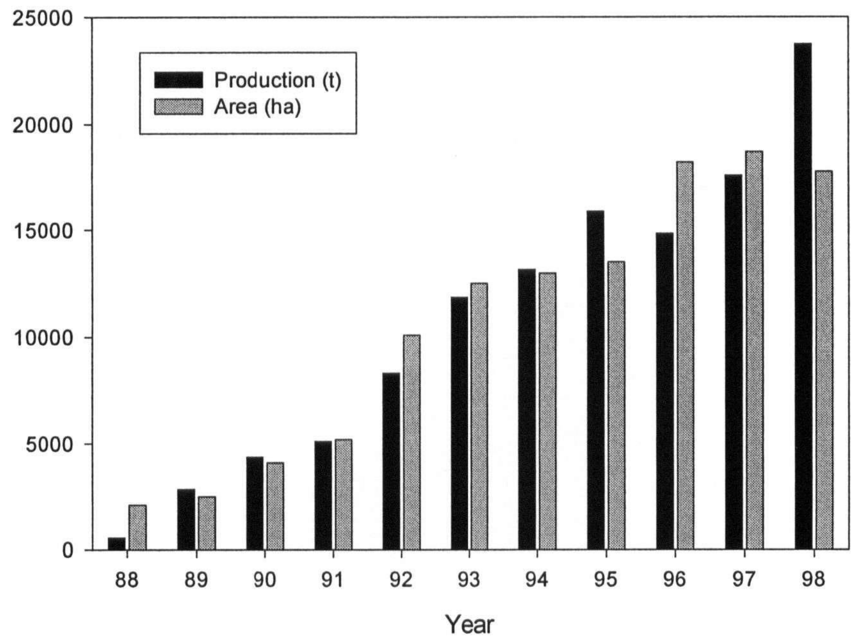

Fig. 3-4. Shrimp farming growth in Mexico (SEMARNAP 1998)

The Pacific white shrimp, *Litopenaeus vannamei*, is the most common species of shrimp being farmed in Mexico. *L. vannamei* is farmed mainly under semi-intensive production methods which represented 84.3% of the production and 80% of the total area under farming in Northwest Mexico in 1998 (Table 3-1 and Fig. 3-5). Other marine shrimp species such as *L. stylirostris* and *Farfantepenaeus californiensis* are also farmed, although in minor quantities.

Table 3-1. Shrimp farming production, number of farms, area, and production method in Sonora, Sinaloa, and Nayarit in 1998 (SEMARNAP 1998).

State	**Production t yr^{-1}**	**# Farms**	**Area (ha)**	**Production method**					
				Extensive		Semi-intensive		Intensive	
				# Farms	Area (ha)	# Farms	Area (ha)	# Farms	Area (ha)
Sonora	6934	33	4411	1	170	30	4004	2	237
Sinaloa	12719	119	10886	29	2115	87	8537	3	234
Nayarit	2140	75	1753	40	544	31	1123	4	85
Total	21794	227	17051	70	2830	148	13665	9	556

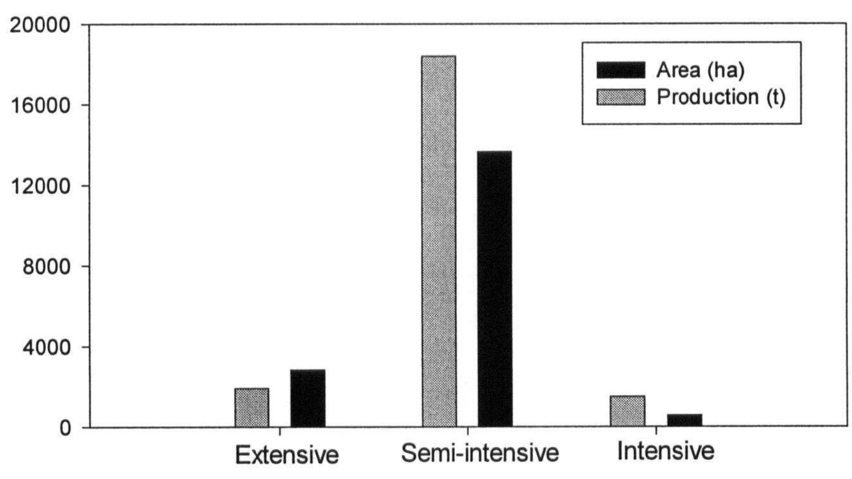

Fig. 3-5. Production and associated area of different shrimp farming methods in Mexico in 1998 (SEMARNAP 1998). Estimates to disaggregate production assumed that average production of semi-intensive methods is two-fold of extensive, and half of intensive methods.

More than 90% of shrimp farming production in Mexico originates from the states of Sonora, Sinaloa and Nayarit (Table 3-1 and Fig. 3-6), which border the Gulf of California, the most important fishing area in Mexico and one of the most productive waters in the world. Sonora and Sinaloa also represent the breadbasket of the country, with intensive forms of agricultural production which result in considerable discharges of excess nutrients, such as N, into coastal waters (Paez-Osuna, Guerrero-Galvan, and Ruiz-Fernandez 1999).

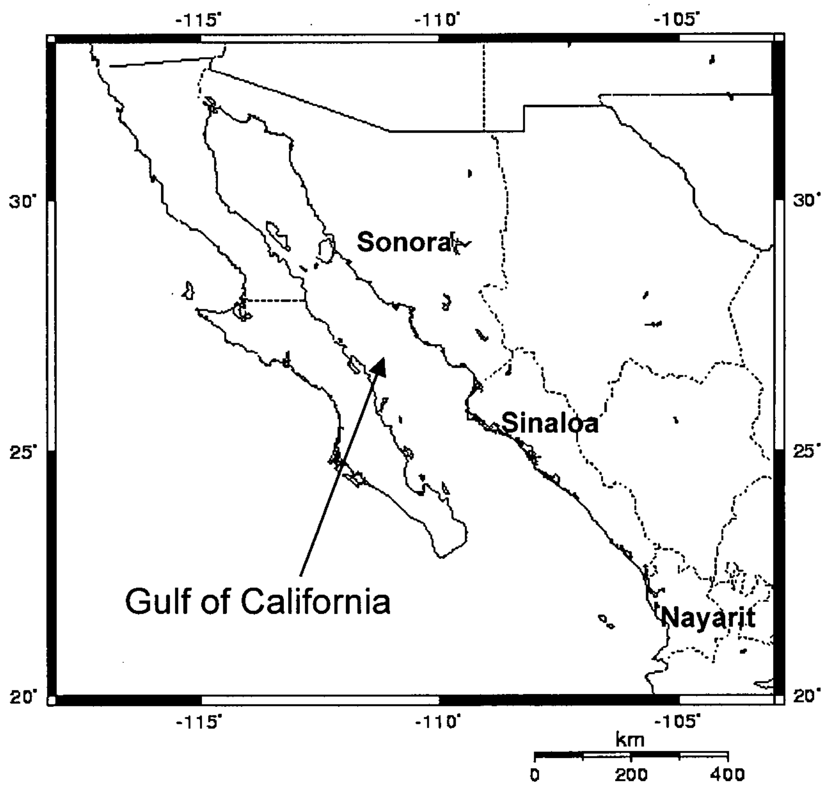

Fig. 3-6. The Gulf of California bordered by the states of Sonora, Sinaloa and Nayarit, the main producers of farmed shrimp in Mexico.

A recent loan of US$ 40 million by the World Bank to the Mexican Government will support an aquaculture project that aims to improve and complete the regulatory framework for the sector (World Bank 1997). The Mexican government recognizes that the aquaculture development framework needs to be improved and strengthened, especially in "the policies and procedures that address environmental and social concerns." Aquaculture, if improperly developed, may have negative consequences on coastal ecosystems and human communities. One of the main components of that project will try to address the environmental concerns related to aquaculture and specifically, the construction of a sound regulatory framework for aquatic health. Therefore it is important that appropriate tools are developed in order to assess aquaculture and understand the environmental interactions associated to its development and potential intensification.

3.2 Aquacultural systems: a review

Aquacultural systems, like their agricultural counterparts, are ecological systems managed by humans to produce food or other useful products. They are characterized by their dependency on natural resources and services, and the degree of manipulation of their natural processes (Conway 1987). A system can be defined as "a complex whole, a set of connected things or parts, or an organized body of material or immaterial things" (Oxford Dictionary). Food production systems are characterized by their functioning (i.e., processes) and structure (i.e., physical and biological components) and depend importantly on four main resources: 1) water, in both quality and quantity; 2) nutrients, in the form of feed and/or fertilizers; 3) land, or space; and 4) seed, or stock.

3.2.1 Production methods

Aquacultural systems undergo various degrees of manipulation of their processes and extend from those systems with little intervention to those with a high degree of management. In general, production methods are categorized according to their production capability, or yield, and the intensity of use and management of the four resources mentioned above. In an attempt to increase yields, there is a progression of increasing stocking densities, which are supported by corresponding feed and water management levels.

Aquacultural production methods may be categorized according to the management intensity and corresponding use of resources (Pillay 1994). The three basic categories are extensive, semi-intensive and intensive.

Accordingly, shrimp farming can be categorized into the three categories (Table 3-2). *Extensive shrimp farming* is a variation of what may be termed the traditional methods and it is highly dependent on the natural productivity, with no supplemental food. This method usually depends on tidal water exchange for wild seed supply and maintenance of water quality. Farming densities are low and production is limited by the capacity of water to provide oxygen and dilute wastes. Production systems that utilize this method may be considered low-input and generate small nutrient loading to the environment. *Semi-intensive shrimp farming* utilizes higher stocking densities, supplementation of natural with artificial feeds, and fertilization to maintain natural productivity. At higher densities, availability of natural food becomes the first limiting factor, which requires supplemental feeding and/or fertilization. If the capacity of water to provide oxygen and assimilate potentially toxic metabolites is exceeded, water exchange becomes necessary. *Intensive shrimp farming* relies on higher stocking densities when compared to the other methods, and there is a complete reliance on artificial feeds. Water flow (volume per unit of time) is commonly increased to minimize effects on water quality by ammonia excretion, among other factors. This production method is normally characterized by producing high nutrient loads (Handy and Poxton 1993).

Table 3-2. Characteristics of shrimp farming at different levels of intensity. (From Lee and Wickins 1992)

	Extensive	**Semi-intensive**	**Intensive**
General characteristics	Low density; often as polyculture with fish and crabs	Moderate density of mix or single shrimp species	High density monoculture
Water system	Tidal flushing	Daily tidal/or pumped water exchange	Continuously controlled pumping
Fertilization	None or organic, prior to filling	Organic or inorganic on filling and as required	Organic or inorganic as required
Feed	Shrimp rely solely or mostly on natural productivity	Natural productivity + supplemental feeds	Artificially formulated feeds
Aeration	None	Used when needed	Used regularly
Pond sizes	0.5 – 100 ha 0.3 – 1m deep	0.2 - 3 ha 0.8 - 1.5 m deep	0.1 – 1.0 ha 1.2 - 1.5 m deep
Stocking density	0.2 – 5 shrimp m^{-2}	5.0 – 20 shrimp m^{-2}	15 – 50 shrimp m^{-2}
Shrimp size	15 –50 g	15-40 g	15 – 40 g
Yield cycle $^{-1}$	< 1 t ha^{-1}	0.5 – 5.0 t ha^{-1}	5 – 15 t ha^{-1}

There is no a clear dividing line between production methods but a continuum from extensive to intensive. Given the diversity of species, farming systems, and production practices in aquaculture, the classification of production methods is rather arbitrary. The dependence on flowing water is probably the main criterion to separate between extensive, semi-intensive and intensive methods, where production is limited substantially by flow and not by volume (Krom, Neori, and van Rign 1989; Wedemeyer 1996).

The use of a particular production method in aquaculture depends on numerous variables, including availability and quality of water resources, climatic conditions and the socioeconomic context, among others (Gomiero et al. 1997). Also, each production method has characteristic ecological, economic and social implications (Neiland, Soley, and Baron 1997).

3.3 Intensification trend and environmental interactions

As shrimp farming developed, production methods changed, moving from extensive approaches with low inputs toward intensive methods that require high levels of inputs. High profits in shrimp farming have motivated investors to move towards more intensive production practices (Shang, Leung, and Ling 1998). Although most shrimp farming around the world is classified as extensive or semi-intensive, there is a clear tendency in the last few years to adopt semi-intensive and intensive methods. In Thailand, the world leading producer, 85% of farms practice intensive methods (Rosenberry 1998).

Shrimp farming systems, similar to other aquacultural systems, can be characterized by having three stages in the production process: 1) a flow of inputs from the external environment e.g., feeds and fertilizer; 2) a process of production i.e., biological growth of the organism; and 3) a flow of outputs to the external environment e.g., harvesting and waste disposal. Accordingly, the sources of environmental interactions in aquaculture have been classified into the consumption of resources, the production process and the production of wastes (Beveridge, Ross, and Kelly 1994). The increasing intensification of shrimp farming has brought along an increasing use of resources and manipulation of natural processes, which intensifies the interactions with the environment at each of the three stages of the production process.

For aquaculture in general, the increasing environmental interactions may result in an alteration of natural balances in the aquacultural ecosystem, which may also result in various environmental consequences (i.e., impacts). These resulting impacts, can be positive (i.e., beneficial) or negative (i.e., detrimental) and may consist of both biophysical (i.e., ecological) and socioeconomic changes. Positive effects include, for example, the removal of nutrients from eutrophic waters through the farming of molluscs and seaweed (Folke and Kautsky 1992; Inui, Itsubo, and Iso 1991). Adverse effects include, among others, the degradation of receiving waters through the release of excess nutrients in the effluents. Other negative consequences include the increasing conflicts that several aquaculture projects have with other natural resource sectors, namely agriculture, fisheries and tourism that compete for the same resources.

Negative consequences such as pollution are mainly associated with intensive forms of aquaculture, or with the intensification process, through its higher demand for resources (Midlen and Redding 1998); the higher the demand, the more negative the environmental impacts usually (Beveridge, Phillips, and Macintosh 1997). Similar trends have been observed in agricultural systems (Matson et al. 1997). However, intensification (i.e., increasing yields) and extensification (i.e., opening new areas for development) may have similar effects on surrounding ecosystems. Extensification has a considerable requirement for land with a visible impact upon wetlands, while intensification increases the concentration of wastes (Davies and Afshar 1993; Funge-Smith and Briggs 1998).

Environmental impacts of shrimp farming are often associated with biophysical changes at the farming site (Stewart 1995). Although shrimp farming is commonly managed in isolation from adjacent ecosystems it has wider environmental interactions, like intensive agriculture (Matson et al. 1997). Influence on, and from, other ecosystems is particularly evident in more intensive production practices that import resources from other systems (Folke, Kautsky, and Troell 1994).

As discussed above, aquaculture may not only impact on, but it may also be impacted by environmental changes. The discharge of excess agricultural fertilizers (e.g., N) in upstream waters may affect downstream aquacultural farms. Industrial discharges and urban sewage may also affect considerably the water quality required by aquaculture. Negative feedback effects such as the accumulation of metabolites (e.g. ammonia) may affect the aquaculture operation itself.

The major interactions that occur between aquaculture and the environment[4] have been

[4] Physical, biological, social and economic variables that interact with the production system under consideration. (Adapted from Conway 1987)

reviewed extensively in the literature (Barg 1992; Beveridge, Ross, and Kelly 1994; Beveridge, Phillips, and Macintosh 1997; Chua, Paw, and Guarin 1989; Gowen et al. 1990; Iwama 1991; Pillay 1992; Pullin 1993).

3.4 Nitrogen and water quality interactions.

An important element appearing in the interactions between aquaculture and the environment is nitrogen[5] due to its role as nutrient and toxicant (Handy and Poxton 1993; Lorenzen, Struve, and Cowan 1997). Aquacultural production is supported importantly by flows of N as a nutrient, supporting the synthesis of protein that translates into the growth of farmed organisms. But that process results also in the production of nitrogenous wastes, as dictated by the first and second thermodynamic laws (Midlen and Redding 1998). Therefore, water quality may change considerably, directly or indirectly, as a result of the flows of N inputs and outputs. Some nitrogenous wastes, such as ammonia, may be highly toxic to aquatic organisms, affecting the aquacultural production process and the adjacent environment. The accumulation of nitrogenous metabolites is considered one of the major problems in the more intensive forms of aquaculture (Avnimelech 1999).

[5] The sum concentration of all forms of nitrogen is referred as Total Nitrogen, or Total N, while NH_3-N, for example, refers to the amount of nitrogen present as ammonia. Total ammonia, T_{Amm}, refers to the sum of NH_3 and NH_4^+ while TAN refers to the total amount of nitrogen present in that ammonia (i.e. TAN = NH_3 + NH_4^+-N).

In addition to its availability, water quality[6] may be considered the dominant resource in aquacultural operations, and success in aquaculture is highly related to maintaining an appropriate level of water quality (Handy and Poxton 1993; Poxton 1990; Williams 1997). Collapses in the production of shrimp farming are related to disease and water quality deterioration (Barg and Phillips 1997), such as the case of Taiwan in 1987, Philippines in 1989, Indonesia in 1991-92, China in 1992 and 1993 (Primavera 1997) and Ecuador (Olsen 1995). Most shrimp disease (e.g., viral) is thought to occur as a result of poor water quality that stresses and weakens the shrimp, and allows pathogens to invade (Funge-Smith and Briggs 1998). Accordingly, water quality has been identified as a major environmental indicator in aquaculture (Stewart 1995).

Pond water and sediments are the major components of the pond system and are in continuous interaction, influencing the quality of the rearing environment (Funge-Smith and Briggs 1998). In practice, the sediment acts as an important reservoir of microorganisms and chemicals and cannot be separated from the water column. In fact, use of the term "water quality" in aquacultural ponds without referring to the sediments may be highly imprecise (Smith 1993).

3.5 Nitrogen flows in shrimp farming systems

The intensification process of shrimp farming implies a higher degree of manipulation of

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[6] Water quality in aquaculture is defined as the combination of chemical, physical, and biological characteristics of water which have direct or indirect influence on the growth and survival of organisms (Boyd 1990)

N flows (into and from the systems) through increased use of feeds, fertilizers, water exchange and higher yields (i.e., harvest). This is similar to the increasing control of nutrient flows towards the more intensive forms of production in agriculture (Fresco and Kroonenberg 1992).

There are two relevant forms of N in aquaculture: N in nutrients, and organic N. The principal N nutrients are ammonia (NH_3), and the ions ammonium (NH_4^+), nitrite (NO_2^-) and nitrate (NO_3^-). NH_3 and NH_4^+ coexist in equilibrium in water, and chemical analysis measures both forms, the amount of each species depending on pH, temperature and salinity (Wedemeyer 1996). Organic N is that bound to carbon and is present in molecules such as proteins and aminoacids.

Nitrogen may enter and exit shrimp farming systems as organic or inorganic forms, either through natural processes (e.g., N fixation, atmospheric deposition, and ammonia volatilization) or through management practices (e.g., feeding, fertilization, water exchange, and sediment removal).

3.5.1 Natural flows

Natural flows of N such as fixation, deposition and volatilization, and their processes, have been reviewed in aquatic ecosystems (Howarth, Marino, and Cole 1988; Howarth et al. 1988; Seitzinger 1988) and in aquaculture in particular (Hargreaves 1998). These flows and processes are poorly understood in aquaculture, and their contribution to the total N flow varies much among different systems. In general their contribution appears to be minor, except for some aquacultural systems in tropical regions (Hargreaves 1998).

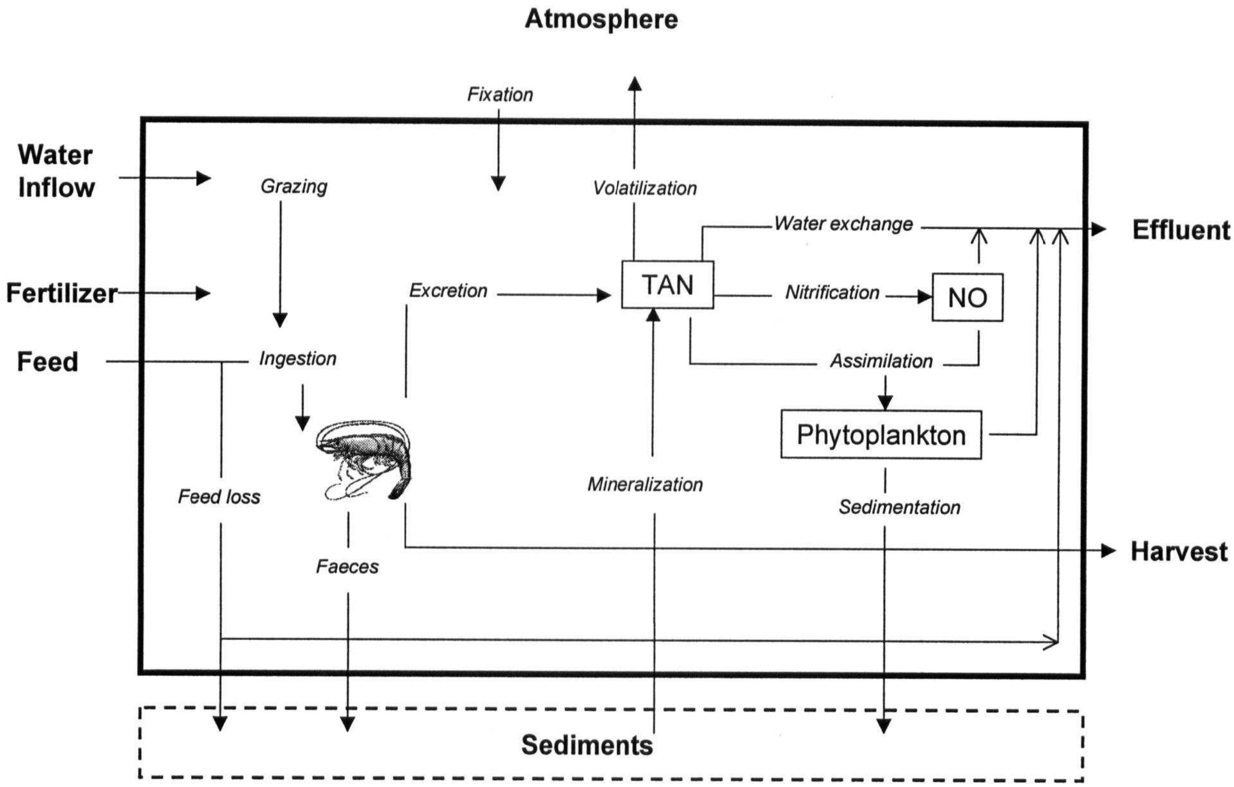

Fig. 3-7. Major natural and managed nitrogen flows in shrimp farming systems. TAN = total ammonia nitrogen; NO = nitrite and nitrate.

3.5.2 Managed flows

In contrast to natural flows, managed flows of N such as feeding, fertilization and water exchange may contribute substantially to the total N flow in semi-intensive and intensive shrimp farming practices (Hargreaves 1998). While extensive shrimp farming relies on primary production that may be enhanced by fertilization, semi-intensive and intensive shrimp farming receive high N input mainly in the form of artificial feed (organic form of N (Table 3-3).

Nitrogen from formulated feeds represents the highest N input in semi-intensive and intensive forms of shrimp farming. Feeds may account for more than 90% of total N input in semi-intensive to intensive shrimp farming (Briggs and Funge-Smith 1994; Hargreaves 1998). Table 3-3 compares feed N as a proportion of total N inputs in shrimp farming systems as well as in agricultural production systems.

3.5.2.1 Shrimp growth

Nitrogen from formulated feed that is ingested by shrimp may be retained as animal tissue, contributing to growth. The incorporation of feed N into shrimp biomass has been estimated through the measuring of N in the shrimp carcass;

(3-1) N Retention = ((Final Body N – Initial Body N) / (Total dietary N supplied)) x 100

and/or through nutrient mass-balance calculations.

Nitrogen retention values in shrimp range from 17% (Funge-Smith and Briggs 1998) in outdoor ponds, to 63% using semipurified diets in indoor tanks (Velasco, Lawrence, and Neill 1998), and there exists a wide variation among results (Montoya et al. 1999). Typical values are in the range of 20 to 40% (Phillips, Lin, and Beveridge 1993). Nitrogen retention for fish in general range from 18-49% (Handy and Poxton 1993), while in farmed fish the range is from 11% (for carp polyculture) to 36% (Hargreaves 1998).

Table 3-3. Feed nitrogen as percent of the total nitrogen input into aquacultural and agricultural systems.

	Feed N %	*Place*	*Reference*
Aquaculture			
Shrimp farming	92	Thailand	(Briggs and Funge-Smith 1994)
Shrimp farming	78	Thailand	(Funge-Smith and Briggs 1998)
Shrimp farming	76	Mexico	(Paez-Osuna et al. 1997)
Integrated farming	14	Hungary	(Olah, Pekar, and Szabo 1994)
Agriculture			
Intensive livestock production	93	Europe	(Brouwer 1998)
Intensive dairy farms	67	USA	(Dou et al. 1998)
Dairy farms	35	Europe	(Brouwer 1998)
Global animal production	18	World	(Van der Hoek 1998)
Global agricultural sector	7	World	(Van der Hoek 1998)
Integrated dairy farms	2	USA	(Dou et al. 1998)

Nitrogen contributing to growth may also originate from the consumption of natural foods as shrimp grown on farms are typically carnivorous, feeding on molluscs, polychates and other crustaceans (Goddard 1996), although this contribution is poorly understood. Some studies suggest that grazing may account for more than 50% of *L. vannamei* growth (Anderson, Parker, and Lawrence 1987; Martinez-Cordova, Pasten-

Miranda, and Barraza-Guardado 1998). The contribution of natural foods appear to be inversely related to stocking densities, and shrimp nutritional requirements in intensive practices may rely mostly on formulated feeds (Fast and Lannan 1992). Still, grazing may contribute importantly in the extensive to semi-intensive range of shrimp farming (Reymond and Lagardere 1990).

