An organic matter and nitrogen dynamics model for the ecological analysis of integrated aquaculture/agriculture systems: I. model development and calibration

D.M. Jamu 1, R.H. Piedrahita *

Department of Biological and Agricultural Engineering, University of California, One Shields Avenue, Davis, CA 95616, USA

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Abstract

A dynamic and mechanistic mass balance model (Integrated Aquaculture/Agriculture System model, or IAAS) for predicting nitrogen and organic matter outputs from aquaculture ponds and their subsequent recycling in conventional agriculture practices has been developed and calibrated using data from Honduras, Thailand and Malawi. The model, developed using Stella™ modeling software (High Performance Systems, Inc. Hanover, New Hampshire), simulates individual fish growth, organic matter, nitrogen (organic, total ammonia and nitrate), dissolved oxygen and phytoplankton, crop growth, soil nitrogen concentrations and soil water balance. Processes included in the model are fish growth, crop biomass growth, allochthonous and autochthonous organic matter production, organic matter decomposition, nitrogen input, nitrogen mineralization, nitrification, denitrification, diffusion, uptake and leaching. The model has been calibrated using literature and observed parameter values from experiments. The calibration procedure involves running the model using inputs from observed data, comparing the model output to observed data, and making appropriate adjustments to parameter values until a general fit between the model and observed values is observed. The structure of the model allows users to modify parameter values to suit different simulation scenarios via a user interface display that also includes graphs and tables for model output. © 2002 Elsevier Science Ltd. All rights reserved.

Keywords: Nitrogen cycling; Organic matter; STELLA; Aquaculture; Dynamic model

1. Introduction

Integrated aquaculture/agriculture systems are increasingly being promoted as an environmentally sustainable method for producing aquatic and terrestrial crops. In integrated systems, wastes from one system component are recycled as inputs to another system component. Through waste recycling, integrated aquaculture/agriculture systems can be used to treat aquaculture effluents, increase farm productivity through efficient resource utilization, spread financial risk through diversification and reduce system nutrient losses (Singh et al., 1996; Williams, 1997).
To take advantage of the attributes of the integrated aquaculture/agriculture system stated above, adequate scientific information is required on the important processes governing nitrogen dynamics and system productivity, and how they affect different functions of the integrated system. Because of the complexity of integrated aquaculture/agriculture systems (Edwards et al., 1988), it is impractical to conduct basic field experiments that answer questions concerning the contribution of different processes to the structure and function of the system (Thornley and Verberne, 1989). Previously, researchers have used conventional tools for agroecosystem analysis like energy budgets and bioresource flow conceptual models to describe and analyze integrated systems. However, this approach does not capture the dynamic properties of the system (Conway, 1987; Lightfoot et al., 1993).

Simulation models are useful tools in the analysis of complex systems and biogeochemical cycling of nutrients (Anderson, 1992). Models assist scientists in conceptualizing, summarizing and analyzing complex phenomena (Hall and Day, 1977). Simulation models have been used to simulate different terrestrial and aquatic systems (Hall and Day, 1977; France and Thornley, 1984). Existing models developed to analyze integrated aquaculture/agriculture systems have largely been empirical (van Dam, 1995; Schaber, 1996, steady state (Dalsgaard, 1996) or have failed to incorporate major components of the system (van Dam, 1995; Dalsgaard, 1996; Schaber, 1996). The empiricism of current integrated systems models, their lack of dynamical properties, and exclusion of important system components limit their extendability and utility in the ecological analysis of dynamic ecosystems.

A new model (Integrated Aquaculture/Agriculture System model or IAAS) has been developed for simulating organic matter and nitrogen dynamics through aquaculture and integrated conventional agriculture. Given the scope of the material related to the description of IAAS, it was decided to present it in two parts. The objective of this first part is to present an overview of the model and some details of significant model components. A brief overview of the model calibration is also presented in this first part. Application of the model to identify important processes and priority areas for future research in integrated aquaculture/agriculture systems is presented in the second paper (Jamu and Piedrahita, 2002). Further details are available elsewhere (Jamu, 1998).

2. Model overview

The model described in this paper is a dynamic, deterministic and mechanistic model that simulates fish and crop production, organic matter, phytoplankton, dissolved oxygen, terrestrial leaf area index, fresh and stable organic matter content in terrestrial soil. For the solved oxygen (DO), and nitrogen dynamics in an integrated aquaculture/agriculture system. The model builds upon existing pond ecosystem models (Nath, 1996; Piedrahita, 1990) and a general crop model (SUCROS1) (Spitters et al., 1989). The model, developed using Stella™ modeling software (High Performance Systems, Inc, Hanover, New Hampshire), has a simulation time step of 0.125 d and the model differential equations are solved numerically using Euler’s method. The model can be used to study long-term (i.e. several years) trends by simulating consecutive production cycles.