3.5.2.2 Waste nitrogen

Feed N not incorporated into shrimp tissue is considered as waste, and represents an important management problem in the more intensive forms of shrimp farming (Lorenzen 1999; Phillips, Lin, and Beveridge 1993).

Nitrogen that was not absorbed by the shrimp gut and retained, either from formulated feeds or natural foods, is excreted through the gills or disposed of as faeces (Handy and Poxton 1993). Aquatic invertebrates such as shrimp are ammonotelic, whose main excretory product is ammonia as NH_3 (Randall, Burggren, and French 1997). Ammonia excretion accounts for more than 85% of nitrogenous excretion in marine shrimps (Cockcroft and McLachlan 1987). Ammonia may also be produced by mineralization of uneaten feed and faeces; levels of 10-20% of uneaten feed are often reported in semi-intensive shrimp farming, although these levels may be as high as 50% in more intensive systems (Wyban, Sweeney, and Kanna 1988).

Most (> 90%) of the waste N in semi-intensive and intensive aquaculture, including shrimp farming, enters the water column as ammonia, which may be highly toxic to the farmed organisms and has to be maintained at low levels (Burford and Glibert 1999; Hargreaves 1998).

Table 3-4. Nitrogen waste loading, in kg per tonne of fish produced.

Species	Production system/ method	Nitrogen waste $kg \cdot t^{-1}$ fish	Place	Ref
Penaeids				
Litopenaeus stylirostris	Ponds (semi-intensive)	157.2	New Caledonia	1
Penaeus monodon	Ponds (intensive)	102.3	Thailand	2
L. stylirostris	Ponds (extensive)	60.8	New Caledonia	3
L. vannamei	Ponds(semi-intensive)	58	Mexico	4
L. vannamei	Ponds (semi-intensive)	28.6	Mexico	5
Salmonids				
Onchorynchus mykiss	Tanks	124.2	Ireland	6
O. mykiss		103.8	UK	7
O. mykiss	Cages	81	Sweden	8
O. mykiss	Raceways	47-87	USA	9
Salmo salar	Tanks	71	Scotland	10
Grouper	Cages (fed trash fish)	321	China	11
Bream	Cages	211	Japan	12
Yellowtail	Cages	68-109	Japan	13
Turbot	Tanks	51	France	14
Cyprinids				
Carps (Polyculture)	Ponds	23.9	China	15
Carp	Cages	0.1-0.2	Indonesia	16

(1and 3) (Martin et al. 1998); (2) (Briggs and Funge-Smith 1994); (4) (Paez-Osuna, Guerrero-Galvan, and Ruiz-Fernandez 1998); (5) (Paez-Osuna et al. 1997); (6) (Foy and Rosell 1991); (7) (Phillips, Beveridge, and Muir 1985); (8) (Enell and Lof 1983); (9) (Axler et al. 1997); (10) (Kelly, Stellwagen, and Bergheim 1996); (11) (Leung, Chu, and Wu 1999); (12 and 13) (Watanabe 1991); (14) (Mallekh, Boujard, and Lagardere 1999); (15) (Guo and Bradshaw 1993); (16) (Costa-Pierce and Roem 1990)

Shrimp rely on diets with high N content i.e., high fishmeal content. These diets have a high potential for N pollution and N loads can increase if there is overfeeding and diets are unstable and highly soluble (Handy and Poxton 1993). The feeding nature of shrimp (i.e., external mastication) contributes importantly to the leaching of nutrients from formulated feeds, particularly from poorly bound pellets (Goddard 1996). Food wastage is considered an important environmental issue and the most important source of N pollution in semi-intensive and intensive aquaculture production methods (Ackefors and Enell 1990; Seymour and Bergheim 1991).

Nitrogen waste may exit the shrimp-farming pond through the effluent, through sediments or through volatilization. Nitrogen nutrients and organic N discharged through the effluent can be transported in solution or attached to solid particles. Particulate N may also exit the system through sedimentation, while ammonia may escape as ammonia gas through volatilization. Volatilization of ammonia is particularly important when the pH is high and a significant proportion of T_{Amm} is present in un-ionized form.

An important management objective in shrimp farming is to reduce the concentration of toxic N added by excretion and feed loss. Intensive forms of shrimp aquaculture rely on high rates of water exchange to maintain ammonia below toxic levels, while some semi-intensive systems rely on low water exchange and biological N transformations.

Semi-intensive and intensive farm effluents may be produced with ammonia content higher than receiving waters but sufficient dilution of wastes occurs usually in seawater to become toxic (Handy and Poxton 1993). However, the periodic discharging of the nutrient rich effluents into receiving waters has the potential to negatively impact these systems, and excess nutrient loads on the environment have resulted in a series of problems (Pullin 1993). Considerable nutrient discharges were associated with the collapse of shrimp production in Thailand in 1989-1990 and further problems in 1992 and 1993, with an estimated loss of US$30 million and the abandonment of 45,000 ha. (Briggs and Funge-Smith 1994).

Few nutrient budgets have been performed in semi-intensive and intensive shrimp farming, but those studies indicate that 35% of the total N that enters the pond is discharged through effluents into the adjacent environment. If sediments also are flushed out after harvest, a common practice in many areas to clean ponds, N discharged may be higher than 65% (Briggs and Funge-Smith 1994).

Most shrimp farming takes place in semi-static ponds (i.e., water exchange is frequent and of high volume, but not continuous) for which pollution is of higher relevance than in other forms of intensive aquaculture, such as salmon farming in cages. These effects are intensified when there is a concentration of farms in a particular area and waste metabolites are discharged through the effluents.

3.6 Toxicity of nitrogenous compounds

Ammonia (NH_3) is highly toxic to shrimp, in contrast to ammonium (NH_4^+) which has a lower toxicity. Generally molecules in un-ionized forms are more toxic because they are able to pass through membranes more easily than ionized forms.

Although mortalities in aquaculture have been linked to exposure to high levels of ammonia (Tarazona et al. 1987), its toxicity is reflected more often as a reduction of growth or immunocompetence of the cultured organisms (Hargreaves 1998). There is a higher incidence of physiological stress, lowering resistance to disease, in more intensive production practices (Midlen and Redding 1998).

Table 3-5. Acute toxicity of ammonia to penaeid species (From Zhao, Lam, and Guo 1997).

Species	***Stage***	LC_{50} *(mg* NH_3*-N·L* $^{-1}$*)*			
		24 h	**48 h**	**72 h**	**96 h**
Penaeus monodon	nauplii	0.54			
	zoeae	0.76			
	mysis	2.17	1.3		
	postlarvae	4.7	2.5	1.54	1.04
	juveniles	2.68	2.33		1.69
Marsupenaeus japonicus	nauplii	1.31			
	zoeae	0.97			
	mysis	1.08			
	postlarvae	1.98			
Fenneropenaeus indicus	nauplii	0.29			
	protozoeae	0.95	1.18		
	mysis	3.17			
F. chinensis	nauplii	0.25			
	zoeae	0.34			
	mysis	1.08			
	postlarvae	1.85			
	juveniles	3.88			
Farfantepenaeus paulensis	nauplii	4.25	1.73	1.06	
	zoeae	1.79	1.13	0.85	0.73
	mysis	2.91	1.62	1.28	0.85
	postlarvae	1.4	0.5	0.33	0.32
	juveniles	1.47	1.22	1.15	1.1

Nitrite (NO_2^-) may also be toxic to shrimp, but in general crustaceans have a lower sensitivity to NO_2^- when compared to teleosts. Nitrite causes haemoglobin to reduce to metahaemoglobin, which is less capable of binding oxygen. However, crustaceans have haemocyanin, containing copper, in contrast to teleosts which have haemoglobin, containing iron (Wickins 1976a).

3.7 Assimilative capacity of the environment.

To prevent water quality deterioration, aquaculture depends importantly on the capacity of the environment to assimilate nitrogenous wastes, particularly ammonia. In general food production systems, such as aquaculture, rely importantly on this assimilative capacity[7] as a waste treatment system (Cairns 1977). If the assimilative capacity is exceeded, wastes may deteriorate the water quality, and stress substantially the cultured organisms by affecting their growth and survival, and the integrity of the supporting ecosystems.

Phytoplankton assimilation represents the main pathway for the removal of dissolved N metabolites, particularly ammonia. Ammonia may be transformed to nitrate, via nitrite through nitrification. Both ammonia and nitrate may be utilized by phytoplankton leading to particulate organic N (Lorenzen, Struve, and Cowan 1997). Ammonia is preferred over nitrate as substrate, as the latter is energetically less favourable to assimilate (Hargreaves 1998). Dissolved N, which is assimilated, is then converted into particulate N (i.e., phytoplankton biomass) and removed from the water column through sedimentation. Phytoplankton crashes may lead to dramatic increases in ammonia concentration in aquacultural ponds (Hargreaves 1998; Krom, Neori, and van Rign 1989), as well as to peaks in oxygen consumption leading to hypoxic stress and even

[7] "The ability of an aquatic ecosystem to assimilate a substance without degrading or damaging its ecological integrity i.e., structure and function" (Cairns 1977)

mortality.

Shrimp farming utilizes relatively low rates of water exchange, and therefore it relies importantly on phytoplankton to reduce ammonia levels through assimilation (Lorenzen 1999). Thus, the conversion of potentially toxic, dissolved N into organic N through stable phytoplankton populations is an important mechanism of maintaining appropriate water quality in shrimp farming ponds (Hargreaves 1998).

In summary, N in shrimp farming represents an important integrator between the farm and the environment, through various managed and unmanaged fluxes that may be the source of important interactions affecting aquacultural production itself, and the environment. The consideration of N interactions among important components of the shrimp farming system, such as shrimp, water, phytoplankton, sediments and atmosphere, and the use and production of various nitrogenous inputs and outputs appears relevant in the assessment of shrimp farming systems, from both the biophysical and the socioeconomic dimensions.

CHAPTER 4. METHODOLOGY

Chapter 4 contains the methodological approach of this study. To assess the long-term productivity potential of shrimp farming I developed the Nitrogen Productivity (NP) indicator through the merging of Nitrogen Efficiency (NE), an existing indicator, and Ammonia Assimilative Capacity (AAC), an indicator that I also developed. A shrimp farming system was divided into shrimp and water subsystems. The performance of the shrimp subsystem was assessed through NE, while the performance of the water subsystem was assessed through AAC. The tools for the analysis consisted of 1) a shrimp pond simulation model based on available data in the literature and used to calculate the indicators; and 2) a N budget model, to quantitatively relate the contribution of N inputs that were not considered in the simulation modelling.

4.1 Analytical framework

From the literature review and analysis in Chapters 2 and 3, I selected the multiple input-output approach to productivity measurement as the most suitable indicator to assess the performance of shrimp farming systems in the long term. From that review and analysis, I also selected nitrogen (N) as an appropriate currency for the calculation of multiple input-output productivity in shrimp farming. The development of a conceptual model for the thesis indicated that a comprehensive, more integrated approach to productivity measurement required: 1) the consideration of the capacity of the farmed organisms to retain N; and 2) the capacity of the environment to assimilate the nitrogenous wastes generated during the production process. Ammonia is the most important nitrogenous waste in aquacultural production, as discussed in Chapter 3, and therefore I focused on the capacity of the environment to assimilate that metabolite. Fig. 4-1 describes diagrammatically the N flows, into and from an aquacultural system, and the proposed indicators to assess its performance.

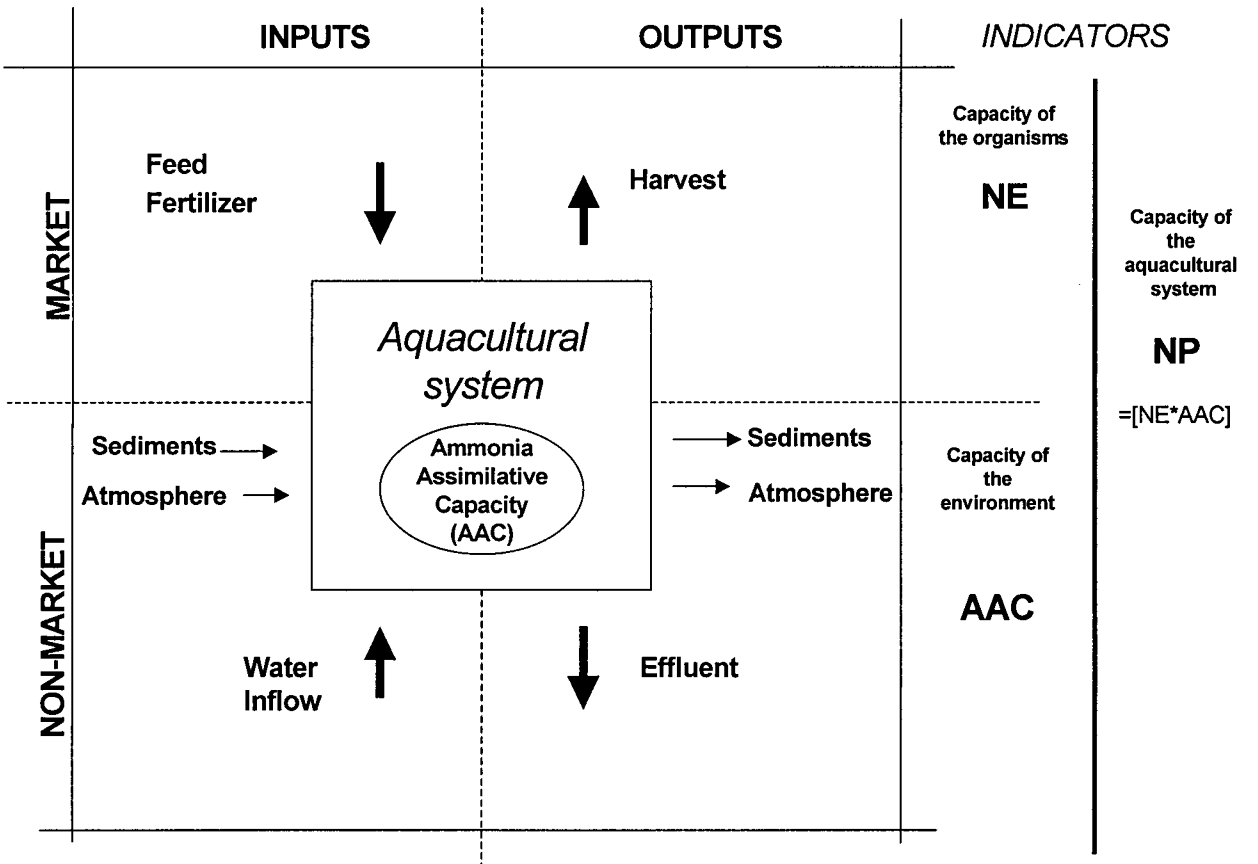

Fig. 4-1. Schematic diagram of nitrogen as a currency in the assessment of long-term productive potential of aquacultural systems. NE = Nitrogen Efficiency, or the capacity of farmed organisms to retain nitrogen; AAC = Ammonia Assimilative Capacity, or the capacity of the environment to assimilate ammonia; NP = Nitrogen Productivity, or the capacity of aquacultural systems to remain productive in the long term.

4.2 Indicators.

4.2.1 Nitrogen Productivity (NP)

To assess shrimp-farming systems, I developed the NP indicator. This indicator combined Nitrogen Efficiency (NE) and Ammonia Assimilative Capacity (AAC) as the product of both indicators:

(4-1) NP = (NE) (AAC)

Although NP was not a straightforward ratio of outputs over inputs as in the economic approach of TFP or TRP, the NP indicator embraced the concept of multiple input-output productivity by considering inputs and outputs that have N in common, and influence production and water quality. NP aims to indicate the long-term productive potential of aquacultural systems.

4.2.2 Nitrogen Efficiency (NE)

To assess N retention in the farmed organisms, I used NE. NE reflects the capacity of the organisms (i.e., shrimp) to retain N as biomass. NE can be calculated with either a biological (NEBiol), or an economic (NEEcon) focus. NEBiol is the ratio of N in harvest over all N inputs:

(4-2) NEBiol = N (harvest) / N (feed + fertilizer + fixation + water inflow)

NEEcon is the ratio of N outputs that have a market value over purchased N inputs:

(4-3) NEEcon = N (harvest) / N (feed + fertilizer)

In this study I focused on the calculation of NEBiol in accordance with the criteria reviewed in Chapter 2 for the multiple input-output productivity assessment. Therefore, unless indicated, in this study NE refers to the biological approach to NE measurement.

4.2.3 Ammonia Assimilative Capacity (AAC

To assess the assimilative capacity of the environment in terms of ammonia, I proposed to calculate a ratio of Total Ammonia Nitrogen (TAN) inputs to the maximum TAN concentration (below 1 $mg \cdot L^{-1}$) in the water column, reached during the farming period. I termed this indicator as Ammonia Assimilative Capacity (AAC);

(4-4) AAC = TAN input / Max TAN level (below 1 $mg \cdot L^{-1}$)

AAC reflects the capacity of the environment to maintain levels of TAN below 1 $mg \cdot L^{-1}$ at different rates of TAN input, during the whole farming period.

A "safe range" of unionized ammonia nitrogen (NH_3-N) for penaeid shrimp is that below 0.1 $mg \cdot L^{-1}$ (Chin and Chen 1987; Wickins 1976b), equivalent to 1.0 $mg \cdot L^{-1}$ of TAN at a pH slightly above 8.2, a salinity between 27‰ and 33‰, and a temperature of 28 °C. In semi-intensive shrimp farms, the pH fluctuates normally between 7.9 to 8.5 (Briggs and Funge-Smith 1994; Paez-Osuna, Hendrickx-Reners, and Cortes-Altamirano 1994). At a pH of 8.4, 0.1 mg·L $^{-1}$ of unionized ammonia is equivalent to 0.8 mg·L $^{-1}$ of TAN (Table 4-1). Therefore I assumed that levels of 1mg·L $^{-1}$ of TAN, or above, were of high toxicity to shrimp and therefore TAN levels should be maintained below that level at any time (Lorenzen, Struve, and Cowan 1997). Therefore 1 mg·L $^{-1}$ of TAN represented the threshold in my indicator.

Table 4-1. Concentration of Total Ammonia Nitrogen (TAN) equivalent to 0.1 mg·L $^{-1}$ of NH_3-N in water at 28° C, at a constant pressure of 1 atm and different values of salinity and pH (From Whitfield 1974).

	Salinity				
pH	*0 ‰*	*24 ‰*	*27 ‰*	*30 ‰*	*33 ‰*
6.8	22.3	26.1	26.7	27.2	27.8
7.0	14.1	16.5	16.9	17.2	17.6
7.2	8.9	10.5	10.7	10.9	11.1
7.4	5.7	6.6	6.8	6.9	7.1
7.6	3.6	4.2	4.3	4.4	4.5
7.8	2.3	2.7	2.8	2.8	2.9
8.0	1.5	1.7	1.8	1.8	1.9
8.2	1.0	1.1	1.2	1.2	1.2
8.4	0.7	0.8	0.8	0.8	0.8

4.2.4 Goals of aquaculture: maximize NE and AAC

According to my indicators, there were two aquaculture goals with respect to nitrogen: 1) maximize the amount of N that is harvested, reflected by NE; and 2) maximize the assimilative capacity of the environment in terms of ammonia, reflected by AAC.

4.3 Tools for the analysis.

To conduct the nitrogen-based analysis of shrimp farming, I utilized a simulation model and a N budget model. The simulation model aimed to assess the N partitioning of managed and unmanaged N flows, particularly internal N flows related to metabolic processes (e.g., shrimp growth and ammonia excretion), the assimilation of nitrogenous wastes by phytoplankton, and the exit of N from the system through volatilization, water exchange and sedimentation. The objective of using the simulation model was to quantify the consequences of changes in shrimp farming management practices on NE and on AAC. Such quantification allowed relating the indicators, NE and AAC, into a single, wider indicator, NP, which assessed shrimp farming behaviour in relation to the partitioning of N into useful outputs and nitrogenous waste.

The N budget model aimed to quantitatively relate N flows into and from the pond system (i.e., N inputs and outputs) regardless of internal processes. The objective of using the N budget was to quantify the relative contribution in shrimp farming of N inputs such as fertilizer and fixation which were not considered in the simulation model.

4.3.1 Simulation model

I built and ran a shrimp pond simulation model by using Stella® 5.1 Research Version, a simulation modelling software, and Microsoft Excel® 97, a spreadsheet software. The data to build the model was obtained mostly from published information, applicable to the case in Northwest Mexico, and to the farming range examined. The simulation was based mainly on two management variables that characterize intensification of aquacultural systems: stocking density (*Do,* in number of shrimp per square meter) and water exchange rate (*W*, as a fraction of pond volume per day). The range of the management variables that the model simulated corresponded to the *Do* range (1-50 shrimp·m $^{-2}$) used from extensive to intensive production methods in outdoor ponds (Table 3-2), and to an ample range of *W* (0 -1·d $^{-1}$). Although the common use of *W* is below 0.2·d $^{-1}$, I simulated a wider range to explore the implications of higher *W* values on the nitrogen-based indicators (NE, AAC and NP) of shrimp farming behaviour.

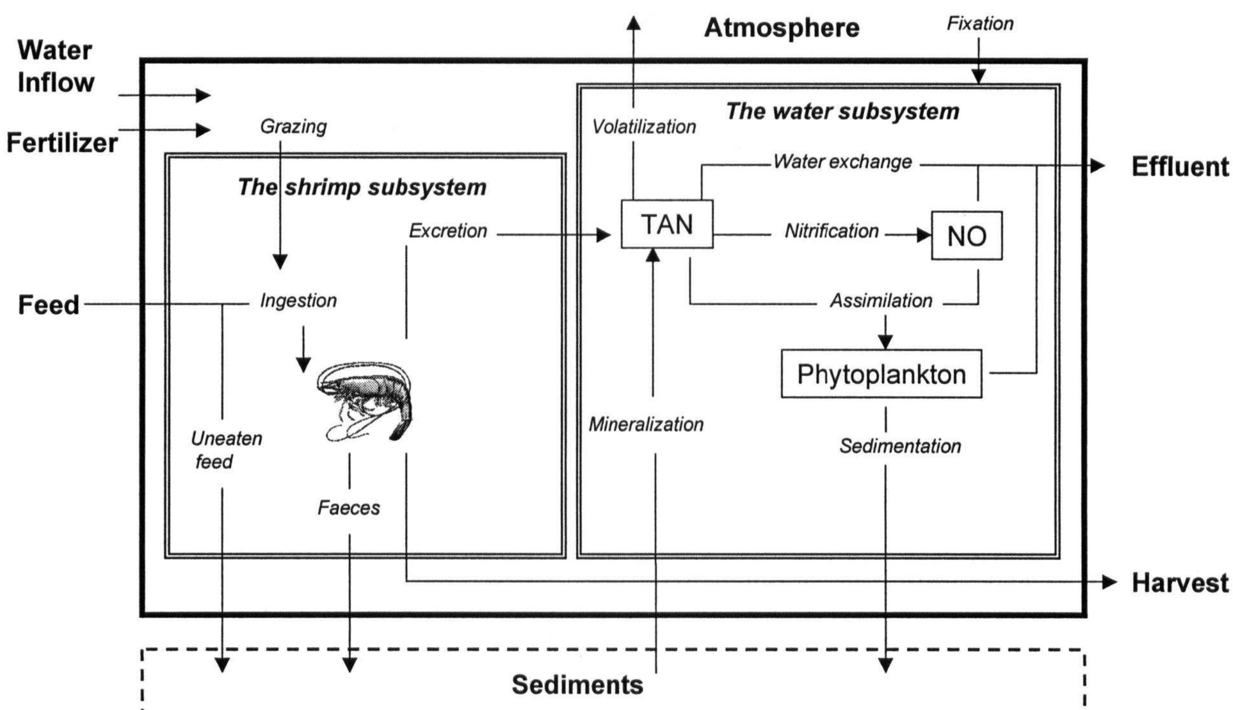

Fig. 4-2. The shrimp-farming pond system, divided into shrimp and water subsystems, and the nitrogen flows considered in the simulation modelling. TAN = total ammonia nitrogen; NO = nitrites and nitrates. All N from feed and faeces was assumed to be in particulate form, and to become sedimented.

The shrimp-farming pond simulation model was built from the merging of two models: the shrimp subsystem model, which I developed, and the water subsystem model, a modification of an existing N dynamics model (Lorenzen, Struve, and Cowan 1997). Both subsystems were connected through the production of ammonia, which entered the water column through the excretion by the shrimp and the mineralization of uneaten feed and faeces (Fig. 4-2).

The model ran in 1-d (24-h) steps, for a period of 120 d, which is a typical farming period of the shrimp, *Litopenaeus vannamei*, in Northwest Mexico. All along the farming period, the pond model assumed optimal values of environmental parameters (i.e., 28 °C, 30 $^{o}/_{oo}$ and a pH = 8.2) which have been reported for the farming of *L. vannamei*. Also, the model assumed that oxygen levels in the pond were appropriate (> 5 $mg \cdot L^{-1}$).

4.3.1.1 The shrimp subsystem

This model described the growth of the Pacific white shrimp, *L. vannamei*, based on the retention of N in shrimp biomass, where the somatic growth was equivalent to the amount of N retained. One kg of shrimp weight gain was equivalent to 178.5-g protein, and to 28.56-g of N, or 2.85% of N on a wet weight basis, according to *L. vannamei* body composition (Boyd and Teichert-Coddington 1995). Nitrogen in protein (PN) was calculated by assuming a proportion of 6.25 of crude protein (CP) to N, equivalent to 16% of N in CP (Holland et al. 1991).

The shrimp model considered formulated feeds and grazing as sources of N to the subsystem. The model represented the flow of N from feed (FN) and its partitioning into N in uneaten feed (UFN) and ingested N (IN). IN also derived from grazing. In turn, IN was partitioned into digested N (DN), N in faeces (fN), N retained in biomass (RN) and N excreted as ammonia (EN).

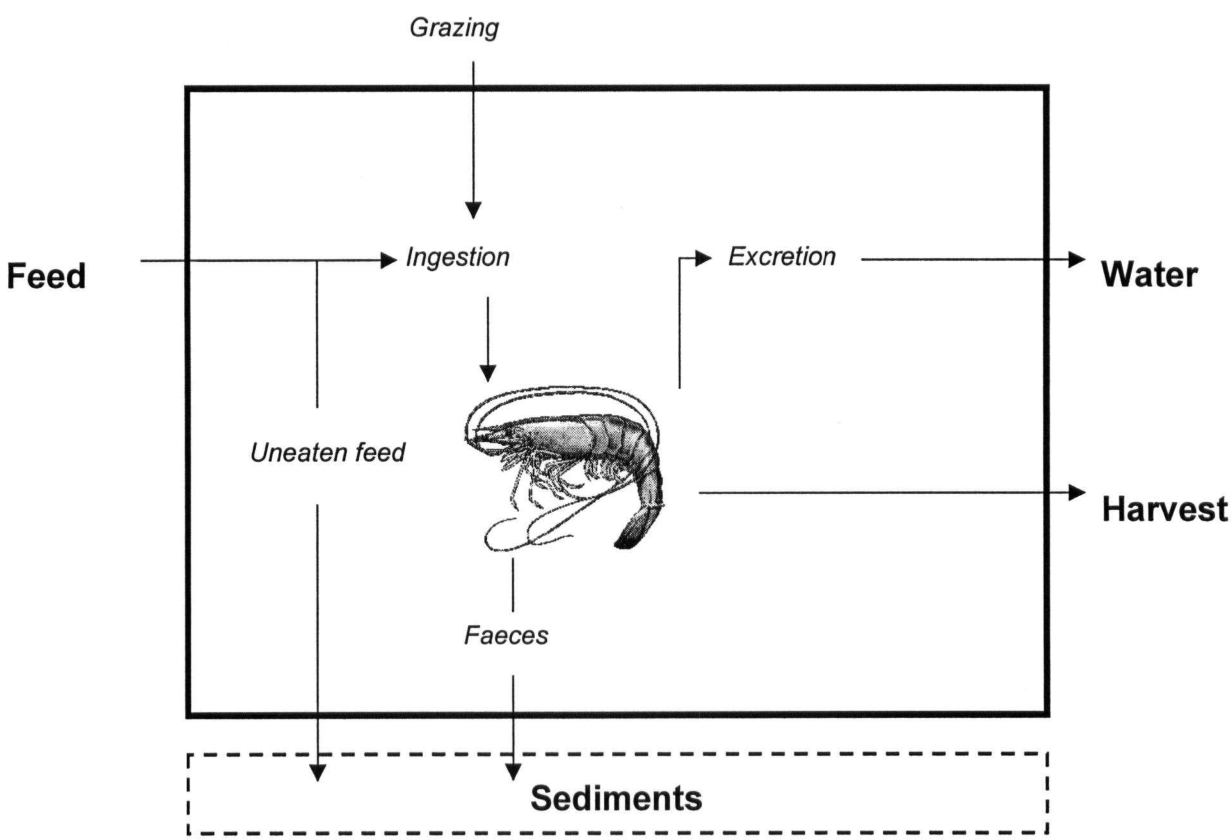

Fig. 4-3. The shrimp subsystem. All N from uneaten feed and faeces is assumed to be in particulate form, and to become sedimented

Formulated feeds, containing 30% crude protein (FP, feed protein) are common in the extensive to intensive shrimp farming range, and I considered it as a baseline in the model. Such protein content is reported on a dry matter basis (oven-dry feed) which was assumed to be 88% (DMF), according to a typical shrimp feed composition chart (Purina 1998). The amount of feed supplied per day was calculated by using a typical feeding chart (Aquaculture Zeigler International 1997; Purina 1998) and the feeding rate (FR) was calculated as a fraction of individual shrimp biomass (ISB), according to an equation derived from the feeding chart:

(4-5) $$\text{Feeding rate (FR)} = 0.12 \cdot (\text{ISB})^{-0.5}$$

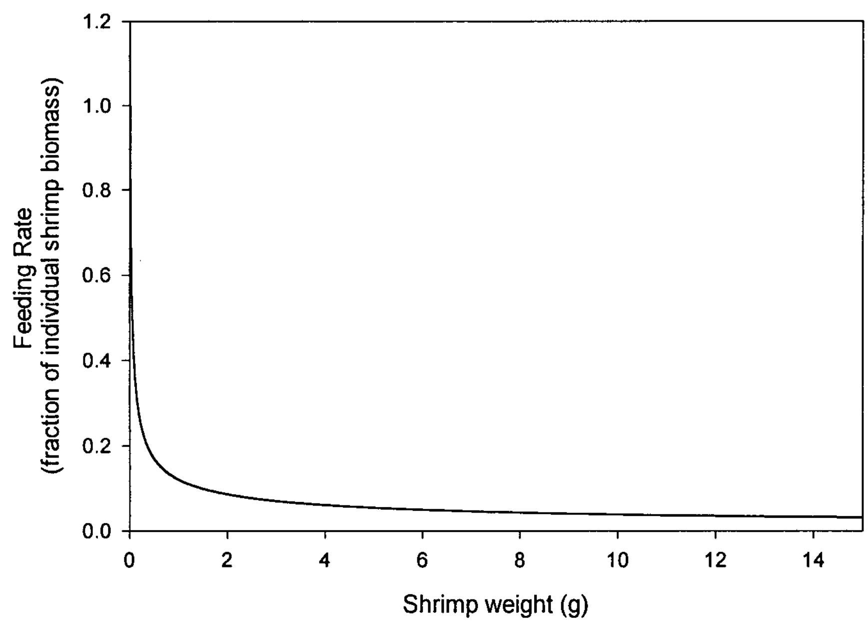

Fig. 4-4. Typical feeding rate (FR) for *L. vannamei* as a fraction of individual shrimp biomass (ISB).

FR was multiplied by the corresponding shrimp biomass (SB) per ha, to calculate total amount of feed fed. SB was calculated by multiplying ISB by the surviving shrimp density (SD):

(4-6) Total amount of feed fed = (FR) (SB)

where:

(4-7) SB = (ISB) (SD)

The amount of air-dry feed (ADF) was calculated therefore as:

(4-8) ADF = (FR) (ISB) (SD)

and the amount of N in the feed (FN) was calculated as:

(4-9) FN = (ADF) (DMF) (FP) (PN)

Uneaten feed, or the feed that was not ingested by the shrimp and the proportional amount of N (UFN), was considered to be a constant proportion (15%) of the feed fed, according to common reported values (Primavera 1993). Ingested N (IN) was calculated as:

(4-10) IN = FN – UFN

Ingested N from formulated feeds was assumed to account for 75% of the ingested N, while the remaining 25% derived from grazing. The proportion of N from grazing was a conservative assumption of the wide range of values (0 - 60%) reported in the literature

(Anderson, Parker, and Lawrence 1987; Goddard 1996).

Total IN was partitioned into digested N (DN), and undigested N (UN). DN was either retained in the shrimp biomass (RN), or excreted as ammonia (EN). Protein digestibility of the feed (PD) was assumed to be 90% (Montoya et al. 1999). Undigested N (UN), equal to:

(4-11) $$UN = 100 - PD$$

ended as faeces (fN). RN was assumed to be 30% of IN, a value that corresponded with reported values for shrimp (Montoya et al. 1999) as well as with the average for a wide range of aquatic and terrestrial animals (Bowen 1987). Excreted N (EN) was the difference between IN and (fN + RN)

(4-12) $$EN = IN - (fN + RN)$$

I assumed in the model that no N was utilized for maintenance, as the protein retention in marine shrimp appears to be high (> 90%) (Forster and Gabbott 1971; Montoya et al. 1999).

Nitrogen exited the subsystem in four forms: As faeces (fN) and uneaten feed (UFN) into the sediments, as harvest (SN), and as ammonia (EN) into the water column.

The density of shrimp (SD) at time t was governed by the exponential mortality function:

(4-13) $SDt = Do \cdot \exp^{(-m\,t)}$

where:

Do = initial stocking density (shrimp·m $^{-2}$); and

m = mortality rate (d $^{-1}$).

4.3.1.2 The water subsystem.

The water subsystem model was based on ammonia (TAN) entering the system from shrimp excretion (EN), as well as from the mineralization of uneaten feed and faeces (MUF). More than 90% of N waste in aquaculture, including shrimp farming, enters the water column as ammonia (Burford and Glibert 1999; Hargreaves 1998; Lorenzen 1999), and therefore I considered ammonia as the sole N species entering the water subsystem; 80% of the N in uneaten feed and faeces from the shrimp subsystem was assumed to enter the water column as ammonia, through mineralization, according to reported estimates (Hargreaves 1998).

TAN in the water column followed four different paths: assimilation by phytoplankton, volatilization (*v*), nitrification (*n*) or discharge through water exchange (*W*). Ammonia may be converted through nitrification into nitrite and nitrate (both referred as NO) which may be assimilated by phytoplankton or discharged through water exchange. Phytoplankton may become sedimented or discharged through water exchange.

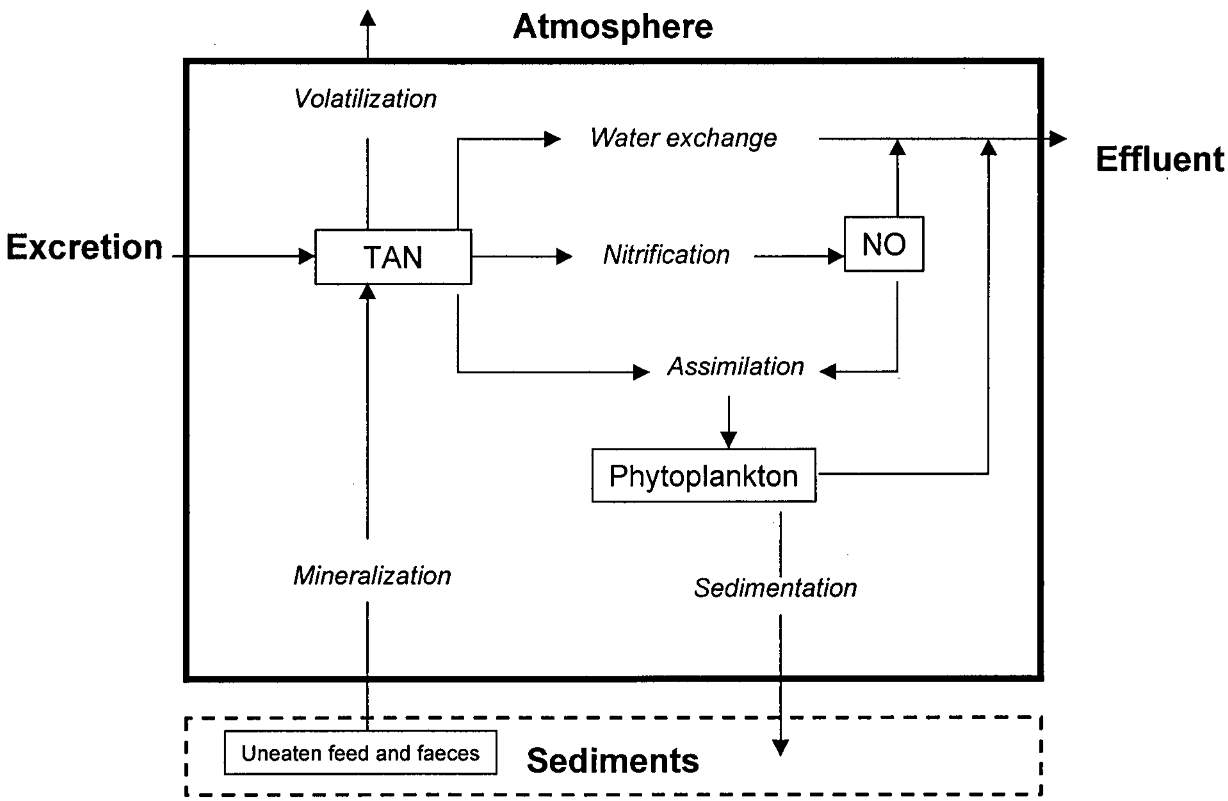

Fig. 4-5. The water subsystem. TAN = total ammonia nitrogen; NO = nitrites and nitrates.

The water subsystem model consisted of three main stocks, or state variables, in the water column. Total ammonia nitrogen (TAN), nitrite and nitrate (NO), and the proportion of chlorophyll *a* (Chl) present in phytoplankton.

State variables in the model were related by the following differential equations:

(4-14) $$dTAN/dt = AI - (n+v+W) \cdot TAN - g \cdot c \cdot Chl \cdot (TAN/(TAN+NO))$$

(4-15) $$dNO/dt = n \cdot TAN - W \cdot NO - g \cdot c \cdot Chl \cdot (NO/(TAN + NO))$$

(4-16) $dChl/dt = g \cdot Chl - (s + W) \cdot Chl$

where:

TAN = concentration of ammonia ($mg \cdot L^{-1}$)

NO = concentration of nitrite-nitrate ($mg \cdot L^{-1}$)

Chl= concentration of chlorophyll *a* ($mg \cdot L^{-1}$)

AI= ammonia input from shrimp excretion, EN; and mineralization of uneaten feed and faeces, MUF ($mg \cdot L^{-1} \cdot d^{-1}$)

n= nitrification rate (d^{-1})

v= volatilization rate (d^{-1})

W= water exchange rate (d^{-1})

t = time (d)

g= growth rate of phytoplankton (d^{-1}) $= g_{max} L_{light} L_N L_P$

g_{max} = the maximum growth rate of phytoplankton in the absence of limitation (d^{-1})

L_{light} = light limitation coefficient = $e/k \{\exp [-(I_o/I_{sat}) \exp (-kz)] - \exp (- I_o/I_{sat})\}$

L_N = nitrogen limitation coefficient = $(TAN/NO)/((TAN + NO) + K_{Sn})$

L_P = phosphorus limitation coefficient = $DRP/(DRP + K_{Sp})$

e = base of the natural logarithm

I_o = incident light at the surface of the pond ($E \cdot m^{-2} \cdot d^{-1}$)

I_{sat} = saturation light intensity for algal growth ($E \cdot m^{-2} \cdot d^{-1}$)

k = extinction coefficient (m^{-1}) = $K_{Chl}\, Chl + K_{other}$

z = pond depth (m)

K_{Chl} = extinction per unit concentration of chlorophyll ($m^{-1} \cdot mg^{-1}$)

K_{other} = extinction due to non-chlorophyll sources (m^{-1})

DRP = dissolved reactive phosphorus ($mg \cdot L^{-1}$)

K_{Sn} = half saturation constant of nitrogen ($mg \cdot L^{-1}$)

K_{Sp} = half saturation constant of phosphorus ($mg \cdot L^{-1}$)

A detailed version of the model is available (Lorenzen, Struve, and Cowan 1997). In the present study, I used the model parameter values that best described N dynamics in a low-intensity farm in Thailand (Lorenzen 1999). Such values were as follows:

Sedimentation rate s (d^{-1})	0.35
Phytoplankton growth rate g_{max} (d^{-1})	1.0
Extinction from non-Chl K_{other} (m^{-1})	2.60
Saturation light intensity I_{sat} ($E \cdot m^{-2} \cdot d^{-1}$)	37.6
Surface light intensity I_o ($E \cdot m^{-2} \cdot d^{-1}$)	40.0
Depth z (m)	1.0
Nitrogen to Chl ratio c	8.9
Nitrogen half-saturation K_{SN} ($mg \cdot L^{-1}$)	0.095
Volatilization rate v (d^{-1})	0.17
Phosphorus half-saturation K_{Sp} ($mg \cdot L^{-1}$)	0.00001
Nitrification rate n (d^{-1})	0.1

For g_{max} I used a 1.0 value, instead of 1.3, according to average values reported for phytoplankton growth in northwestern Mexican shrimp farms (Guerrero-Galvan et al.

1999). I assumed in this model that no evaporation occurred and sediments were considered a different compartment of the pond system, in a similar approach to those utilizing the Ecopath model (Dalsgaard and Oficial 1998).

4.3.2 *Nitrogen budget model.*

To construct a N budget model, I assumed a steady-state, where N in = N out, from the mass conservation principle (Fig. 4-6). I estimated the N budget using two different approaches: economic and biological. In the economic approach I considered N inputs and outputs with a market value (i.e., feed, fertilizer and shrimp harvest), while in the biological approach I considered priced and unpriced N inputs and outputs (i.e., feed, fertilizer, fixation, harvest, volatilization, effluent, sedimentation, and mineralization).

Nitrogen input from fixation was assumed as 25 $mg{\cdot}m^{-2}{\cdot}d^{-1}$, based on data for aquacultural ponds in tropical areas (Acosta-Nassar, Morell, and Corredor 1994; Lin, Tansakul, and Apihapath 1988). Therefore, the total amount of N from fixation during the 120 d of farming was equal to 30 $kg{\cdot}ha^{-1}$.

Nitrogen input from initial shrimp stock (i.e., postlarvae) have represented less than 1% of total N inputs in shrimp farming N budgets (Briggs and Funge-Smith 1994; Martin et al. 1998; Paez-Osuna et al. 1997) for which I considered it as negligible in this study. Similarly, other N inputs such as rainfall and run-off were considered as negligible. Rain may be an important source of N in shrimp ponds, but it varies much in temporal and spatial scales (Hopkins et al. 1993).

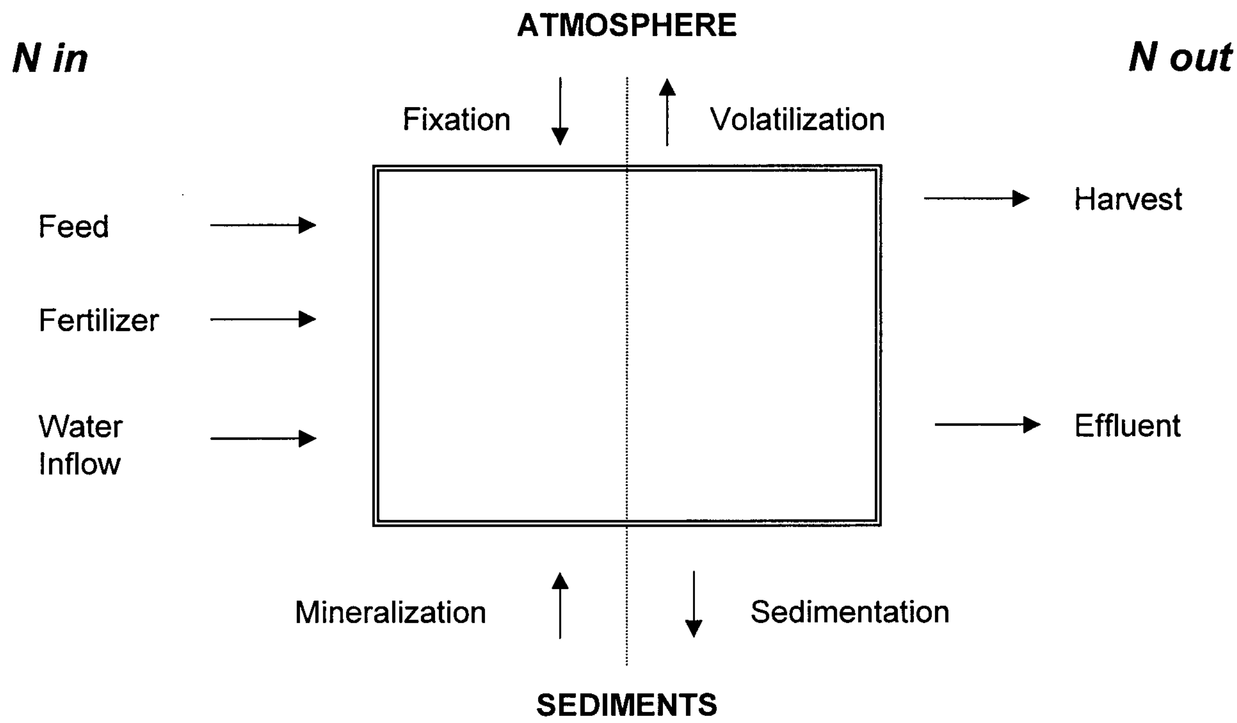

Fig. 4-6. Conceptual model of a N budget in a shrimp-farming pond. Sediments were considered as a separate compartment for ease of analysis.

4.4 The baseline case

To compare the simulation results and N budgets I used a typical semi-intensive shrimp-farming pond in Northwest Mexico, and its typical management practices.

Common *Do* used vary between 12 – 20 shrimp·m $^{-2}$, while *W* vary between 0.02 and 0.07·d $^{-1}$. For this study, I assumed a *Do* = 15 shrimp·m $^{-2}$, and a *W* = 0.04·d $^{-1}$ based on values reported for semi-intensive shrimp farming in Northwest Mexico (Guerrero-Galvan et al. 1999; Paez-Osuna et al. 1997).

The level of protein in the feed was assumed as 30%, as already discussed in the description of the simulation model. The amount of fertilizer assumed, 35 kg urea$\cdot$ha $^{-1}$ cycle $^{-1}$, was also a typical value used in shrimp farms in Northwest Mexico ((Guerrero-Galvan et al. 1999). The amount of N in fertilizer was calculated by multiplying the urea fertilizer amount by 0.45 (Boyd 1990). Thus, the total amount of N from fertilizer was equal to 15.75 kg$\cdot$ha $^{-1}\cdot$cycle $^{-1}$

A farming period of 120 d and a survival rate of 0.7 were used in the baseline case, which corresponded to values commonly reported for the farming of *L. vannamei* in Northwest Mexico (FIRA 1998; Paez-Osuna et al. 1997).

CHAPTER 5. RESULTS AND DISCUSSION

In this Chapter I present the major findings of my nitrogen-based approach to the analysis of shrimp farming. The approach was based on the simulation of important management practices, and their effect on Nitrogen Productivity (NP), the performance indicator that I developed to assess the long-term productivity potential of shrimp farming. Factors affecting Nitrogen Efficiency (NE) and Ammonia Assimilative Capacity (AAC), the indicators from which NP was calculated were also examined. The results of the model simulations were compared to the Mexican case and the implications are discussed. Nitrogen budgets constructed under economic and biological approaches are also presented, as well as the results and discussion of related N interactions of shrimp farming, predicted by the model simulations and which are relevant to this study.

5.1 The indicators: best operating conditions and implications

I used the model to simulate changes in shrimp stocking density (*Do*) and water exchange rate (*W*), the two main management variables that I previously identified as highly characteristic of the level of intensification in shrimp farming (Sec. 3.2.1). The model simulations allowed assessing the effect of management practices on the performance indicator NP, through the calculation of NE and AAC. Factors that affected NE and AAC were analyzed and the sensitivity of the model predictions was conducted by varying each factor by 50% (i.e., simulations were set at 1.5, 1.0 and 0.5 times its baseline value), using the Stella® software tool for sensitivity analysis. An exception was the level of protein in the feed for which values of other frequently used protein level in feeds (25 and 35%) were tested against the 30% protein, baseline value of the model.

5.1.1 *Nitrogen Productivity (NP)*

The highest value of NP (i.e., best operating conditions) corresponded to: 1) the highest *Do* (50 shrimp·m $^{-2}$) and the highest *W* (1·d $^{-1}$) simulated (Fig. 5-1) or; 2) a *Do* = 37.25 shrimp·m $^{-2}$ and *W* =0, for *W* values below 0.2·d $^{-1}$ (Fig. 5-2), assuming a survival rate of 0.7 as in the baseline case. I ran the simulations in the low range (0 – 0.2·d $^{-1}$) of *W* as this represents the common range in most extensive to intensive, outdoor shrimp farming in Northwest Mexico (Paez-Osuna et al. 1997) and worldwide (Hopkins et al. 1993).

Changing the survival rate also affected the value of NP (Fig. 5-3). The impact of survival rate on NP was in two forms; 1) by influencing the value of NE, or the amount of N in shrimp harvest (Fig. 5-9); and 2) by influencing the amount of TAN that entered the water column during the farming period through excretion and mineralization of uneaten feed and faeces (Fig. 5-23).

Using the Solver tool in the Excel® software, I calculated the optimal *Do* and *W* that corresponded to the highest NP value, at different survival rates (Table 5-1). A *Do* = 50 shrimp·m $^{-2}$ and a *W* = 1·d $^{-1}$ corresponded to the highest NP for all survival rates (from 1 to 0.5), for the full range (0 – 1·d $^{-1}$) of *W* (Table 5-1). In the lower range of *W* (0 – 0.2·d $^{-1}$), the highest NP value corresponded to a *Do* that varied from 28.99 to 47.14 shrimp·m $^{-2}$, for the different survival rates, and to a *W* = 0 for all survival rates (Table

5-1).