The aquaculture ecosystem model developed here is, however, markedly different from previous models because of its explicit treatment of organic matter and nitrogen dynamics and the inclusion of aquaculture pond sediments as part of the mass balance calculations. Nitrogen is treated as a major component in this model because of its role as a major limiting nutrient controlling the diversity, dynamics and productivity of terrestrial and aquatic systems (Vitousek et al., 1997). On the other hand, organic matter provides the main link for integration of aquaculture and agriculture, and plays a pivotal role in the recycling of nutrients in ecological systems (Ruddle and Zhong, 1984; Voght et al., 1986).

Although the agriculture component model incorporated in the integrated model is not very different from the original SUCROS1 model, the soil nitrogen and water balance models have been greatly simplified, and now include simple and reduced statements to describe soil water and nitrogen processes and their effects on crop growth. The model strengths over existing integrated aquaculture/agriculture models (van Dam, 1995; Dalsgaard, 1996; Schaber, 1996) are that the present model is largely mechanistic and includes the important components of the integrated aquaculture/agriculture system, while previous models are largely empirical and exclude the major system components/processes. The model has been developed using a modular approach where modules, sub-modules, and sub-models are developed based on biophysical and functional similarity of the state variables (Table 1). The model consists of two modules, i.e. aquaculture and agriculture (Table 1). Each module consists of mass balance calculations for the state variables which describe the state or condition of the system (Shoemaker, 1977). Figure 1 shows the major components (sub-modules) of the model and the main pathways and linkages between them and the surrounding environment. The model contains 19 state variables that are expressed in units of concentration: kg ha\(^{-1}\) for nutrient and biomass concentrations and number of individuals ha\(^{-1}\) for population densities (Table 1). Thirty-nine variables and parameters can be modified by users. Initial values must be input for the following state variables: fish weight, chlorophyll \(a\) concentration, dissolved oxygen, terrestrial leaf area index, fresh and stable organic matter content in terrestrial soil. For the...
Table 1
Modules, sub-modules, state variables and external forcing functions for N dynamics in integrated aquaculture/agriculture system

<table>
<thead>
<tr>
<th>Module</th>
<th>Sub-module</th>
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<td>Fish biomass, phytoplankton and organic matter biomass, Total ammonia nitrogen (TAN), Dissolved oxygen (DO)</td>
<td>Water temperature, solar radiation</td>
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<td></td>
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<td>Fish population</td>
<td>Fish population</td>
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<td>Feed (natural, artificial); nitrogen and carbon concentration</td>
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<td></td>
<td>Phytoplankton</td>
<td>Phytoplankton biomass, DO, TAN and nitrate-N; fish biomass</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>Water quality</td>
<td>Organic matter biomass; nitrate-N and TAN concentration; organic nitrogen; fish biomass</td>
<td></td>
</tr>
<tr>
<td>Fish pond sediment</td>
<td>Pond sediment organic matter</td>
<td></td>
<td>Carbohydrates and crude protein, cellulose, lignin and stable organic matter concentrations; TAN concentration; nitrate-N concentration, organic nitrogen, DO</td>
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<td>Pond sediment nitrogen</td>
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<td>Soil evaporation</td>
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<td>Fresh organic matter</td>
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<td>Soil temperature</td>
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<td>Soil nitrogen</td>
<td>Crop biomass; soil water; organic matter biomass, water quality</td>
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<tr>
<td>Terrestrial crop</td>
<td>crop growth (leaf, stem, storage/grain and root)</td>
<td></td>
<td>Root biomass; root length; storage biomass; fresh organic matter biomass; soil nitrogen; pond sediment organic matter</td>
<td>Air temperature; soil water, solar radiation</td>
</tr>
</tbody>
</table>
other state variables (Table 1), initial values can be set to zero when their values are not known.

The aquaculture and agriculture modules can be run separately or as a single integrated aquaculture/agriculture model. Integration of aquaculture and agriculture is established through the creation of water, organic matter, and nutrient pathways between the two modules. The major pathways between the two systems are through the utilization of: (a) agricultural plant wastes as pond fertilizers, (b) pond water to irrigate agricultural crops and (c) pond sediments to fertilize agricultural crops. The transfer of plant waste to fertilize the aquaculture pond is assumed to occur after harvest of the agricultural crop and irrigation occurs during the fish production cycle. In cases where sediment organic matter is removed from the pond at the end of the fish production period, the sediment mineral nitrogen and organic matter concentrations are used as the initial sediment values for the next production cycle.

3. Agriculture module components and processes

The agriculture module is used to simulate crop biomass growth, soil organic matter and nitrogen concentrations, and crop soil water loss through crop transpiration and soil surface evaporation. The module consists of two main sub-modules: terrestrial crop and soil. Each of the sub-modules is further broken down into sub-models as shown in Table 1.