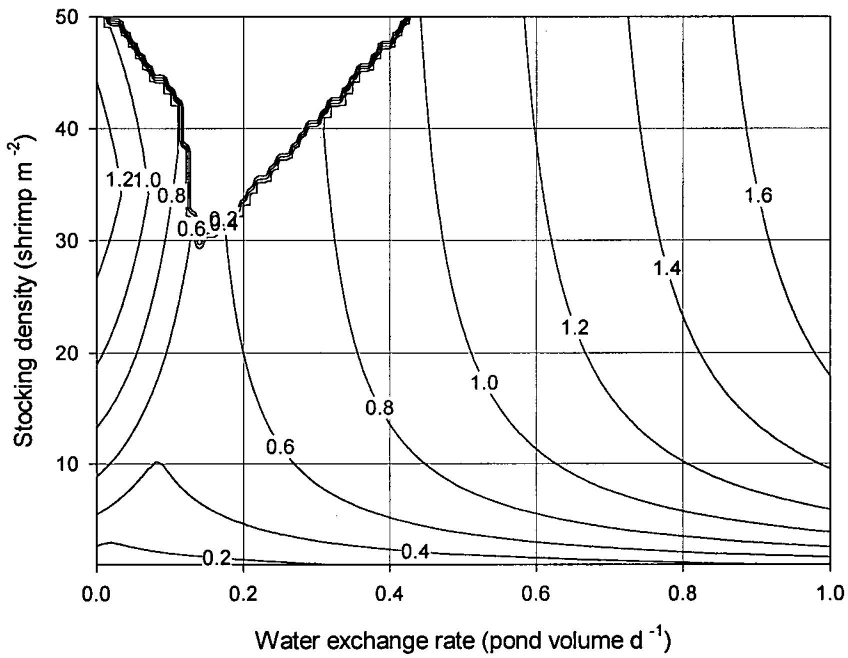

Fig. 5-1. Nitrogen Productivity (NP) as a function of stocking density (*Do*) and water exchange rate (*W*) in shrimp farming, assuming a survival rate of 0.7 as in the baseline case. Values on the upper-left corner were above the threshold of 1 mg·L $^{-1}$ of TAN in the AAC indicator. NP = (NE) (AAC).

The model predicted in general, according to the NP indicator, that a better performance of shrimp farming resulted at 1) higher *Do* and higher *W*, or; 2) higher *Do* and lower *W* when compared to the Mexican baseline case (Fig. 5-2). Adopting best management practices (i.e., with the highest NP) implied that the Mexican case, a semi-intensive production method, would become 1) more intensive with regard to both *Do* and *W*, in the first case; or 2) more intensive with regard to *Do* and less intensive with regard to *W* in the second case, according to general criteria to categorize shrimp production (Table 3-2).

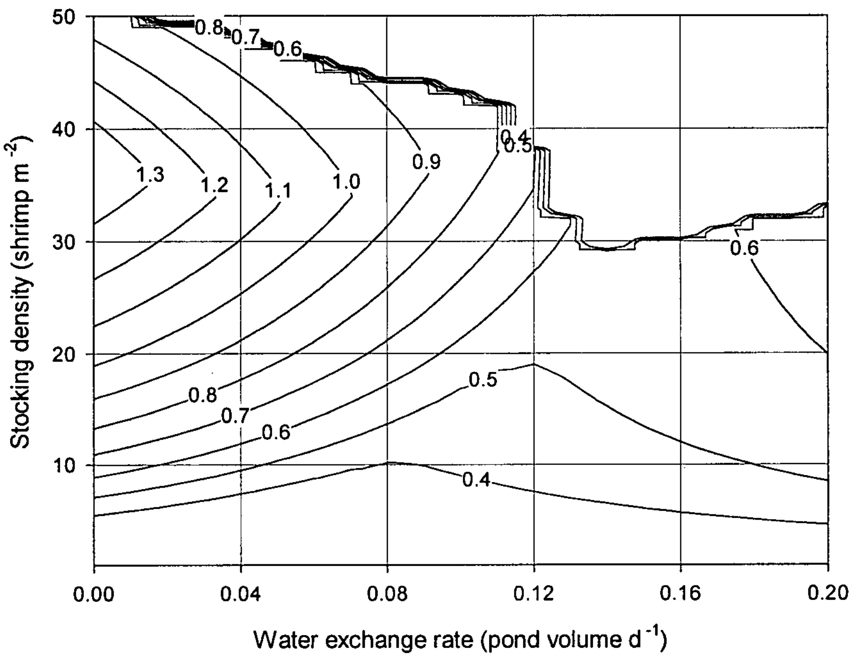

Fig. 5-2. Nitrogen Productivity (NP) as a function of stocking density (*Do*) and water exchange rate (*W*) in shrimp farming, for *W* values between 0 and 0.2. Survival rate was assumed as 0.7, as in the baseline case. Values on the upper-right corner were above the threshold of 1 mg·L $^{-1}$ of TAN. NP = (NE) (AAC).

5.1.1.1 *Implications of higher water exchange rate*

NP values higher than that for the baseline case were obtained only with *W* higher than 0.25·d $^{-1}$, either with higher or lower *Do*. For example, a NP value equal to 0.7141, as in the baseline case, corresponded to *Do* = 35 shrimp·m $^{-2}$ and *W* = 0.2531·d $^{-1}$, or to Do = 10 shrimp·m $^{-2}$ and *W* = 0.3715·d $^{-1}$ (Fig. 5-1).

In general, increasing *W* in the range from 0 to 0.2·d $^{-1}$ under a constant *Do*, decreased the value of NP. This NP trend was associated to the behaviour of the AAC indicator in the same *W* range, in which a higher *W* resulted in higher ammonia levels in the pond, reducing the value of AAC. I discuss this AAC behaviour further in Sec. 5.1.3.

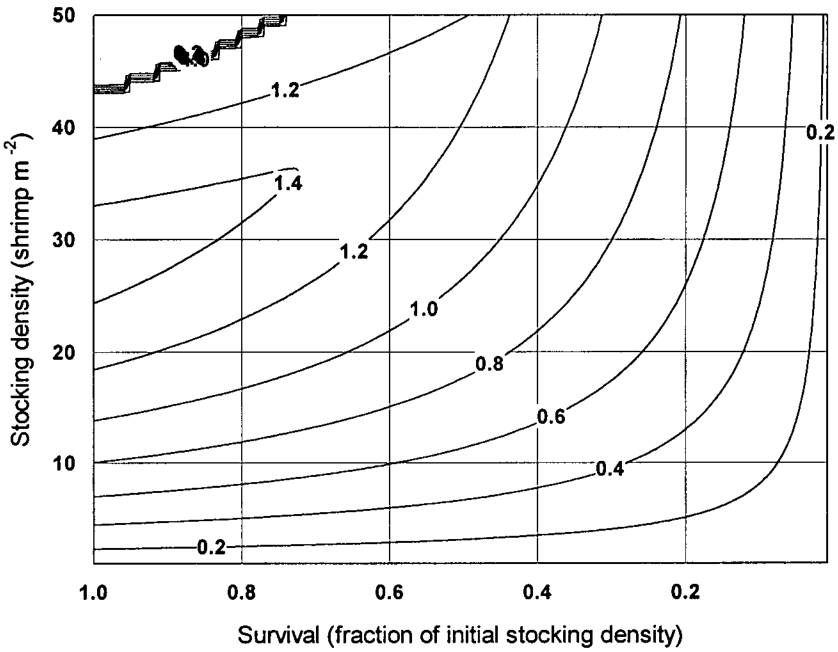

Fig. 5-3. Nitrogen Productivity (NP) as a function of stocking density (*Do*) and survival rate in shrimp farming, under no water exchange (*W*). Values on the upper-left corner were above the threshold of 1 $mg{\cdot}L^{-1}$ of TAN. NP = (NE) (AAC)

Table 5-1. Stocking density (*Do*) and corresponding water exchange rate (*W*) for maximum Nitrogen Productivity (NP), or optimal case, at different survival rates, as predicted by the model

	Water exchange (*W*)							
	Full range (0-1 d^{-1})				Low range (0-0.2 d^{-1})			
Survival rate	**NP**	**Do**	***W***	***Water usage $m^3{\cdot}t^{-1}$ shrimp**	**NP**	**Do**	***W***	***Water usage $m^3{\cdot}t^{-1}$ shrimp**
1.0	1.91	50	1	180037.7	1.52	28.99	0	2566.2
0.9	1.87	50	1	193407.1	1.49	31.24	0	2558.3
0.8	1.83	50	1	209373.3	1.44	33.97	0	2546.9
0.7	1.79	50	1	228388.1	1.39	37.25	0	2533.6
0.6	1.74	50	1	252987.2	1.33	41.55	0	2516.0
0.5	1.68	50	1	284512.2	1.26	47.14	0	2494.0

*Includes water required to fill the pond

W values in the range of 0.1 to 0.25$\cdot d^{-1}$ have been considered a useful practice to ameliorate water quality problems in shrimp farming ponds, such as those generated by phytoplankton crashes (Funge-Smith and Briggs 1998). However, W values higher than $0.25 \cdot d^{-1}$ may be impractical because of negative physical effects like scouring of the pond bottom that may increase turbidity and potentially stress the organisms. Although the pumping cost represents a relatively small fraction of total cost in shrimp farming (Lee and Wickins 1992; Shang, Leung, and Ling 1998), high values of W may represent a substantial part of the production cost that merits analysis. Increasing W from zero to $0.15 \cdot d^{-1}$ raised energy costs by 31.5% in the farming of *L. vannamei* (Hopkins et al. 1996).

Moderate increases in W led to a considerable higher proportion of effluent N in dissolved form (Fig. 5-36). Having N in dissolved form in the effluent has a greater potential for the recovery of nitrogenous wastes (Lorenzen 1999), and the recovery of N waste appears to be important from the ecological point of view. I discuss this issue further in Sec. 5.3.3.

5.1.1.2 Implications of lower water exchange rate

Comparing to the baseline case, the model predicted that by increasing Do towards 37.25 shrimp$\cdot m^{-2}$ and reducing W towards zero, the NP value was maximized (Table 5-1). With the same Do (37.25 shrimp$\cdot m^{-2}$), an equal NP value would be obtained only with W as high as $0.74 \cdot d^{-1}$ (Fig. 5-1).

My model predicted that for the Mexican baseline case, the maximum levels of ammonia reached during the farming period (Fig. 5-4) were below those reported as toxic for penaeid shrimp (Table 4-1), under the assumption of a pH of 8.2 (Sec. 4.3.1). A simulation of the optimal case (Do =37.25 shrimp·m $^{-2}$ and W =0) showed higher TAN levels, when compared to the baseline case, with an increasing trend at the end of the farming period (Fig. 5-4).

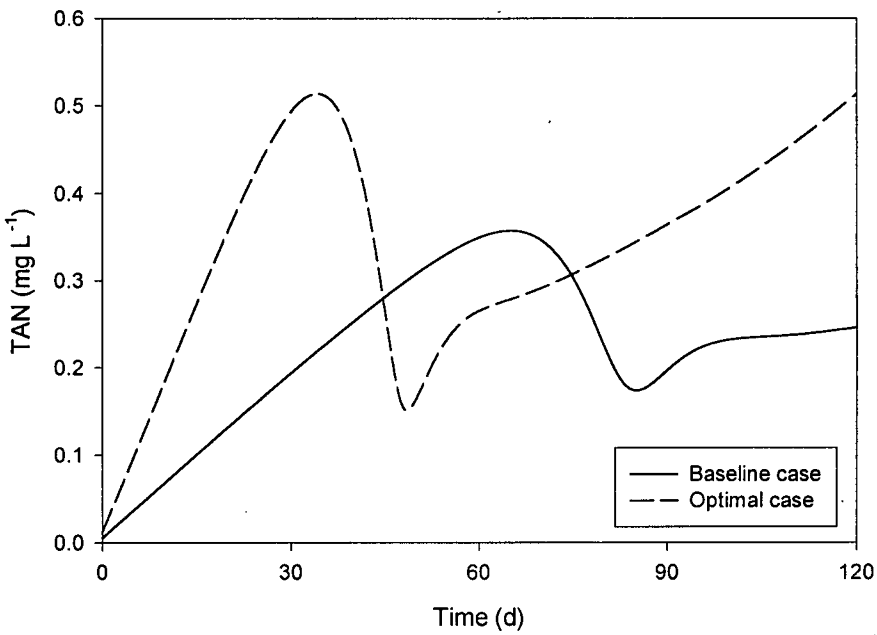

Fig. 5-4. Comparison of ammonia (TAN) levels in the pond for the baseline case and the optimal case, as predicted by the simulation model. Survival rate = 0.7

The amount of TAN inputs increased from excretion of a higher number of shrimp, and the associated mineralization of a higher amount of faeces and uneaten feed. Under these conditions, unstable phytoplankton populations may result in a considerable increase in ammonia levels in the pond, as the simulation of different growth rates of phytoplankton predicted (Fig. 5-5). This might be complicated if, for example, high levels of ammonia were present in the water inflow. Thus, although ammonia for the optimal case was also below toxic levels, there might be an increased risk of water quality deterioration at higher *Do* values.

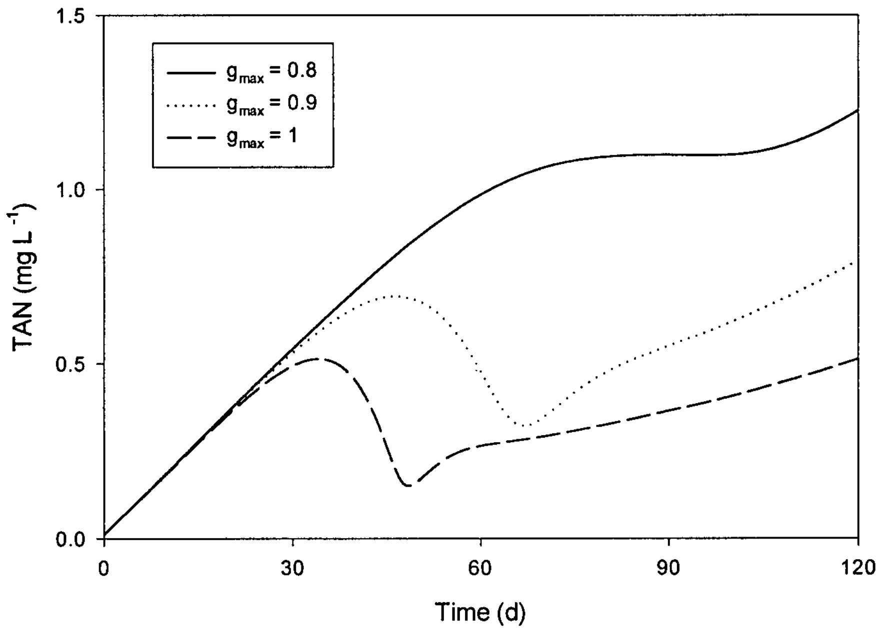

Fig. 5-5. Ammonia (TAN) levels in the pond at different rates of phytoplankton growth, as simulated for the optimal case. g_{max} refers to the maximum growth rate of phytoplankton and g_{max} = 1 was the default value in the model.

There has been a recent interest in the adoption of reduced or zero water exchange regimes in shrimp farming in outdoor ponds (Funge-Smith and Briggs 1998; Green et al. 1998; Hopkins et al. 1996). That interest has been driven mainly by the deterioration of surrounding waters (e.g., estuaries and canals), some evidence that water exchange is associated to disease outbreaks, and the cost of pumping.

Experiments with shrimp farming at *Do* in the range of 20 to 40 shrimp·m $^{-2}$, and no water exchange, have resulted in moderate to high survival. *L. setiferus* farmed at a *Do* = 22 shrimp·m $^{-2}$ had a survival rate of 0.81 (Hopkins et al. 1993), while *L. vannamei* at a *Do* = 38.2 shrimp·m $^{-2}$ resulted in a survival rate of 0.91 (Hopkins et al. 1996). However, the farming of *L. setiferus* at Do = 44 shrimp·m $^{-2}$ led to *en masse* mortality (Hopkins et al. 1993). In the case of no water exchange (i.e., closed systems) there is indication of high organic loading which may lead to stressful conditions (Funge-Smith and Briggs 1998). Therefore, increasing shrimp stocking densities should be approached with caution.

Maximizing survival of the farmed organisms represents an important goal of food production systems. Assuming that shrimp survival rates higher than 0.9 were obtained, best management practices would be achieved by adopting *Do* values around 30 shrimp·m $^{-2}$ and no water exchange (Table 5-1). This *Do* values concurred with the 20 – 30 shrimp·m $^{-2}$, *Do* range recommended as a maximum in the farming of shrimp in outdoor ponds (Briggs 1994). Economic analyses have indicated that shrimp production with survival rates lower than 0.5 was not viable in shrimp farming enterprises in Northwest Mexico (FIRA 1998). Therefore I considered an overall viability (both biophysical and economic) only to predictions with NP values associated to survival rates of 0.5 or higher.

Reducing *W* to zero may have other important implications that my model was not able to evaluate. Shrimp ponds in semiarid regions may have considerable evaporation leading to an increased salinity beyond appropriate ranges for shrimp farming (Bray, Lawrence, and Leung-Trujillo 1993). *W* values lower than $0.05 \cdot d^{-1}$ have negatively affected the growth and survival of *L. vannamei* in the Mexican State of Sonora (Martinez-Cordova 1995).

5.1.2 Nitrogen Efficiency (NE)

5.1.2.1 Specification of nitrogen inputs

The value of NE was dependent on the specification of the N inputs of interest, as previously discussed in Sec. 2.4.1. I calculated NEEcon, or the efficiency of priced N inputs, by considering; 1) N from feed only, NEEcon (Feed only); and 2) N from feed and fertilizer, NEEcon (Feed + fertilizer). Similarly, I calculated NEBiol, or the efficiency of priced and unpriced N inputs by considering; 1) no grazing, NEBiol (No grazing); and 2) grazing, NEBiol (Grazing). To calculate NE, I included N inputs from fertilizer (for NEEcon), and fertilizer and fixation (for NEBiol) which were not part of the simulation model, and therefore they were not reflected in shrimp harvest.

NEEcon values ranged between 0.12 and 0.29, while NEBiol values ranged between 0.06 and 0.27. These values fell mostly within the range of reported values, or those calculated from reported N budgets, of NE in shrimp farming (Briggs and Funge-Smith 1994; Hopkins et al. 1993; Martin et al. 1998; Paez-Osuna et al. 1997) fish (Bleken and Bakken 1997) and terrestrial animal production (Van der Hoek 1998).

The more intensive the farming (i.e., higher *Do*), feed became increasingly the dominant

N input influencing the value of NE (Fig. 5-6 and Fig. 5-7), consistent with other findings in both aquaculture and agriculture (Table 3-3). The difference between NEEcon (Feed only), NEEcon (Feed + fertilizer) and NEBiol (No grazing) tapered off towards the more intensive farming side of the simulation. However, the proportion between NEEcon (Feed + fertilizer) and NEBiol (Grazing) remained unchanged because grazing was considered in the model as a constant proportion of ingested N from feed. In practice, grazing appears to be inversely related to stocking density (Fast and Lannan 1992), which would also result in the NEBiol (Grazing) graph becoming closer to NEEcon (Feed only)(Fig. 5-7).

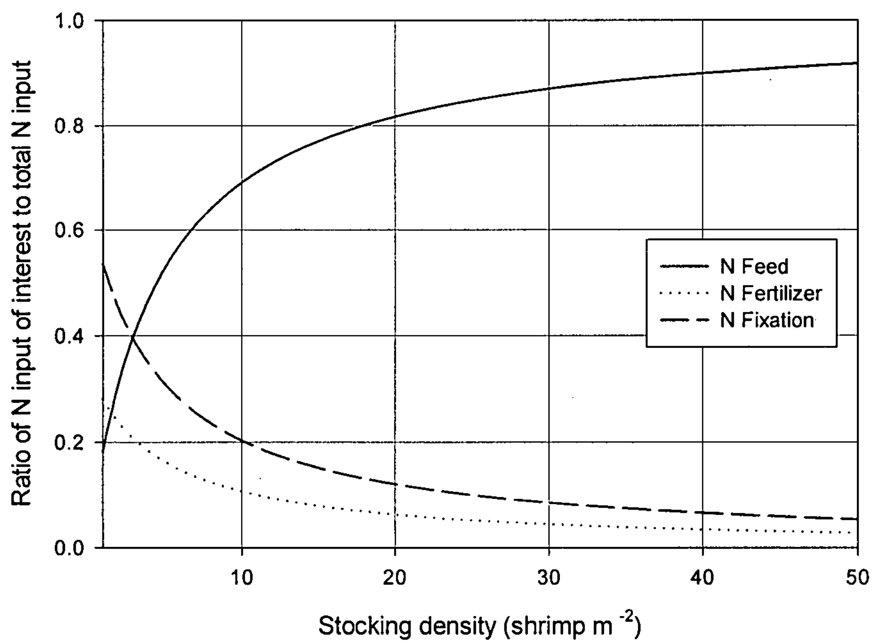

Fig. 5-6. Ratio of a particular N input, either feed, fertilizer, or fixation, to total N inputs.

The amount of N fertilizer would also be expected to depend on farming intensity (i.e., *Do*). However, there is a lack of information regarding the relationship between fertilizer use and *Do* values in shrimp farming (Boyd 1990), thus difficult to include in the model simulation. Unmanaged N inputs, such as N fixation, may also be influenced by some management practices associated with more intensive forms of farming, such as higher *W* and aeration. These relationships are also poorly known.

In the low-density range (i.e., extensive shrimp farming), N from fertilizer and fixation accounted for a considerable proportion of total N inputs. However, for *Do* values of 15 shrimp·m $^{-2}$ and higher, feed comprised more than 75% of total N inputs (Fig. 5-6).

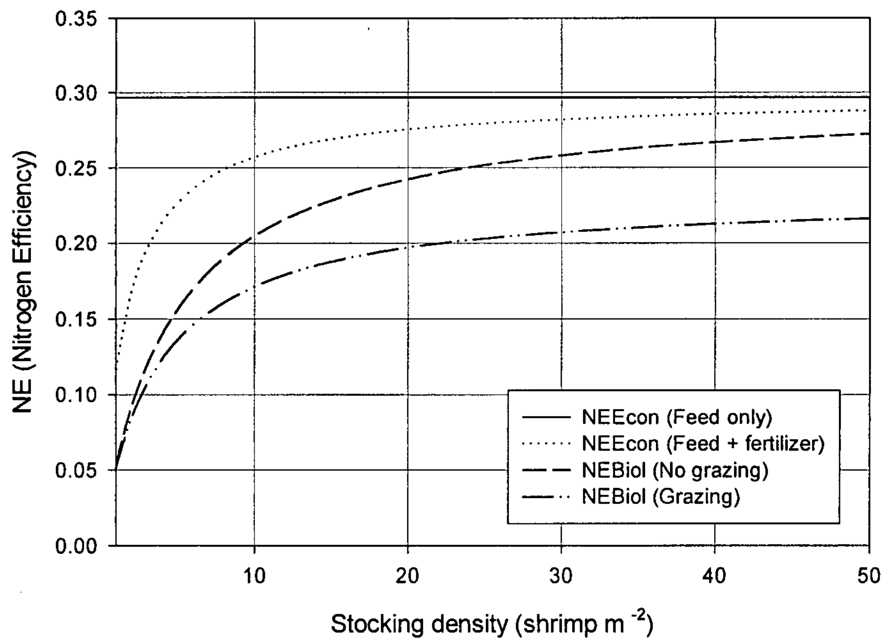

Fig. 5-7. Nitrogen Efficiency (NE) as a function of stocking density (*Do*) in shrimp farming. NEEcon refers to the economic approach to NE, or the consideration of priced N inputs, while NEBiol refers to the biological approach to NE, or the consideration of both priced and unpriced N inputs. NE = N in harvest / N inputs. Survival rate = 0.7 as in the baseline case, and no N was considered in the water inflow.

Nitrogen in the water inflow may also contribute importantly to N inputs in the low *Do* range. The water inflow represented 62% of N inputs in the farming of *L. stylirostris* at a Do = 1 shrimp·m $^{-2}$. However, at a Do = 30 shrimp·m $^{-2}$, feed became the dominant input, and N in the water inflow represented only 10.75% of total N inputs (Martin et al. 1998).

An exception was a considerable amount of N in the water inflow, when high *W* values were simulated. Fig. 5-8 shows a simulation model scenario of considering 0.25 mg·L $^{-1}$ of TAN in the water inflow, a TAN level reported for some shrimp farms in northwest Mexico (Paez-Osuna et al. 1997).

Therefore values of NEBiol may become strongly dependent on both high values of *Do* (and its associated high feed input) and *W* (when either N level is high or *W* is high). The presence of N (either in organic or inorganic forms) is a common feature of shrimp farms that rely on estuarine water, such as many in Northwest Mexico (Guerrero-Galvan et al. 1999). Nitrogen in the water inflow represented 76% of total N input using a *W* = 0.25·d $^{-1}$ in the farming of *L. setiferus,* even at a high *Do* (44 shrimp·m $^{-2}$) (Hopkins et al. 1993). While NEEcon would be equal to 0.16 for that example, NEBiol would represent only 0.04.

Nevertheless, the fate of N inputs other than feed, and their contribution towards shrimp production (i.e., shrimp growth) is poorly known. Therefore, calculating NEBiol may be considered a misrepresentation of the real N flux into shrimp production (i.e., biomass). On the other hand, NE values in aquaculture are reported often as a ratio of N in harvest to N in feed (i.e., NEEcon (Feed only)). This suggests that the efficiency of N feed may be often overestimated due to the lack of knowledge, and consideration, of the contribution of other N inputs (Goddard 1996).

For example, a potential source of error in the calculation of NEEcon (Feed only) may be the contribution of N from natural foods that may compensate for the uneaten feed and thus may lead to underestimate the uneaten feed proportion. Reducing the feeding rate per day from 3% to 1.5% did not affect *L. vannamei* growth, resulting in an almost a two-fold value of the Food Conversion Ratio[8] (FCR) (Martinez-Cordova, Pasten-Miranda, and Barraza-Guardado 1998). NEEcon (Feed only) values as high as 0.67 may be inferred from some studies in outdoor shrimp ponds (Green et al. 1998), but it is highly probable that grazing contributed considerably to shrimp growth in this study as *Do* was low (7 shrimp·m $^{-2}$). Had grazing contributed 50% of the N in shrimp biomass, the real NEEcon (Feed only) value would be 0.33, closer to values reported elsewhere.

The variability in values highlighted the need to specify N inputs in order to use NE as a benchmark. The consideration of all N inputs (i.e., NEBiol) conveys a more comprehensive approach and may serve as a benchmark against which to compare the subsequent performance of a particular shrimp farming enterprise (i.e., system). Still, calculating the amount of N from unmanaged inputs (e.g., fixation, water inflow or rain) is troublesome because of their measurement or because of their temporal and spatial variability. Estimating the amount of N from feeds and inorganic fertilizer is

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[8] Food Conversion Ratio (FCR) is the ratio of total feed fed (dry weight) to total increase in biomass (wet weight).

comparatively easy. Nevertheless, the management of an individual N component influences very often other components, as has been observed in agriculture (Dou et al. 1998), and therefore management practices that focus on a single N component may be regarded as inappropriate. Ideally, management practices should consider all N system components and flows.

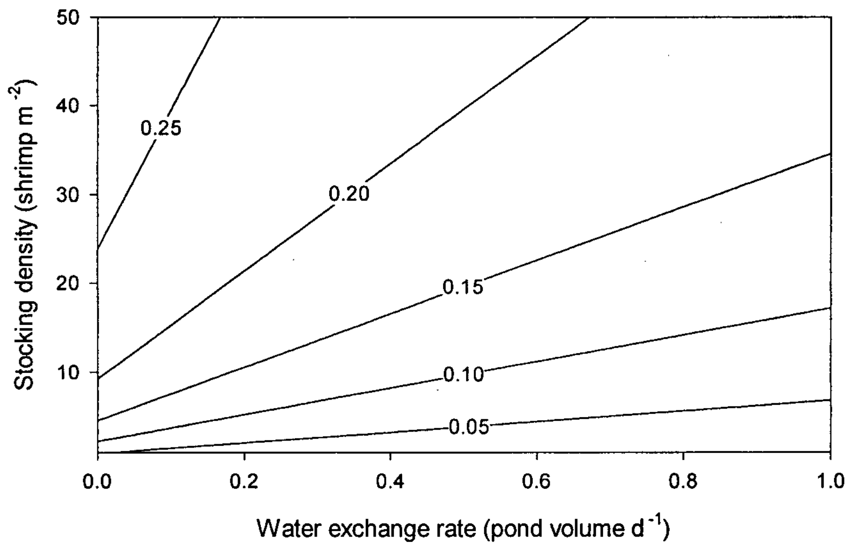

Fig. 5-8. NEBiol values as a function of shrimp stocking density (*Do*) and water exchange rate (*W*), assuming 0.25 mg·L^{-1} of TAN in the water inflow.

5.1.2.2 Survival.

Values of both NEEcon and NEBiol were clearly dependent on shrimp survival rates (Fig. 5-9) which affected the numerator of both NEEcon and NEBiol indicators. The higher the survival, the higher the NE. A clear trend was observed in the farming of *L. stylirostris.* NEEcon (Feed only) values of 0.33, 0.28, 0.26, 0.29, 0.19, and 0.17

corresponded to *Do* values of 1, 4, 7, 15, 22 and 30 shrimp·m $^{-2}$, with survival rates of 0.92, 0.94, 0.83, 0.79, 0.42 and 0.38 respectively (Martin et al. 1998).

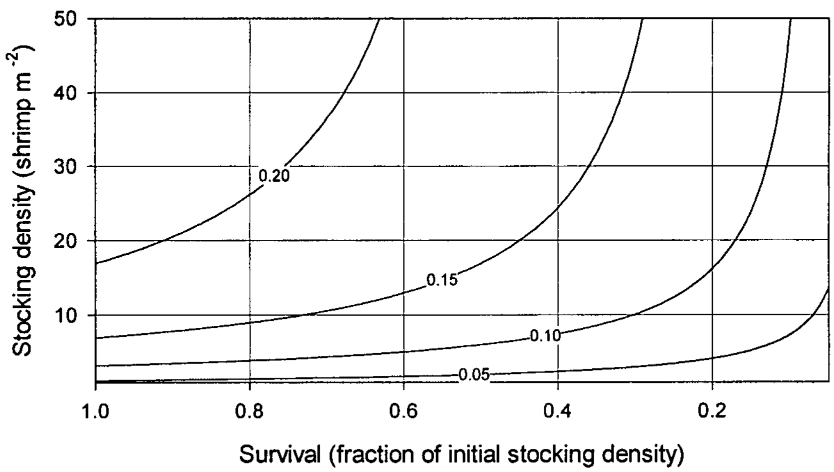

Fig. 5-9. NEBiol as a function of stocking density (*Do*) and survival. Water exchange rate (*W*) = 0.04·d $^{-1}$

5.1.2.3 Protein level in feed.

An increment in the protein level resulted in a proportional increment in growth, as my model was not able to account for differential consumption rates as a function of protein level (Fig. 5-10). Thus, the final shrimp biomass (at harvest) was very sensitive to the level of protein in the feed (Fig. 5-11).

The improvement of feeds may contribute substantially to increase NE in shrimp farming. Although a level of 30% protein is considered as appropriate, *L. vannamei* fed diets of 30% and 20% protein, with same protein: energy ratio performed similarly in growth (Hopkins, Sandifer, and Browdy 1995). For example, assuming that 1 kg of shrimp was produced at a FCR of 2, the use of 20% protein feed would result in NE equal to 0.5 while the use of 30% protein feeds would result in a NE of just 0.33. At current costs per tonne in Mexican Pesos for January, 2000, of Mex$ 4,010 and Mex$ 5,290 for the 20% and 30% protein feeds respectively (Purina Brand), changing from a 30% to a 20% protein feed would represent a 22.5% reduction in feed cost (Table 5-2). Given that feed represents the largest N input and the largest variable cost in semi-intensive and intensive shrimp farming (Hargreaves 1998), feeds and feeding-related issues are crucial in the improvement of biophysical and economic efficiency in aquaculture.

5.1.2.4 Uneaten feed.

My model considered feeding as a proportion of surviving shrimp biomass, and therefore the amount of feed fed was calculated accordingly. In practice, the amount of feed fed is calculated over an estimated surviving biomass, which often leads to overfeeding, resulting in a higher proportion of uneaten feed and consequently in a lower NE.

Common uneaten feed estimates represent 10-20% of total feed in semi-intensive to intensive shrimp farming. High levels of lost feed (i.e., formulated feeds) originate in part from the external mastication of shrimp, which results in considerable leaching of nutrients (Goddard 1996). Therefore, pellets of appropriate characteristics (i.e., more stable) may improve substantially the NE indicator.

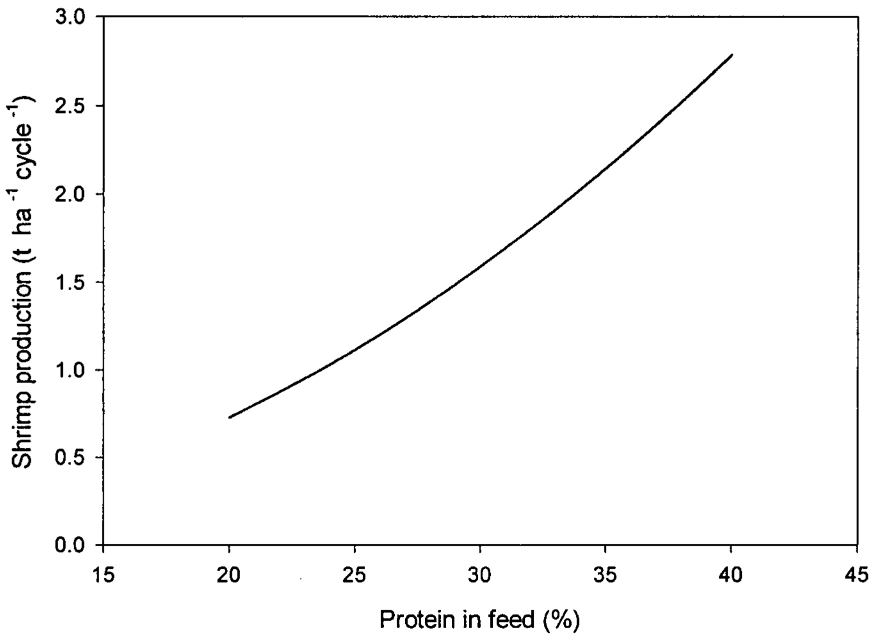

Fig. 5-10. Shrimp production, in tonnes per cycle, as a function of protein level in the feed.

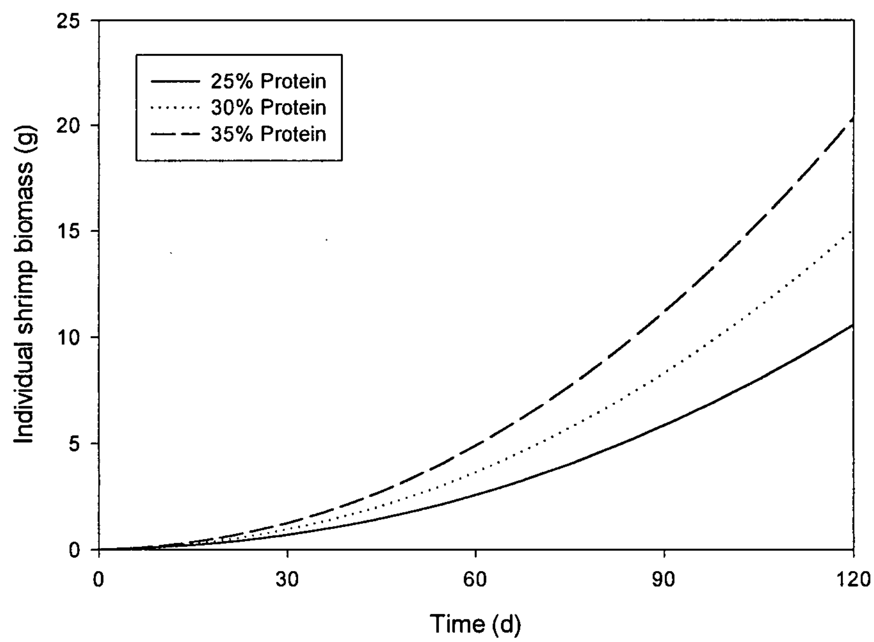

Fig. 5-11. Sensitivity of shrimp biomass to protein level in the feed.

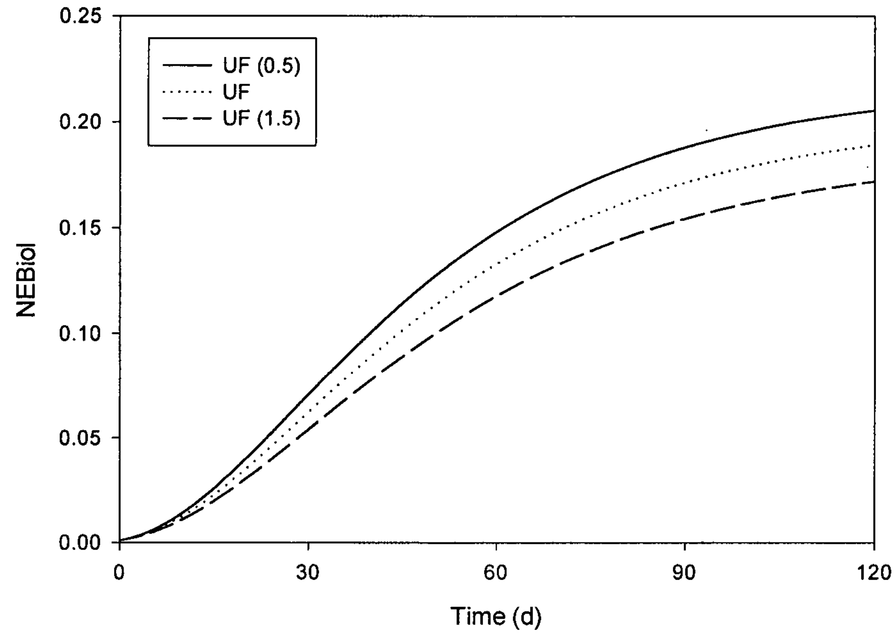

Fig. 5-12. Sensitivity of NE to the level of uneaten feed, at 50% variation.

A sensitivity analysis indicated that NE was not highly sensitive to N lost through uneaten feed (Fig. 5-12). However, uneaten feed levels may be as high as 50% in the more intensive side of shrimp farming (Wyban, Sweeney, and Kanna 1988). Still, N from uneaten feed may not be considered as a total loss, as the nutrient may contribute to fertilizing the pond.

5.1.2.5 Grazing.

The contribution of grazing to shrimp growth, and its influence on NE was not highly sensitive on my model (Fig. 5-13). My model assumed that grazing was a constant proportion of ingested N from formulated feeds and therefore changes in the inputs were reflected proportionately on the outputs.

The literature indicates that the contribution from grazing appears inversely related to *Do* (Fast and Lannan 1992) and therefore more intensive practices may rely less on grazing. Few studies have addressed the contribution of natural food in shrimp farming as the issue is inherently difficult due to the variability in the pond environment. It is well known that the consumption of natural foods contributes to shrimp growth under farming conditions (Goddard 1996). Still, the extent of this contribution is poorly known. A study on the growth of *L. vannamei* indicated that the consumption of natural foods may contribute up to 60% of shrimp growth (Anderson, Parker, and Lawrence 1987). The consumption of natural foods may represent an important contribution to shrimp growth mainly in the extensive to semi-intensive range of shrimp farming practices (Reymond and Lagardere 1990).

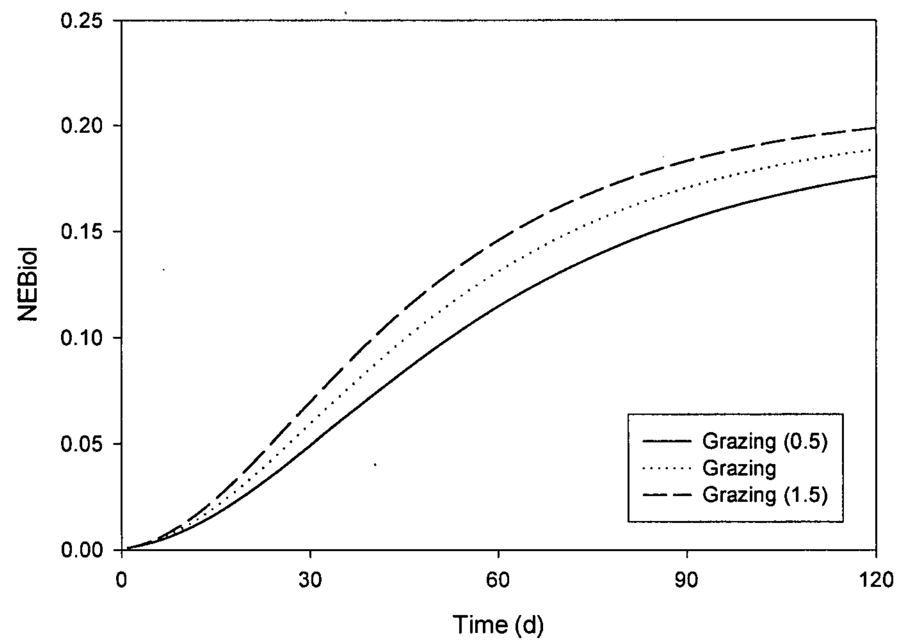

Fig. 5-13. Sensitivity of NE to the level of nitrogen from grazing, at 50% variation. The baseline case assumed a 25% contribution from grazing to the total N ingested by shrimp.

Management may influence the contribution of grazing by the enhancement of primary productivity through fertilization (Martinez-Cordova, Pasten-Miranda, and Barraza-Guardado 1998). Although uneaten feed may contribute to fertilizing and this process is taken for granted by many shrimp farmers, the practice of fertilizing through feed may represent a considerable economic cost (Funge-Smith and Briggs 1998).

5.1.2.6 Nitrogen retention

The model assumed that a constant proportion (30%) of N ingested was retained (i.e., converted into biomass). Therefore, an increased protein level in the model simulations resulted in an increased amount of N retained per unit of consumption, increased size and bigger final biomass (Fig. 5-14). Thus, the proportion of N retention affected considerably NE, as reflected by the sensitivity analysis shown in Fig. 5-15.

Feed consumption rates as a function of shrimp size, or biomass, are poorly understood (Goddard 1996). Knowledge of that relationship would allow it to be incorporated into my simulation model as to determine, for example, a daily limit of N incorporation, which the model did not consider.

There is a direct relationship between FCR and N retention and an inverse relationship between FCR and waste production in shrimp farming, as theoretically shown for *L. vannamei* in Fig. 5-16, and practically observed in the farming of *L. stylirostris* (Martin et al. 1998).

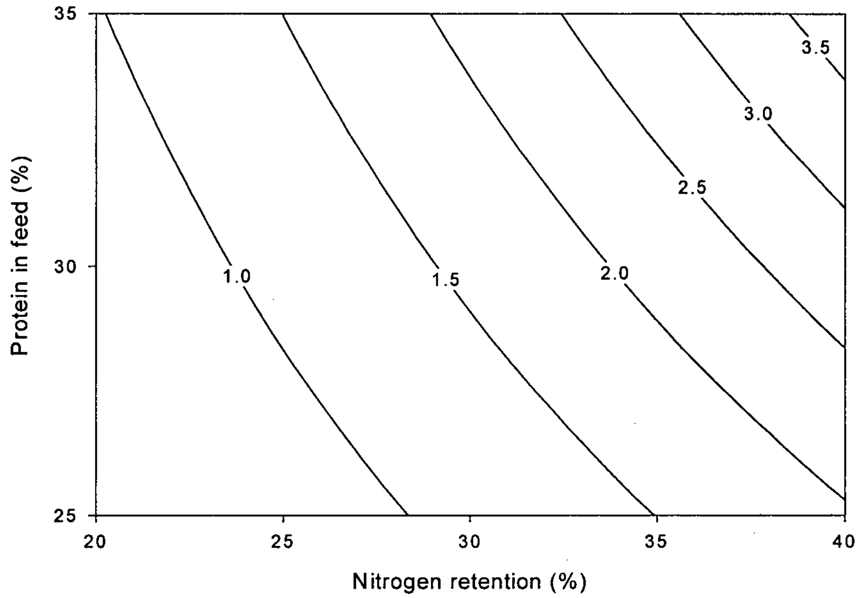

Fig. 5-14. Shrimp production, in tonnes per ha, as a function of nitrogen retention in biomass, and level of protein in feed. 1 ton of shrimp is equivalent to 28.56 kg of N, according to *L. vannamei* composition (Boyd and Teichert-Coddington 1995)

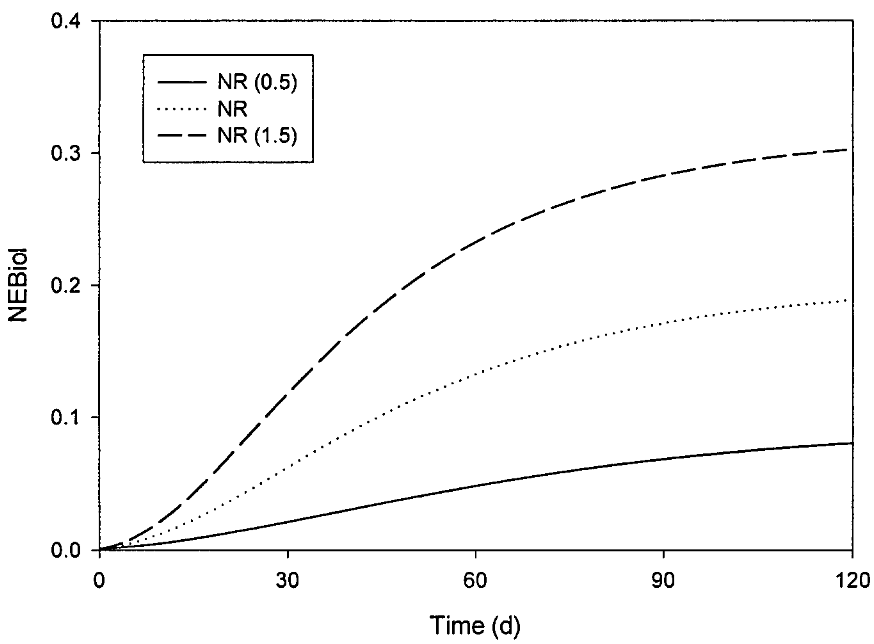

Fig. 5-15. Sensitivity of NE to the level of nitrogen retention in biomass, at 50% variation. The baseline case assumed that 30% of N ingested was retained as biomass.

Theoretical loss of nitrogen in shrimp farming.

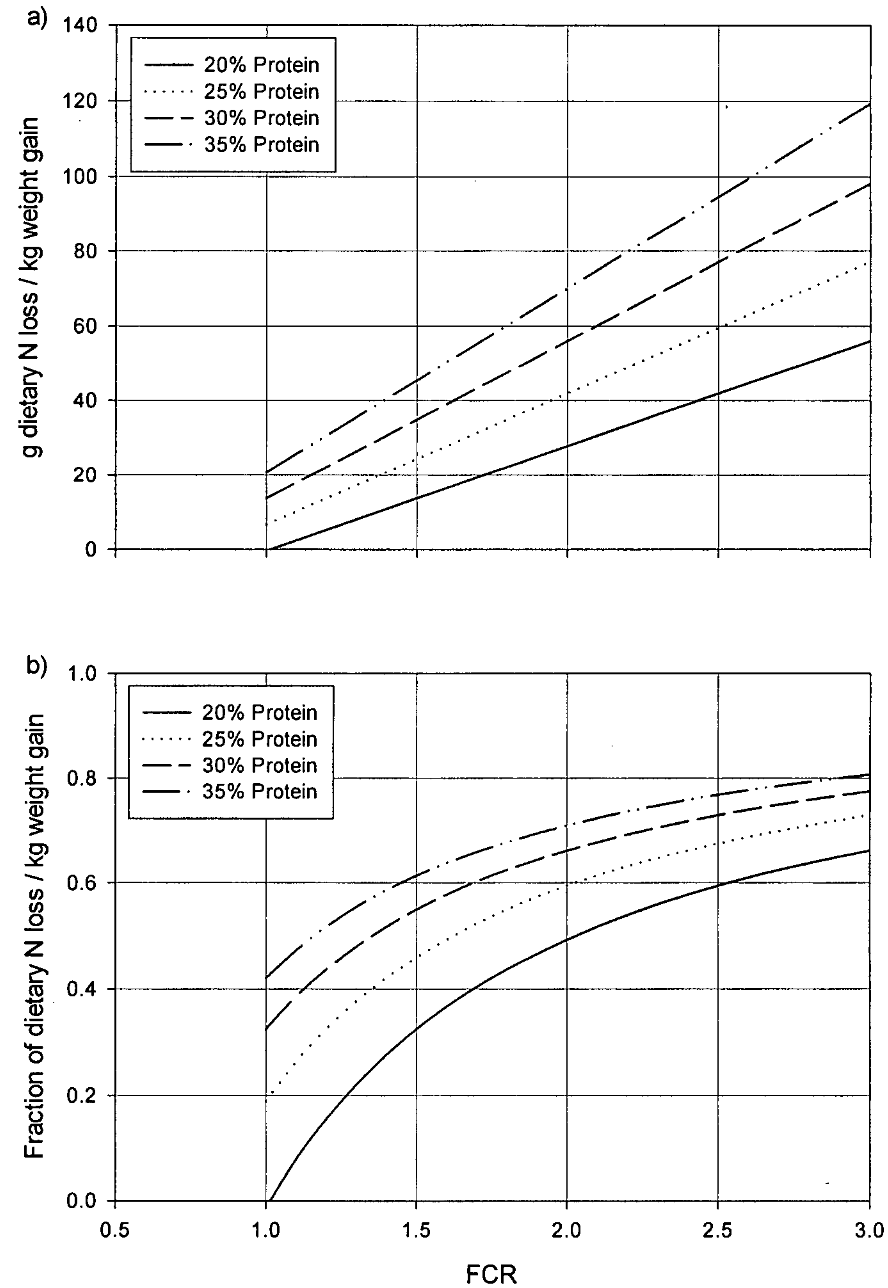

Fig. 5-16. Theoretical relationship in *Litopenaeus vannamei* between food conversion ratio (FCR) and *a*) loss of dietary N (g) per kg weight gain; *b*) the fraction loss of dietary N per kg weight gain. Data are for feed containing 20, 25, 30 and 35 % crude protein expressed as air-dry weight and 88% oven-dry weight, common levels in shrimp farming. One kg of weight gain is assumed to be equivalent to 178.5 g protein = 28.56 g N. (Shrimp composition from Boyd and Teichert-Coddington 1995; Feed composition from Purina 1998).

Table 5-2. NE, dietary N lost and the cost of feed at different feed protein levels, assuming the production of 1 t of shrimp at a FCR = 2.

Protein level (%)	Total N in feed (kg)	Total N retained in biomass (kg)	NE (Feed only)	Dietary N lost per t of shrimp produced		Cost of feed per t of shrimp produced (in Mexican Pesos)
				kg	%	
20	56.32	28.56	0.5	27.76	49.29	$8,200.00
25	70.40	28.56	0.4	41.84	59.43	$9,300.00
30	84.48	28.56	0.34	55.92	66.19	$10,580.00
35	98.56	28.56	0.29	70.00	71.02	$11,960.00

One kg of shrimp weight gain is assumed to be equivalent to 178.5 g protein = 28.56 g N. (Shrimp composition from Boyd and Teichert-Coddington 1995; Feed composition from Purina 1998). Feed cost in Mexican Pesos for January 2000.
FCR = Amount of feed fed (dry weight) / increase in biomass (wet weight)

Table 5-3. NE, dietary N lost and the cost of feed at different FCR, assuming the production of 1 t of shrimp and the use of a 30% protein feed.

FCR	Total N in feed (kg)	Total N retained in biomass (kg)	NE (Feed only)	Dietary N lost per t of shrimp produced		Cost of feed per t of shrimp produced (in Mexican Pesos)
				kg	%	
1.0	42.24	28.56	0.5	14.48	34.2	$5,290.00
1.5	63.36	28.56	0.45	34.8	54.92	$7,935.00
2.0	84.48	28.56	0.34	55.92	66.19	$10,580.00
2.5	105.6	28.56	0.27	77.04	72.95	$13,225.00
3.0	126.72	28.56	0.23	98.16	77.46	$15,870.00

One kg of shrimp weight gain is assumed to be equivalent to 178.5 g protein = 28.56 g N. (Shrimp composition from Boyd and Teichert-Coddington 1995; Feed composition from Purina 1998). Feed cost in Mexican Pesos for January 2000.
FCR = Amount of feed fed (dry weight) / increase in biomass (wet weight)

Assuming that the same growth was achieved with feeds containing 20 and 30% protein, the 22% reduction in cost by adopting the 20% protein feed (already discussed above) is accompanied by a 47% increase in NE and a 50% reduction in dietary N that is lost to the environment. Similarly, the possibility of passing from a FCR of 2.5 to a FCR of 1.5 implies a 40% reduction in cost, a 66% increase in NE, and a 54% reduction in dietary N being lost to the environment (Table 5-2 and Table 5-3). Minimizing FCR relates to nutrition, feeding practices and other parameters such as *Do*, and therefore, minimizing FCR is an important issue regarding the improvement in NE of shrimp farming, and the reduction of nitrogenous waste.

Nitrogen retention is controlled mainly by physiological limits and it would be reasonable to speculate that molecular biology and genetic engineering may overcome some of these limits in the future (Ehui and Hertel 1992).

In conclusion, the model predictions suggested that by changing management practices, such as *Do*, feeds and feeding, and those that potentially increase the survival of shrimp, the value of NE may be increased. However, as reviewed in Chapter 2, the assessment of NE requires a clear definition of boundaries in order to use NE as a benchmark. For example, calculating NE as the proportion of edible parts of shrimp (i.e., tails) would result in different values as tails represent 65% of *L. vannamei* bodymass (FIRA 1998). Assuming a constant proportion of N in the whole body, NEEcon in the baseline case would be reduced proportionately, from 26.92% to 17.55% if tails were used. While NEEcon for round salmon in Norway represented 25.00%, NEEcon calculated through edible parts represented only 15.62% (Bleken and Bakken 1997).

5.1.3 Ammonia Assimilative Capacity (AAC).

The model predicted AAC values across the whole range of farming intensity, as shown in Fig. 5-17. In order to obtain values of the indicator one order of magnitude lower, I divided AAC by 10. The maximal AAC value within the whole range was equal to 8.26, corresponded to $W = 1.0 \cdot d^{-1}$, and was independent of *Do* (Fig. 5-17).

For *W* values below $0.2 \cdot d^{-1}$, the maximal AAC value was 6.58 and corresponded to Do = 37.25 $shrimp \cdot m^{-2}$, and to W = 0 (Fig. 5-18), the same values obtained for the highest NP in this range. An AAC value of 6.58 was also obtained with $W = 0.74 \cdot d^{-1}$, and higher AAC values corresponded to higher *W* values (Fig. 5-17).

TAN inputs in the simulation model originated from the shrimp subsystem (i.e., excretion and mineralization of uneaten feed and faeces) as well as recycling (i.e., mineralization of phytoplankton) from the water subsystem. However, the water subsystem model was based on net rates of ammonia (i.e., ammonia was produced and assimilated again in the phytoplankton compartment) which was removed through sedimentation, volatilization and water exchange. The consideration of net rates in the water subsystem implied that the gross rate of phytoplankton growth and ammonia recycling were much higher. However, the important issue in the management of water quality is the concentration of ammonia originated through the net rates of N flows. Unless that process can be manipulated to modify net rates, the gross rates and the rapid cycling of N within the plankton is not directly relevant to management.

The model predicted that in the *W* range between 0 and $0.2 \cdot d^{-1}$, TAN levels increase with increasing *W* values (Fig. 5-19 and Fig. 5-20). A possible explanation is that phytoplankton was removed through water exchange faster than its capacity to assimilate ammonia, which accumulated in the water column, as observed in other shrimp farming studies (Hopkins et al. 1996; Lorenzen, Struve, and Cowan 1997). This may have important implications as water exchange is considered by farmers as an effective strategy to improve water quality in shrimp farming ponds, and its rate is increased when problems arise. Also, this *W* range predicted by the model corresponded with the common range to which *W* is increased (0.05 to $0.1 \cdot d^{-1}$) in semi-intensive to intensive shrimp farming in Northwest Mexico.

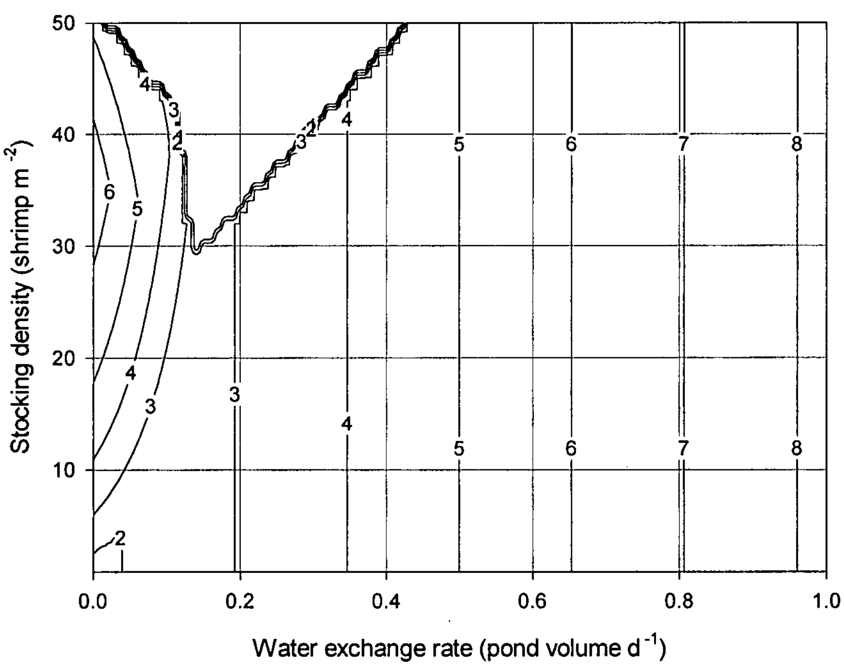

Fig. 5-17. Ammonia Assimilative Capacity (AAC) for the whole range of farming intensity. AAC is the ratio of total TAN inputs to the maximum TAN level (below 1 $mg \cdot L^{-1}$) reached during the farming period, and divided by 10. The area above the thick line on the upper-left corner corresponds to TAN levels above 1 $mg \cdot L^{-1}$.

Maximum levels of TAN normally occurred at the end of the simulated farming cycle (i.e., 120th day), where total biomass was highest, though at *Do* between 3 and 43 shrimp·m $^{-2}$, and W between 0 and 0.14·d $^{-1}$, the model predicted that maximal TAN levels may occur earlier (Fig. 5-22). However, those maximal TAN values were below 1 mg·L $^{-1}$, except for a small region around Do = 40 shrimp·m $^{-2}$ and W = 0.12·d $^{-1}$. Peaks of TAN levels in these range coincided with rapid growth of phytoplankton and the assimilation rate of TAN appeared to exceed TAN production causing a drop in TAN concentration (Fig. 5-23).

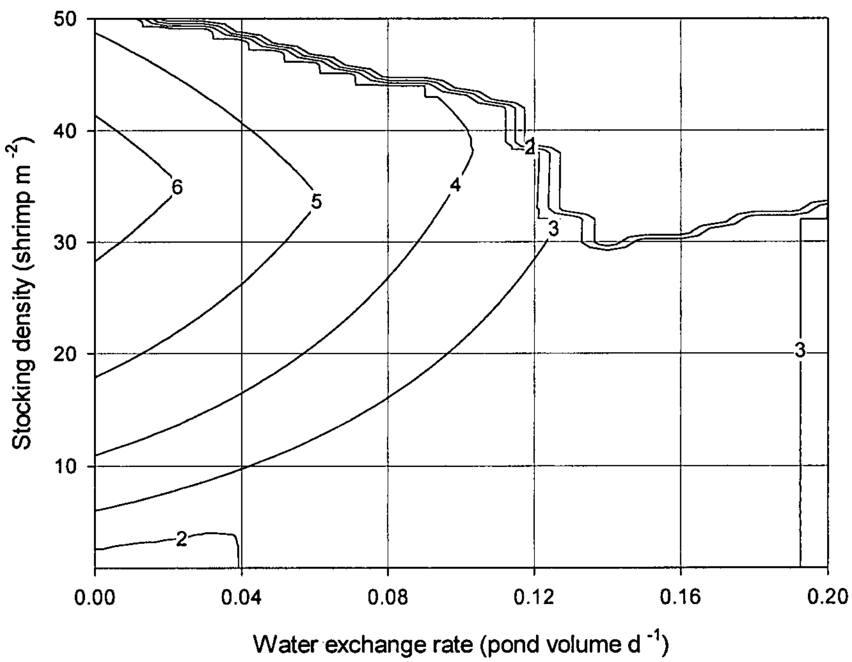

Fig. 5-18. AAC for water exchange rates (W) between 0 and 0.2·d $^{-1}$.

The prediction of ammonia behaviour in the water column for the Mexican case showed an steady increase in ammonia levels to a maximum close to 0.35 mg·L $^{-1}$ of TAN around day 60th (half of farming period), a subsequent drop in ammonia levels down to 0.2 mg·L $^{-1}$ of TAN, and a further increase reaching a level close to 0.25 mg·L $^{-1}$ by day

120th (end of farming period) (Fig. 5-23).

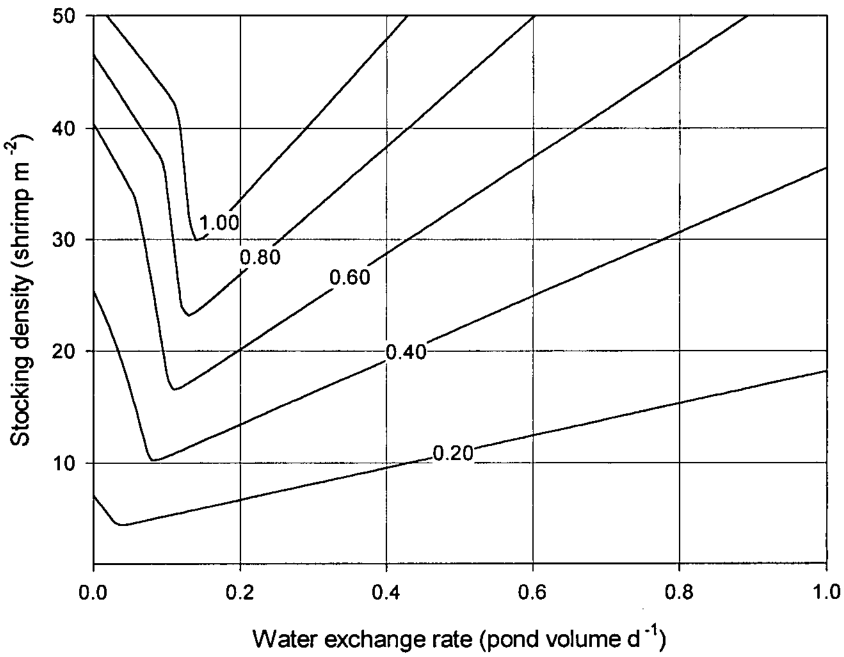

Fig. 5-19. Maximal ammonia level (TAN, in mg·L $^{-1}$) in the pond as a function of stocking density (*Do*) and water exchange rate (*W*).

One study that assessed water quality in semi-intensive shrimp farms in Northwest Mexico reported a similar behaviour of ammonia in both dry and rainy seasons, during a similar production period to that considered in my model, (Guerrero-Galvan et al. 1999). In that study, a maximum ammonia level (approx. 1 mg·L $^{-1}$ of TAN) occurred between day 60th and 90th in the rainy season, while a maximum level (approx. 0.42 mg·L $^{-1}$ of TAN) occurred between the 50th and the 70th day during the dry season. However, in that study, high values of ammonia in the pond during the rainy season corresponded to high values of ammonia in the water inflow.

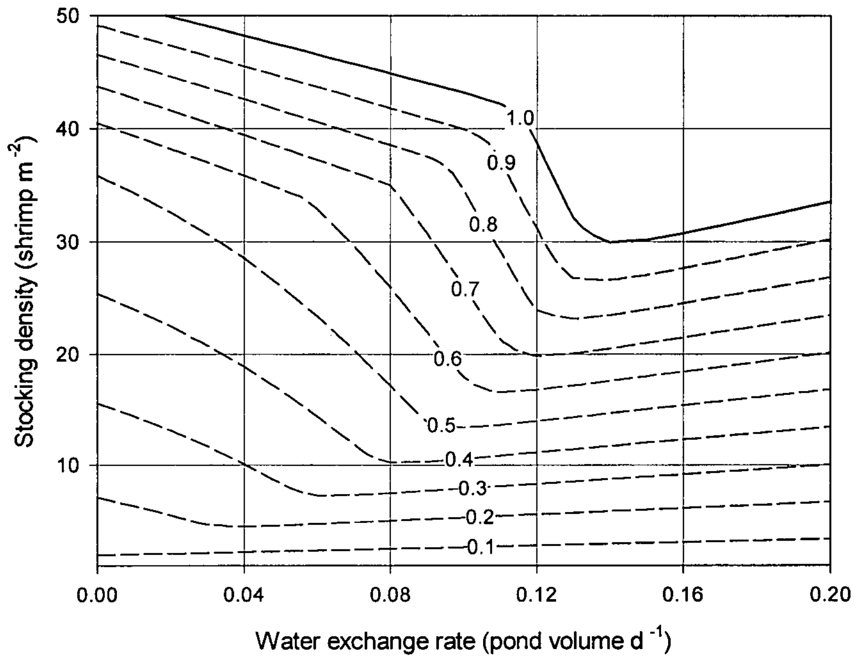

Fig. 5-20. Maximal ammonia level (as TAN, in mg·L^{-1}) as a function of shrimp stocking density (*Do*), and water exchange rate (*W*) at levels below 0.2·d^{-1}.

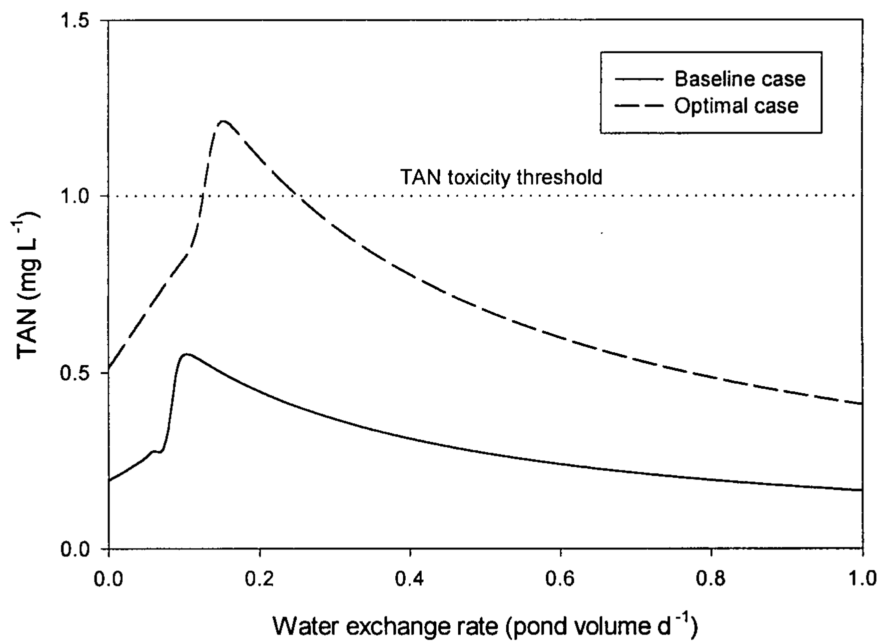

Fig. 5-21. TAN level (mg·L^{-1}) in the pond at the end of the farming cycle, for the Mexican baseline case (Do =15 shrimp·m^{-2}) and the optimal case (Do = 37.25 shrimp·m^{-2}), as a function of water exchange rate (*W*).

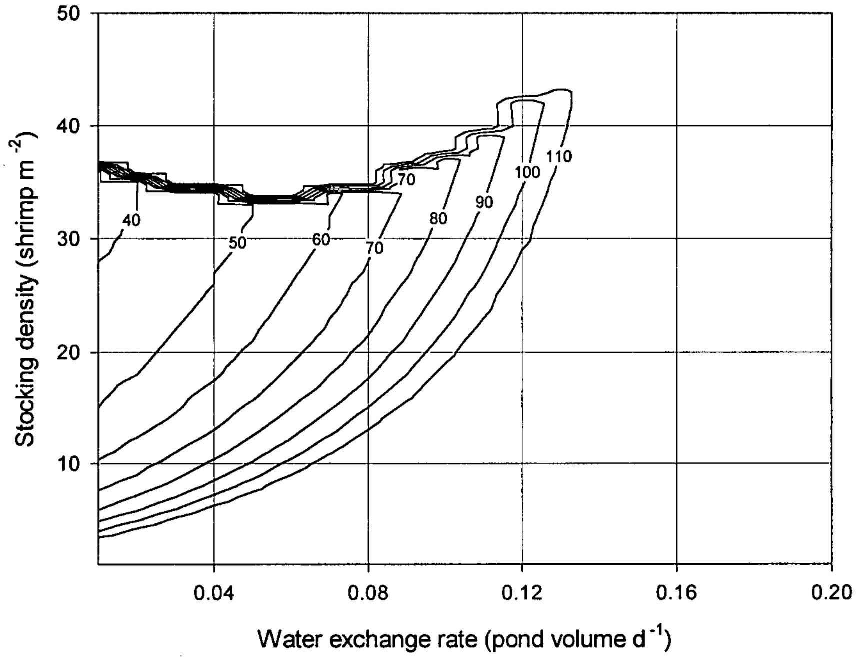

Fig. 5-22. Day of the farming period at which maximal TAN occurred. Simulated farming period = 120 days.

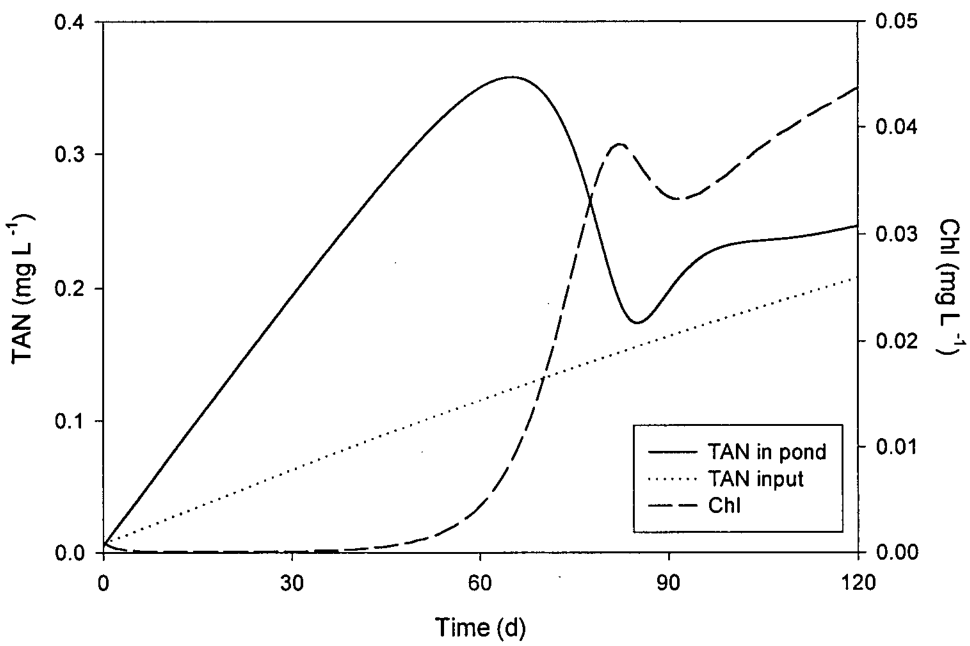

Fig. 5-23. Ammonia input from excretion and mineralization, ammonia level in the pond and phytoplankton growth (as chlorophyll *a*) for the baseline case (*Do* = 15 shrimp·m $^{-2}$; *W* = 0.04·d $^{-1}$; survival rate = 0.7), as predicted by the model.

Unionized ammonia levels predicted for the baseline case (max TAN = 0.36 mg·L $^{-1}$) were below those reported as toxic for penaeid shrimp (Sec. 3.6). The toxicity of ammonia in semi-intensive production methods may not occur commonly due to relatively low *Do* values and the associated low amounts of feed per unit of area. However, high pH levels (between 8.5 to 8.7) have been reported in shrimp farming ponds in Northwest Mexico (Guerrero-Galvan et al. 1999). At these levels the proportion of unionized ammonia increases considerably and the control of pH becomes crucial to maintain unionized ammonia at low levels. Values around 3.0 mg·L $^{-1}$ of TAN have been reported in semi-closed and closed shrimp farming systems in Thailand, although at higher *Do* (Funge-Smith and Briggs 1998). These high levels of ammonia coincided with stressful conditions, slow growth and disease outbreaks, and the perception that the "carrying capacity" of shrimp farming ponds had been exceeded. Those conditions appeared between the 100th and 120th day of farming (Funge-Smith and Briggs 1998).

Toxic levels of ammonia in shrimp should be approached with caution. Toxicity of ammonia to organisms in brackish and marine waters is poorly understood (Handy and Poxton 1993). Besides, chronic effects of ammonia levels are poorly understood in penaeid shrimp. Toxicity values have often been extrapolated from short-term, acute tests and from values obtained with fish studies (Lee and Wickins 1992)

5.1.3.1 Phytoplankton growth.

The assimilation of ammonia through phytoplankton was an important path for the removal of that metabolite from the water column. The concentration of ammonia in the pond was highly sensitive to g_{max}, the maximum growth rate of phytoplankton (Fig. 5-24). Therefore, changes in the phytoplankton population growth were reflected in ammonia levels in the pond. This is particularly important in practice when phytoplankton dies-off. Similar findings are reported elsewhere (Lorenzen, Struve, and Cowan 1997).

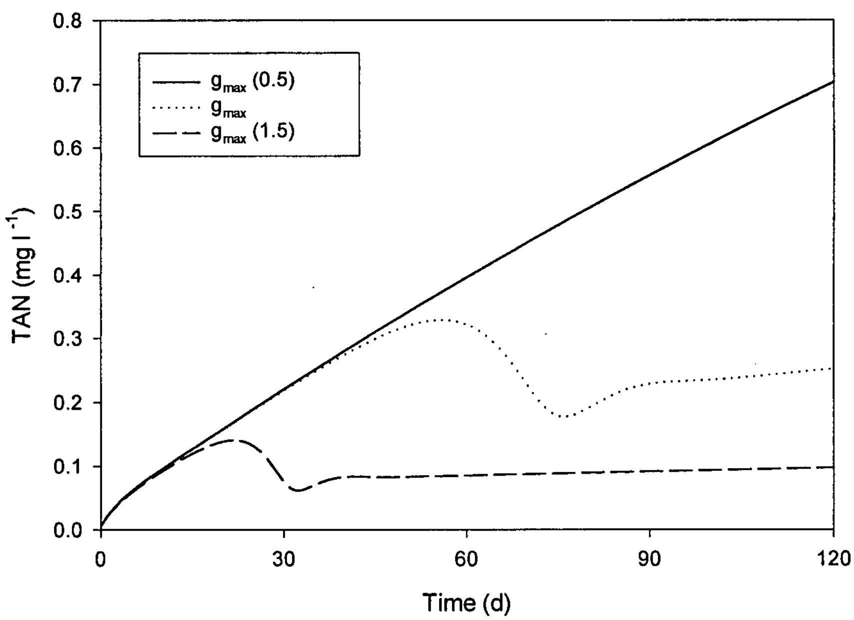

Fig. 5-24. Sensitivity of ammonia (TAN) concentration in the pond to the maximum growth rate of phytoplankton (g_{max}), at 50% variation.

While phytoplankton growth represents a key process to ammonia control in shrimp farming ponds, its management is difficult. Management options to control phytoplankton communities are mainly fertilizing-related, but compared to freshwater aquaculture, fertilization of brackish-water ponds is poorly understood (Boyd 1990). Encouraging the growth of certain species may potentially attain the stabilization of phytoplankton blooms (Burford and Glibert 1999).

5.1.3.2 *Sedimentation rate*

Ammonia levels in the pond were also highly sensitive to sedimentation rates of phytoplankton, as shown in the sensitivity analysis (Fig. 5-25). Still, the manipulation of sedimentation rates of phytoplankton is poorly known and does not represent a practical alternative at present (Lorenzen 1999).

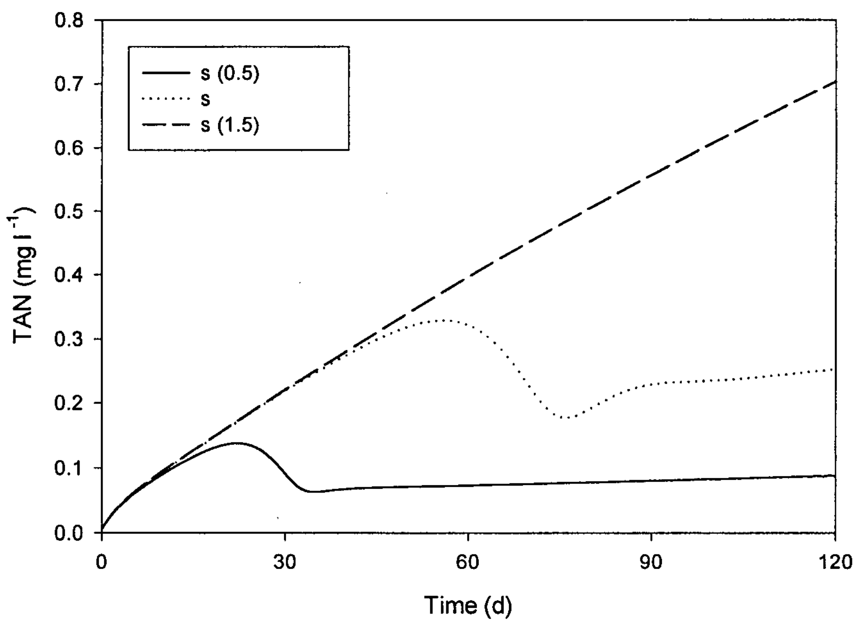

Fig. 5-25. Sensitivity of ammonia (TAN) in the pond to sedimentation rate at 50% variation.

However, other alternatives to ammonia control in the pond look promising, such as the control of inorganic N by its assimilation through bacteria fed carbohydrates, resulting in a microbial protein that may serve as feed in the pond (Avnimelech 1999).

5.1.3.3 Protein level

Among the factors that may affect ammonia excretion is the partitioning of N among metabolism, activity and growth, which may influence the partitioning of metabolic N waste. Such partitioning is dependent on factors such as energy intake, temperature, stress and age (Cui and Wootton 1988). Therefore factors that affect NE, through the partitioning of N into growth, are also expected to influence ammonia excretion such as the level of protein in feed (Fig. 5-26) and feed and feeding related factors (Fig. 5-16).

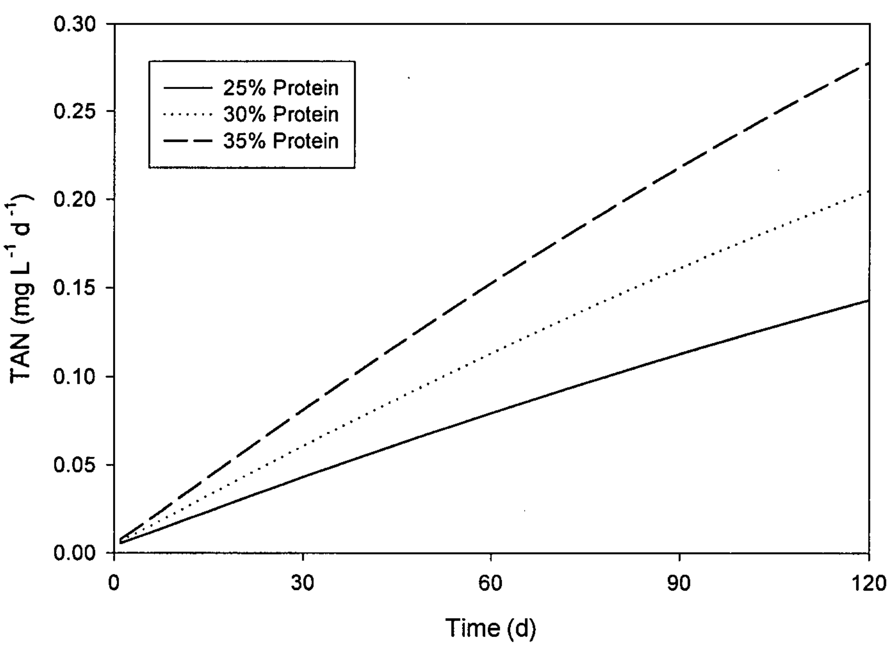

Fig. 5-26. Sensitivity of ammonia excretion to protein level

5.2 Nitrogen budgets

Nitrogen budgets for the baseline (*Do* =15 shrimp·m $^{-2}$ and *W* =0.04·d $^{-1}$) and the optimal (*Do* =37.25 shrimp·m $^{-2}$ and *W* =0) cases were calculated in kg N·ha $^{-1}$·cycle $^{-1}$, under economic and biological approaches. Nitrogen data for feed, volatilization, effluent and sedimentation originated from the simulation model. Nitrogen inputs from water inflow, fertilizer and fixation were not considered in the simulation model, and therefore they were not reflected in the model predictions. Surplus of inputs over outputs was assumed as a loss from the system.

The two approaches to N budgeting identified feed as the largest N input in semi-intensive (i.e., the baseline case) and intensive (i.e., the optimal case) shrimp farming, similarly to previous findings discussed in Sec. 5.1.2. Under an economic approach, feed N represented 90.66% in the baseline case while it represented 96.06% in the optimal case (Fig. 5-27 and Fig. 5-28). Under a biological approach, feed N comprised 76.96% in the baseline case while it represented 89.24% in the optimal case. The increasing contribution of feed N along the intensification trend has also been discussed in Sec. 5.1.2.

Feed N accounted for 78% of total N inputs, when all inputs (i.e., biological approach) were considered in intensive shrimp farming in Thailand (Funge-Smith and Briggs 1998).

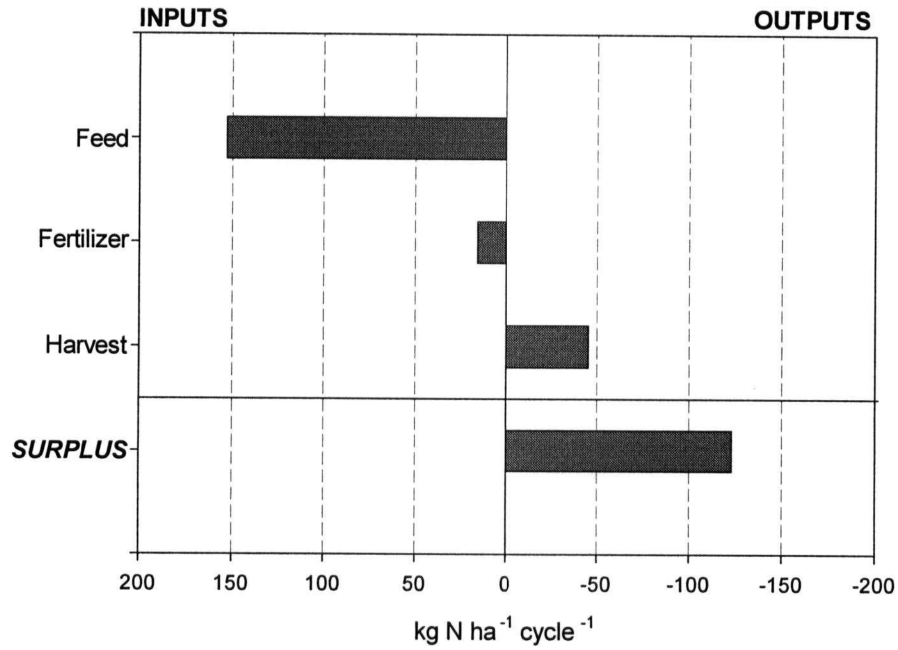

Fig. 5-27. Economic Nitrogen budget for the baseline case.

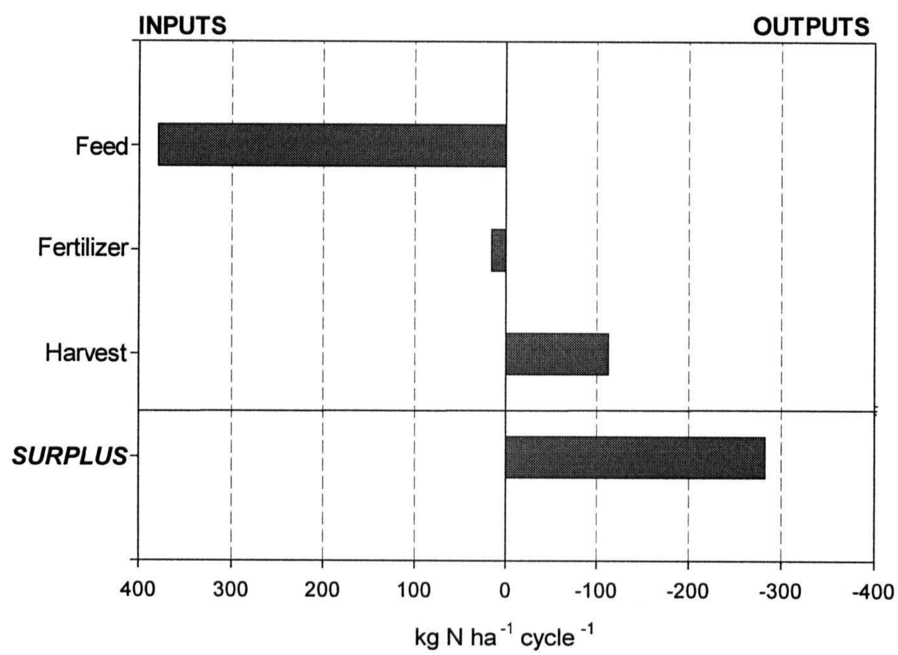

Fig. 5-28. Economic Nitrogen budget for the optimal case

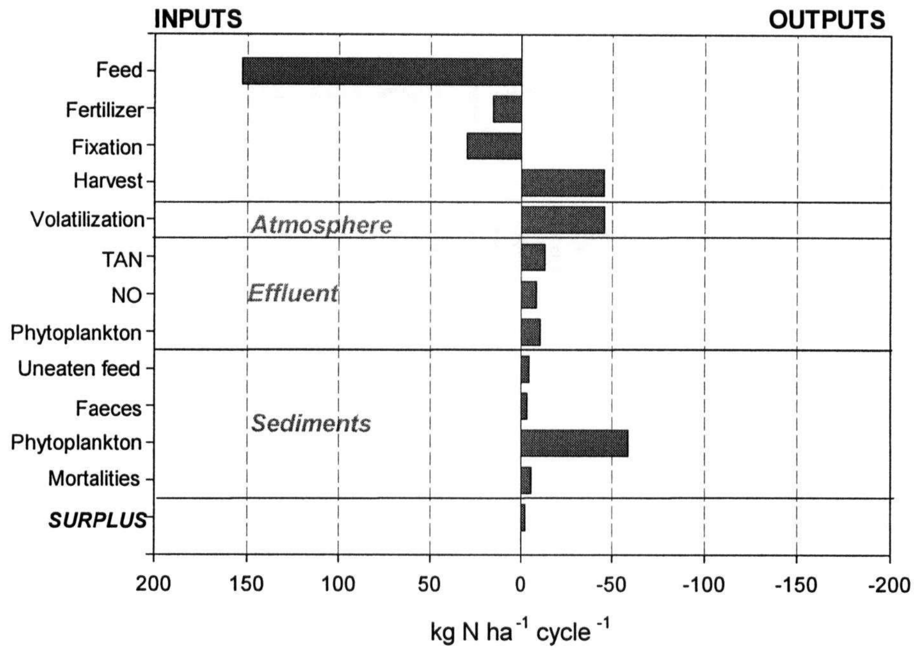

Fig. 5-29. Biological Nitrogen budget for the baseline case.

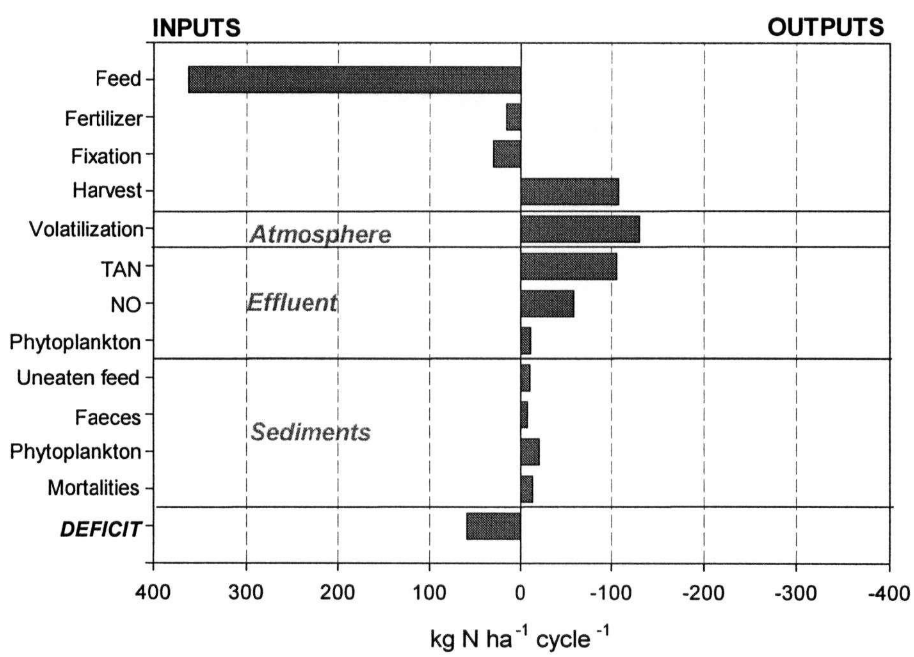

Fig. 5-30. Biological Nitrogen budget for the optimal case.

Fixation represented 15.11% of N inputs in the baseline case, but it accounted for only 7.05% in the optimal case. Therefore the model predictions suggested that N from fixation may be an important contributor to total N inputs only towards the extensive side of shrimp farming.

Fertilizer represented the smallest contributor of the N inputs examined. It accounted for 7.93% in the baseline case and only 3.7% in the optimal case. It is well known that for the most part, fertilizing of the pond in the semi-intensive and intensive shrimp farming stems from uneaten feed, faeces and excretion.

Sediments represented the major sink of N at low and no-water exchange (Fig. 5-29 and Fig. 5-30). Predicted values for the baseline case (37%) concurred with values reported for intensive shrimp farming (30%) using similar values of *W* (Briggs and Funge-Smith 1994).

The model prediction of a high amount of N remaining in the sediments at the end of the farming period highlighted the need to consider an appropriate removal and disposal of those sediments. Sediments are commonly removed after harvest, either by flushing or with machinery, as remaining sediments may cause water quality problems in subsequent production cycles (Funge-Smith and Briggs 1998). Flushing of sediments after harvest has been found to release a large amount of particulate N into receiving waters (Smith 1993).

The N budget identified also the volatilization of ammonia as an important sink of N shrimp farming ponds. Unaccounted N in published shrimp farming N budgets (Briggs and Funge-Smith 1994; Martin et al. 1998; Paez-Osuna et al. 1997) has been mostly assumed as a loss through volatilization, and such process has been assumed an important sink for N. Thus, N lost through volatilization has been calculated as the difference between total N inputs and the sum of N in harvest, effluent and sediments. Unaccounted N from feed in shrimp farms has ranged from 13 to 46% (Hopkins et al. 1993) while N budgets in aquacultural freshwater systems have also had considerable amounts of unaccounted N, ranging from 8% (Krom, Porter, and Gordin 1985) to 55% (Daniels and Boyd 1989). For example, a N budget in intensive shrimp farming estimated N volatilization to account for 30% (Funge-Smith and Briggs 1998), while a N budget in semi-intensive shrimp farming accounted volatilization in the range between 9.7 and 32.4% (Martin et al. 1998).

A biological approach appeared of higher value in identifying the relative importance of some N compartments in shrimp farming. A comparison of different approaches to N budgets in agriculture indicated that a biological budget seems appropriate to evaluate the partitioning of N into different compartments. However, the more complex the budget (i.e., including internal processes such as recycling and transfer of N) the more it was able to predict accurately N losses and the potential environmental consequences (Watson and Atkinson 1999).

5.3 Ecological implications of the nitrogen flow.

The analysis indicated that while regenerated N may be considerable in extensive forms of shrimp farming when compared to "new" N, the semi-intensive and intensive methods relied importantly on "new" N mainly in the form of feed, and potentially on water exchange.

In extensive systems the N applied in excess of that required by the shrimp may not be lost, as it remains within the system contributing to internal flows. Surplus N in the more intensive systems, for example in the form of excess feed, will be lost to the environment, thus contributing to potential eutrophication and degradation of water both within and outside the pond.

5.3.1 Shrimp ponds as a sink for ammonia

The model predicted that shrimp ponds might act as an important sink for ammonia. The amount of TAN in the water inflow influenced the maximum level of TAN in the pond during the farming cycle, depending on *W* (Fig. 5-31), and *Do* (Fig. 5-32) values. In theory (as predicted by the model) levels up to 7 $mg \cdot L^{-1}$ of TAN could be assimilated in the baseline case, without exceeding the threshold of 1 $mg \cdot L^{-1}$ in the pond (Fig. 5-32). In practice, such high levels could "break the system" through effects other than in the shrimp. For example, assimilation rates of ammonia by phytoplankton at high levels of ammonia concentration may be affected. Also, the model assumed that water entering the pond was evenly distributed, which may be difficult in practice at low *W* values. The model also assumed continuous water exchange. In practice, water exchange occurs in periods of 8-12 $h \cdot d^{-1}$. Having "static" periods in the ponds may influence N dynamics importantly.

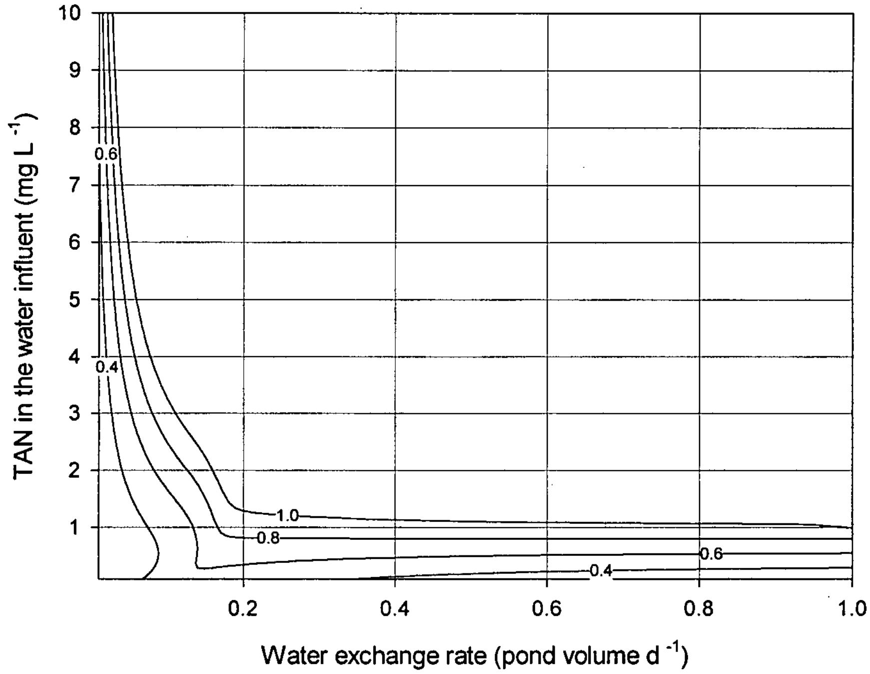

Fig. 5-31. Max TAN (below 1 mg·L $^{-1}$) in the pond as a function of TAN in the water inflow, and water exchange rate (W). Do = 15 shrimp·m $^{-2}$, as in the baseline case.

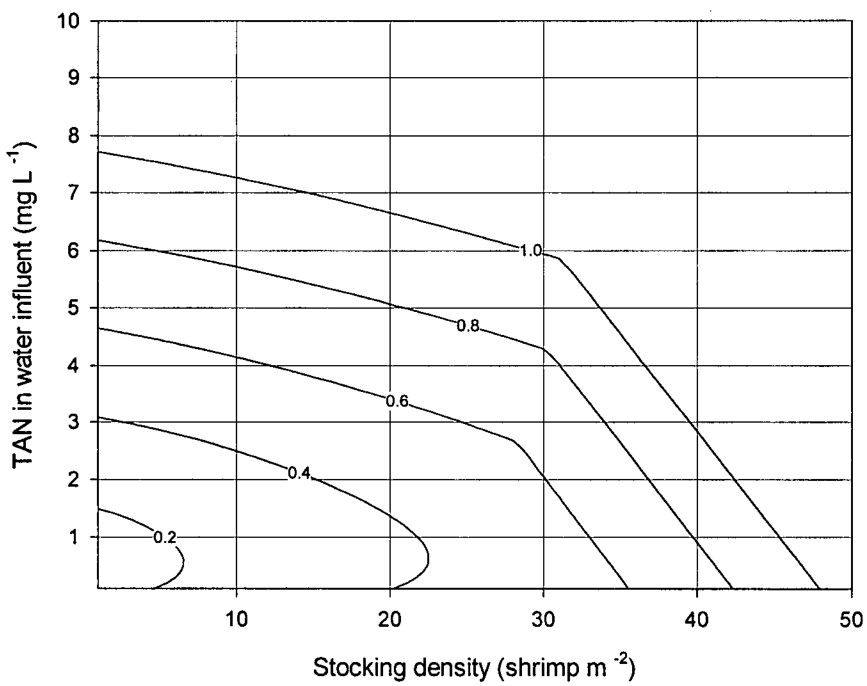

Fig. 5-32. Max TAN (below 1 mg·L $^{-1}$) in the pond as a function of TAN in the water inflow, and stocking density (Do). W = 0.04·d $^{-1}$, as in the baseline case.

Lower levels of TAN in the effluent than in the water inflow have been observed in semi-intensive shrimp farming ponds in Mexico (Paez-Osuna et al. 1997), in Ecuador (Teichert-Coddington et al. 1996) as well as in New Caledonia (Martin et al. 1998). These lower values of dissolved N corresponded to higher values of chlorophyll *a* and Total N in the effluent, which suggested that dissolved N was converted into organic N (i.e., phytoplankton). Adjacent agricultural areas to shrimp farms, such as those in Northwest Mexico, may release considerable amount of excess fertilizers, and potentially generate high levels of ammonia in waters that shrimp farms utilize.

5.3.2 Discharge of surplus nitrogen

The amount of surplus N discharged into the environment, through effluent and sediments, was estimated as the difference between N input in feed and fertilizer minus N loss to harvest and volatilization;

(5-1) $$\text{Surplus N} = \text{N (feed + fertilizer)} - \text{N (harvest + volatilization)}$$

The values of surplus N per unit of production predicted by the model in the low *W* range (0-0.2·d $^{-1}$) (Fig. 5-33) were higher than the 28.6 kg t $^{-1}$ reported for semi-intensive farming of *L. vannamei* in Northwest Mexico (Paez-Osuna et al. 1997), but smaller than those reported for extensive and semi-intensive (Martin et al. 1998) as well as for intensive (Briggs and Funge-Smith 1994) shrimp farming (Table 3-4). However, those studies had considerable amounts of unaccounted N in their budgets and estimated some losses by difference, which may represent a potential source of disparity with the predictions of my model.

The Mexican baseline case represented a N surplus discharge of 48.81 kg·t $^{-1}$, while the optimal case increased only slightly to 54.08 kg·t $^{-1}$ with an almost 2.5-fold harvest yield increment. However, increasing *W* to 0.128·d $^{-1}$, while maintaining almost the same yield increment, reduced the N surplus discharge considerably, to 36.96 kg·t $^{-1}$, due to much higher losses of N through volatilization (Table 5-5 and Table 5-6).

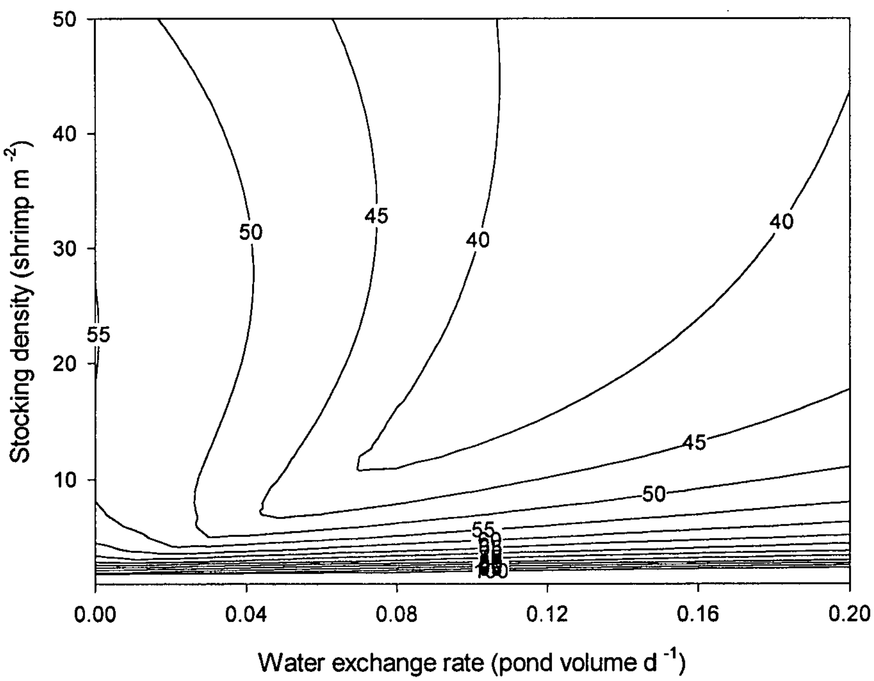

Fig. 5-33. Discharge of surplus N in kg t $^{-1}$ shrimp produced as a function of shrimp stocking density (*Do*) and water exchange rate (*W*). Surplus N = N (feed + fertilizer) – N (harvest + volatilization).

Surplus N discharged into the environment, through effluent and sediments, can be considered in terms of human-equivalent N discharge (Fig. 5-34). A value of 12 g·person $^{-1}$·d $^{-1}$ has been utilized to compare the discharge of N in aquaculture effluents with urban sewage (Bergheim, Sivertsen, and Selmer-Olsen 1982).

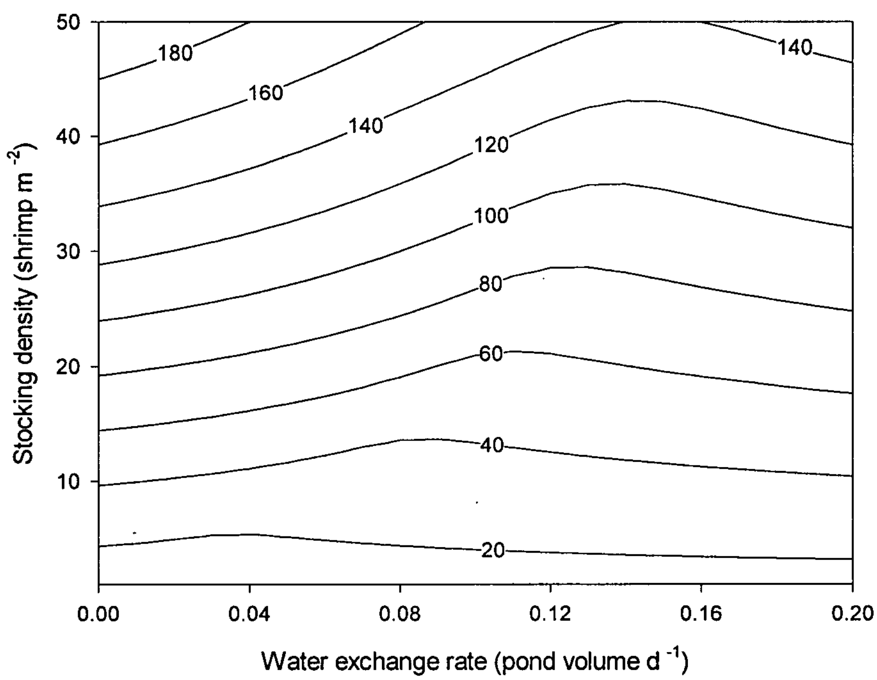

Fig. 5-34. Discharge of surplus N·ha $^{-1}$·cycle $^{-1}$ from shrimp farming, in number of persons-equivalent as a function of shrimp stocking density (*Do*) and water exchange rate (*W*). Surplus N = N (feed +fertilizer) – N (harvest + volatilization). Discharge from 1 person = 12 g·N·d $^{-1}$ (From Bergheim, Sivertsen, and Selmer-Olsen 1982)

Assuming one production cycle per year and a homogenous production among all farms, the 17, 051 ha under shrimp farming in Northwest Mexico in 1998 (Table 3-1) would have a discharge of 1, 323 t N·yr $^{-1}$, equivalent to a 300, 000 population under the baseline case management; and 3, 640 t N·yr $^{-1}$ or the equivalent of a 850,000 population under optimal operating conditions (with no water exchange). Adopting optimal operating conditions, but increasing *W* to 0.128·d $^{-1}$ would increase the N discharge to 2, 377 t N yr $^{-1}$, equivalent to a 550,000 population, yet with a 2.5-fold increase in harvest yield.

5.3.3 Reutilization of surplus nitrogen.

A high proportion of N entering the shrimp farming pond in the baseline case remained trapped in the sediments at the end of the farming period (Fig. 5-35). In practice those shrimp farming sediments contain a high salt content and little organic matter, which prevent them from being utilized as a fertilizer in agriculture.

The model predicted that moderate increases in *W* resulted in considerable higher proportions of dissolved N in the effluent (Fig. 5-36). There is the potential to recover nitrogenous wastes through biological filtration of the effluent, passing it through macroalgae for example. The potential increases by having most of the effluent N in dissolved form (Lorenzen 1999).

Using the solver function in the Excel® software, I calculated *W* values required to discharge levels of 95% and 99% of N in dissolved form in the effluent, both for the baseline and for the optimal case (Table 5-4).

For the baseline case it would be necessary to increase *W* to $0.082 \cdot d^{-1}$ as to have 95%, and to 0.095 d $^{-1}$ to achieve a level of 99%, of effluent N in dissolved form. For the optimal case it was necessary to increase *W* to $0.128 \cdot d^{-1}$ as to have 95%, and to $0.134 \cdot d^{-1}$ to achieve a level of 99% of effluent N in dissolved form. In intensive shrimp farming ponds, levels of dissolved N in the effluent higher than 80% were achieved only with W values higher than $0.6 \cdot d^{-1}$ (Lorenzen 1999). However, those ponds were under more intensive management practices (higher Do values and use of aeration), with higher levels of ammonia input. Also, the growth rate of phytoplankton considered was much higher (g_{max} =1.3) than that of my model (g_{max} =1).

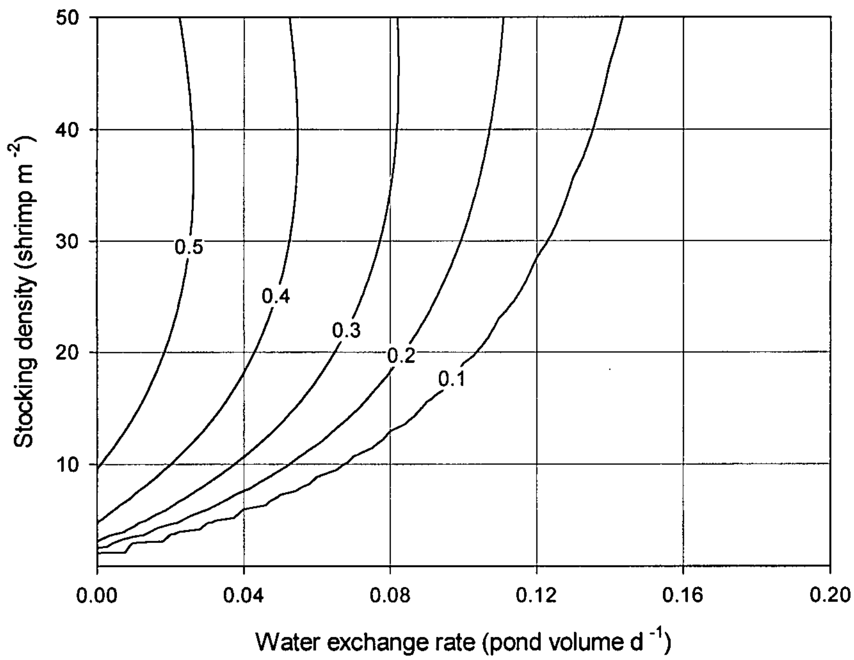

Fig. 5-35. Nitrogen in sediments as a fraction of total N outputs

Increasing W values was associated with a higher level of ammonia in the pond during the production cycle, which would make the system more vulnerable to phytoplankton fluctuations or pH changes, as already discussed in Sec.5.1.3. The increase in the amount of dissolved N in the effluent was also accompanied by an increase in the proportion of N lost through volatilization (Table 5-4), which in practice would be difficult to recuperate. However, from the farmer's perspective, the loss of more than one third of surplus N, as ammonia through volatilization, might be highly desirable from the economic point of view for its removal.

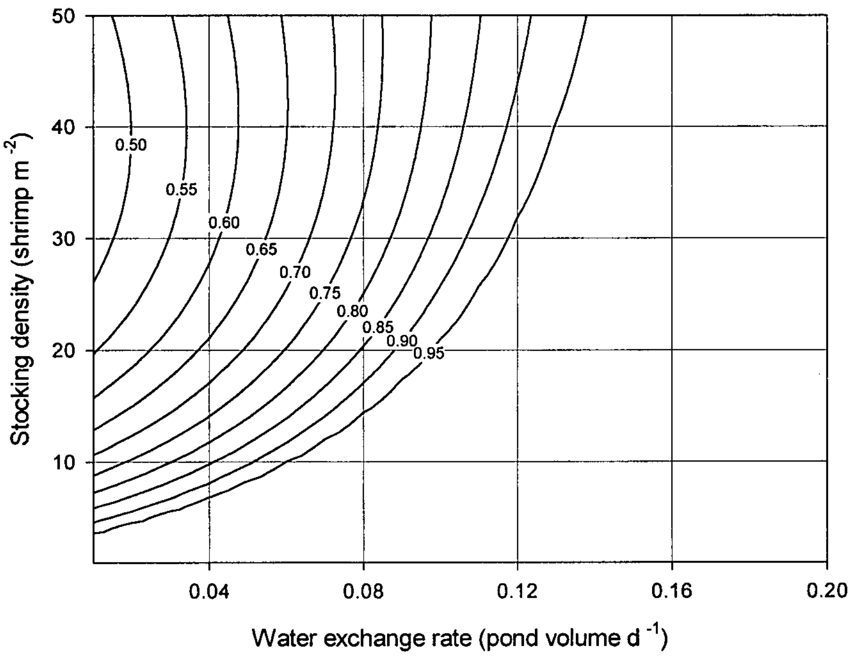

Fig. 5-36. Fraction of effluent nitrogen in dissolved form.

The effluent may be passed to a treatment pond and the dissolved N may be utilized by macroalgae, mangrove or grasses, which would increase the value of NP (through an increase in NE) if the product were considered as part of the harvest (i.e., extending the boundaries of the shrimp farming system). Shrimp farming effluents have been used in the production of macroalgae, which were able to remove up to 64% of TAN (Franco-Nava et al. 1999). The farming of macroalgae could be pursued with the goal of producing an additional species, or to remove excess N if acting as a biofilter only.

However, the consumption of oxygen by those plant species should not affect the system otherwise.

Table 5-4. Comparison of the baseline case and that predicted by the model for highest Nitrogen Productivity (NP) at levels of 95% and 99% of effluent N in dissolved form. Comparison was made at a survival rate of 0.7, as in the baseline case, and at water exchange rates (W) below $0.2 \cdot d^{-1}$.

	Do	*W*	**Effluent N dissolved (in %)**	**Surplus N**			
					%		**Total (kg·ha·cycle)**
				Effluent*	**Volatilization**	**Sediments**	
Baseline	15	0.04	73	21.57	30.24	48.17	150.77
Case	15	0.082	95	42.86	40.53	16.59	151.41
	15	0.095	99	48.32	41.12	10.56	152.10
Optimal	37.25	0.00	0	4.74	18.51	76.76	370.93
Case	35.59	0.128	95	48.77	36.24	14.97	360.12
	30.50	0.134	99	52.51	37.21	10.28	308.96

*Includes N remaining in the pond in the last day, which will be discharged at harvest

Monocultures have been considered as inherently inefficient because a single species is not able to utilize the nitrogenous wastes. Polyculture and integrated aquaculture are considered more ecologically efficient and there is evidence in favour of systems with higher biodiversity as they show increased stability, although there is controversy regarding this issue (Holling et al. 1994). Recent attempts to farming marine shrimp in complete freshwater have had some success in west central Mexico (Avila-Tamayo 1998; De la Torre-Escobosa 1998). The potential to reutilize N through the use the effluent in agricultural irrigation imparts these shrimp farming systems an advantage over the traditional brackish and marine systems. Still, those farms are situated within important agricultural areas and the demand for water may give rise to potential friction with other resource users. The issue is highly interesting and deserves further investigation.

In practice, accumulated sediment is commonly resuspended at harvest, when the pond is drained, or removed at the end of the farming period, as already discussed in Sec. 5.2. The use of moderate increases in *W* would reduce considerably the amount of N remaining in the sediments as predicted by the model (Fig. 5-35).

Except for pumping costs, water is not a cost item in shrimp farming production, or in aquaculture production in general. As water becomes more difficult to obtain in appropriate quantity and quality, its cost may rise substantially (Shang, Leung, and Ling 1998). This may have important implications for the recuperation of N and the trade-off of recuperating it in the effluent against the increased cost of pumping need to be analyzed. The potential long-term, negative effects of N left in sediments or the cost of their removal and disposal needs to be included in the analysis.

5.4 Overall comparison of the Mexican case and best operating conditions

The overall comparison of the Mexican baseline case and the model predictions for best operating conditions highlighted important implications, and various trade-offs that require further research. Those are:

1) The model predicted that optimal *Do* values for the Mexican case, if survival is maximized, are those around 30 shrimp·m $^{-2}$ (Table 5-5), a two-fold value of the

average stocking density currently utilized in Northwest Mexico. As the goal should be to maximize survival, the optimal *Do* can be considered also as a "maximum". Higher *Do* would result in higher ammonia levels in the water column (Fig. 5-20) with a lower AAC, or decreasing capacity of the environment to assimilate wastes (Fig. 5-3) making the system more fragile. These analytical results concurred with general estimates of a 20-30 shrimp·m $^{-2}$ range of maximum *Do* for shrimp farming in outdoor ponds (Briggs 1994), as already discussed in Sec. 5.1.1.2.

2) Assuming that survival rate remains unchanged (0.7), by adopting optimal operating conditions with no water exchange yield increases by almost 150% and water usage decreases by more than 90%, but N deposition in sediments increases by almost 300% (Table 5-5 and Table 5-6). The potential effects of these sediments on the water quality of subsequent production cycles, and the cost of the removal and disposal of them against the savings in water pumping need further investigation.

3) Assuming that survival rate remains unchanged (0.7), by adopting optimal operating conditions with water exchange, yield increases by almost 140% and N loss per unit of production to effluent and sediments decreases by almost 25%, but absolute water usage increases by more than 180% and ammonia level in the water column increases more than three-fold (Table 5-5 and Table 5-6). The absolute amount of N surplus discharged through the effluent into the environment increases by more than 450% (Table 5-6), mainly in dissolved form, which may be advantageous to its recuperation, as already discussed in Sec. 5.3.3. However, research is required to examine the potential effects of this higher discharge volume of dissolved N on

adjacent aquatic ecosystems.

Table 5-5. Comparison of the Mexican baseline case and the optimal cases (and their corresponding stocking densities, *Do*, and water exchange rates, *W*) with regard to yield, FCR and water usage as predicted by the model, at survival rates of 1.0 and 0.7

Survival rate	***Do***	***W***	**Yield ($t \cdot ha^{-1}$)**	**FCR**	*** Water usage** *Per unit of production ($m^3 \cdot t^{-1}$ shrimp)*	*Per area per cycle ($m^3 \cdot ha^{-1} \cdot cycle^{-1}$)*
			Baseline case			
1.0	15	0.04	2.02	2.02	28766.3	58000
0.7	15	0.04	1.59	2.28	36491.8	58000
			Optimal case			
1.0	28.99	0	3.9	2.02	2566.2	10000
0.7	37.25	0	3.95	2.28	2533.6	10000
			Optimal case for N recuperation (95% of effluent N in dissolved form)			
1.0	29.76	0.118	4.00	2.02	37897.8	151600
0.7	35.59	0.128	3.77	2.28	43382.4	163600

*Includes water required to fill the pond. FCR= Food Conversion Ratio

Table 5-6. Comparison of the Mexican baseline case and the optimal cases with regard to ammonia levels in the pond and N loss as predicted by the model, at survival rates of 1.0 and 0.7

Survival rate	**Max. TAN ($mg \cdot L^{-1}$)**	**Nitrogen Loss** *Per unit of production ($kg \cdot t^{-1}$ shrimp)*		*Per area per cycle ($kg \cdot ha^{-1} \cdot cycle^{-1}$)*		
		Atmosphere	*Effluent + Sediment*	*Effluent*	*Atmosphere*	*Sediment*
			Baseline case			
1.0	0.39	24.4	40.1	36.0	49.25	77.9
0.7	0.36	28.7	48.8	32.52	45.61	72.6
			Optimal case			
1.0	0.46	15.7	45.1	16.3	61.18	237.0
0.7	0.51	17.5	54.1	17.7	69.12	286.6
			Optimal case for N recuperation (95% of effluent N in dissolved form)			
1.0	0.97	31.1	29.6	162.3	124.28	39.0
0.7	0.97	34.8	36.9	179.4	131.29	49.2

5.5 Strength and weakness of the approach

The model was based on the best available scientific information, for both the shrimp and the water subsystems. The simulation model allowed an examination of the effect of important management practices such as those that intensify production, on the fate of N in shrimp farming ponds. The identification of important processes provide valuable insights to implementing management schemes that contribute to increasing NE and to maintaining or improving AAC. While previous models of N dynamics assessed either the origin (Montoya et al. 1999) or the fate (Lorenzen, Struve, and Cowan 1997) of ammonia in shrimp farming ponds, my model assessed both. Therefore, my model "connected" external inputs to external outputs, and evaluated the N flow through the pond.

Nevertheless, the simulation model excluded important environmental factors, such as dissolved oxygen, pH, temperature and salinity, which may affect considerably the N partitioning in the pond system. Those factors may affect importantly the shrimp subsystem (e.g., shrimp growth and excretion) and/or the water subsystem (e.g., assimilation of ammonia by phytoplankton and volatilization). Further research is needed to incorporate these factors into the approach and examine their interactions.

5.6 Application of the model

Predictions of the model, with regard to best operating conditions, can be readily utilized as a benchmark to establish research objectives in the farming of shrimp in Northwest

Mexico. The validation of model predictions for the Mexican case study will require implementing monitoring programmes in shrimp farms and the use of nitrogen budgets in the field. Due to the similitude of the Northwest case study with shrimp farming practices in the rest of Mexico, as well as in many other parts of Latin America and worldwide, the use of the model may be extended to assess those shrimp aquacultural practices.

The model may be applied to calculate best operating conditions, through nitrogen interactions, in the farming of other species under different production methods. Estimation of nitrogenous waste production (such as ammonia) can be based on simple principles of nutrition for many aquatic farmed species (Cho et al. 1994; Goddard 1996). However, the application of the model appears more suitable to land-based systems where there is a higher control of both water fluxes (i.e., water inflow and effluent) and potentially of phytoplankton populations. Further research is required in both shrimp farming and other forms of aquaculture, with regard to the contribution of natural N flows (e.g., N fixation) to aquacultural production, the incorporation of managed N (e.g., feeds and fertilizers) into natural processes, and the dynamics of phytoplankton towards the assimilation of nitrogenous wastes.

5.7 Thesis contribution and future directions.

Science develops incrementally, as does its application, driven largely by both curiosity and societal concerns. Aquaculture, as a discipline of recent scientific enquiry is subject to that trend.

The thesis focused on nitrogen as the assessment tool rather than on the more common monetary metric. Nitrogen was selected as it is a less volatile index than economic instruments. The thesis however provides examples of the conversion of the nitrogen currency to monetary values as an approach by which shrimp farmers may make informed decisions. The utility of nitrogen as a metric is that it can represent both production (protein) and environmental contamination (e.g. ammonia, phytoplankton and sediments).

The research is based on the model developed by Lorenzen et al. (1997) for shrimp farming in Thailand. A number of deficiencies in the original model were identified and some of these were confirmed by personal communication; particularly, there was a need to incorporate shrimp physiology and nitrogen partitioning to account for the production-related aspects, including the origin of ammonia. I have incorporated these considerations in the overall model. However, the dynamics of both e.g. shrimp life stages and seasonality, need further refinement.

The thesis moves shrimp farming to a more focused systems approach. Still, it requires further refinement in order to be more readily applicable to shrimp aquaculture. For example, recent research suggests that other components and processes of the shrimp pond system, such as bacterial production, may be important as a source of grazing and for the assimilation of nitrogenous compounds (Avnimelech 1999). Nitrogen entering shrimp ponds through deposition from the atmosphere may also play an important role in the aquaculture ecosystem.

The thesis also advances the science towards a better understanding of the effect of management practices on the outcomes e.g. production, production per unit of cost, partitioning of N within the water column and the biological component of the sediment. The model thus represents a move towards a development of a tool useful for both aquaculture practitioners and policy makers.

Shrimp farmers can make decisions with regard to the benefits and costs of changing their management practices. For example, increasing stocking densities and reducing water exchange would reduce water usage and pumping costs, but a higher amount of sediments would have to be removed and disposed of at the end of the farming period. The coproduction of macroalgae may result in an extra income for the farmer, although pumping costs would be higher due to the increased water exchange required to deliver dissolved nitrogen in the effluent. The benefits and costs of maximizing shrimp survival, through an improvement of the rearing environment, may also be evaluated.

Policy makers may use the outcomes of the model to develop policies that would support, or encourage, certain production practices. For example subsidies, or incentives, may be directed towards farming practices that reduce contamination of adjacent environments through a reduction in water exchange for example, or to the practice of polyculture that would reduce the amount of nitrogenous wastes discharged through the effluent.

CHAPTER 6. CONCLUSIONS.

This nitrogen-based analysis framework that I developed is capable of identifying dominant processes and assessing the effect of important management practices, which affect shrimp production and water quality. The analysis is based on two of the main management variables that identify intensification in aquaculture, stocking density and water exchange rate. Therefore, such analysis identifies important environmental interactions that may occur with the intensification trend in aquaculture, affecting both production and the adjacent environment.

The main value of the nitrogen-based approach is the screening of best operating conditions (i.e., best management practices) which may serve as a benchmark of long-term, productive farming systems against which to compare the performance of shrimp production enterprises in Northwest Mexico. The next phase to assess the analytical framework would be to corroborate model predictions through continuous monitoring and the use of N budgets in the field.

The analysis utilized aquatic simulation modelling and farming systems approaches to productivity assessment. The merging of both approaches appears useful to address, in a quantitative manner, the performance of aquacultural systems.

The model that I developed is amenable to simulation and analytic techniques. Simulation modelling of N fluxes represents an important aid to predict the partitioning of N in shrimp farming ponds, particularly for compartments difficult to measure such as sediments and volatilization. By studying this simplified model it is possible to attribute aquacultural systems behaviour to a specific process or management relationship. The model then, can be linked to a more complex framework for the analysis of wider environmental interactions.

Values of NE and AAC, and the resultant NP suggest that an increased N loss per unit of production and/or decreased capacity to assimilate ammonia may be interpreted as increasing self-pollution, that may affect the production process and potentially the adjacent environment. In other words, a decreasing ratio of N in harvested protein to total N input implies that excess N is entering the environment through aquaculture.

Feeds and feeding practices appear as the most important factors to manipulate towards improvement of NE. With regard to AAC, the growth rate of phytoplankton and its sedimentation rate represent key processes. Still, the manipulation of these processes appears difficult at present, and represents an important area for research.

Although the model predicts that an increase in shrimp stocking density and a reduction in water exchange results in a higher NP for the Mexican case, the amount of N remaining in the sediments increases, which may have important biophysical and economic implications associated with their removal and disposal. Considerable reductions of water exchange may have important implications in areas of high temperature (i.e., high evaporation) such as some areas in Northwest Mexico.

Increased densities convey an increased risk of water quality deterioration and disease, particularly in the face of unstable phytoplankton populations. Moderate increases in water exchange, traditionally used to improve water quality in the pond, give the opposite result. However, moderate increases in water exchange lead to a considerably higher amount of dissolved N in the effluent, with a higher potential for its recovery.

Excess N from aquaculture may be an important resource for the production of additional aquacultural or agricultural species (i.e., polyculture or integrated farming) particularly where the effluent is discrete and can be managed. Excess N may be a common denominator of various important interactions between aquaculture and agriculture, some of which may affect negatively each other. However, N may be the parameter for integration of the sectors, as both, aquaculture and agriculture, could use excesses from each other. Land-based aquaculture, such as shrimp farming, has comparative advantages over water-based aquaculture and various agricultural practices in the reutilization of N waste. The coproduction of other species through nitrogenous wastes may add useful N outputs (i.e., products) in the Nitrogen Productivity calculation, and improve the performance of the production system. This represents an important research avenue worth exploring by the shrimp farming industry in Mexico.

The NP indicator may be developed into an index, such as a Nitrogen Productivity Index (NPI) to evaluate the behaviour of a particular aquacultural system intertemporally with respect to N. Comparing among systems (i.e., interspatially) may be more difficult due to specific characteristics of systems. Similar to the economic approach to multiple input-output productivity, the NPI may be set at 1 (or 100) for a particular year (the base year) and the NP computed for all other years relative to the base. Therefore, an aquacultural system will tend to be more productive in the long-term if it has a non-negative trend in NPI. This analysis may be extended, by drawing the shrimp farm boundaries, in order to estimate the Nitrogen Efficiency (or Cost) at a higher spatial scale. For example, the production of shrimp feeds requires the cultivation of agricultural products and the capture and processing of wild fish into fishmeal, which can be analyzed in terms of nitrogen flows and their partitioning.

An analysis such as this may be useful for Mexican shrimp farmers to become acquainted of the influence of their management practices on the fate of N in the pond environment and the surrounding waters. It is important that socioeconomic analyses are conducted in parallel to strengthen the predictions of my model into a broader scale of economic and social imperatives. For example, there should be an indication of the optimal system (i.e., production method) being financially profitable.

In conclusion, a nitrogen-based analysis provides a more comprehensive understanding of how intensive and semi-intensive aquacultural systems behave with regard to both biophysical and socioeconomic environments.

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