3.1. Terrestrial crop sub-module

The terrestrial crop sub-module simulates the crop biomass growth rate as a function of soil water availability, air temperature, solar radiation and soil nitrogen concentration. The crop biomass growth rate is simulated using the SUCROS1 source–sink model (Spitters et al., 1989). In this approach, gross assimilated carbon feeds a pool of carbohydrates, and the carbohydrates are partitioned into roots, shoots, and storage organs as a function of phenological age of the plant. Information on gross partitioning coefficients of assimilated carbon to different crop components, crop phenological development rate, maximum carbon assimilation rate and respiration rates of the different crop organs is available from the literature (Penning de Vries, 1982). The crop considered in the model is corn (Zea mays).

3.2. Terrestrial soil sub-module

The second sub-module in the agriculture system component of the model is the terrestrial soil sub-module (Table 1, Fig. 1). The terrestrial soil sub-module simulates soil nitrogen and organic matter concentrations, and soil water balance. The modeling of soil nitrogen availability and uptake follows the approach of van Keulen and Seligman (1987) whereby nutrients are available through decomposition of organic material based on first order kinetics. The processes affecting soil nitrogen are fertilization (inorganic and organic), organic matter decomposition, leaching, and crop uptake. It is assumed that nitrogen input through wet deposition in rainfall is insignificant relative to the other sources, and nitrogen is primarily available to the crop plants through mass flow with the transpiration stream.

The terrestrial soil gains organic matter through root death, reincorporation of crop waste and pond water irrigation. The terrestrial soil loses organic matter primarily due to decomposition. In the model, a multiple pool approach, where the organic matter is divided into several pools based on the reactivity of each pool (Thornley and Verberne, 1989), is used to simulate organic matter decomposition. In the present model, the organic matter is divided into four organic matter pools (easily decomposable, moderately easy to decompose, slowly decomposable, and stable). The organic matter pools decay at different rates based on their reactivity expressed in a unit of d^{-1} (in parenthesis) as shown in Eq. (1) (van Keulen and Seligman, 1987).

\[
\text{Easily decomposable (0.8)}
\]
\[
\text{Moderately decomposable (0.05)}
\]
\[
\text{Slowly decomposable (0.0095)}
\]
\[
\text{Stable (8.3 \times 10^{-5})}
\]

The easily decomposable organic matter is made up
of readily metabolizable subgroups of organic matter, composed primarily of carbohydrates and crude proteins. The moderately decomposable organic matter is made up of cellulose, and the slowly decomposable organic matter is composed of lignin compounds. Stable organic matter is defined as the organic matter that has undergone decomposition at least once (van Keulen and Seligman, 1987).

Soil water balance is simulated using simplified calculations for a homogeneous soil layer as affected by evapotranspiration rates, soil type and soil water content at field capacity (Addiscott and Whitmore, 1987). For simplicity, clay and loam are considered to be the two main soil types. Any intermediate soil types between clay and loam are treated as either loam or clay depending on the dominant soil mineral component, e.g. loamy clay is classified as clay. This classification removes the need for fine delineation of soil types.

The leaching of water from the agriculture component through downward transport of water outside the corn root zone is calculated as the difference between actual soil water content and water content at field capacity for the soil type. Water loss due to surface runoff is not taken into account in the calculation of soil water. Irrigation is calculated based on a fixed frequency schedule whereby the total amount of water for irrigation is calculated as the cumulative crop evapotranspiration between irrigations. Irrigation is equal to zero when rainfall is greater than crop evapotranspiration.

4. Aquaculture module components and processes

The aquaculture module consists of the fishpond water column and fishpond sediment sub-modules. The fishpond water column sub-module consists of sub-models that are used to simulate fish growth and population, feed quality, phytoplankton biomass, and water quality (Table 1). The fishpond sediment sub-module is used to simulate pond sediment organic matter and nitrogen dynamics (Table 1).

4.1. Fish growth

The fish growth model simulates the growth of an individual fish using a fish bioenergetic model adapted from Bolte et al. (1994) that is parameterized for Nile tilapia (Oreochromis niloticus). The model (Bolte et al., 1994) has been modified to include the effects of feed quality, feed preference and digestibility of different feed resources on fish growth. The model has also been modified to include a multiplier that accounts for genetic differences between Nile tilapia and other tilapia species. Details of these modifications are described elsewhere (Jamu and Piedrahita, 1996; Jamu, 1998).

4.2. Fish population

The total number of fish in the aquaculture pond is dependent on the initial number stocked, mortality and reproduction:

\[
\frac{d(population)}{dt} = \text{stocking} + \text{birth} - \text{death} - \text{harvest} \tag{2}
\]

Fish harvest in Eq. (2) occurs at the end of the production period. It is assumed that stocking, birth and harvest are equal to zero during the production cycle. The assumption of zero birth rates may be a simplification in so far as mixed sex tilapia production is concerned. However, inclusion of births would require the consideration of age-structured population model. The simulation of age structure is not important in the present model since the effects of recruits on the pond carrying capacity is implicitly included in the fish bioenergetic growth model. Fish population in the pond is assumed to be affected by mortality rates only. Fish mortality in aquaculture ponds is a result of a variety of factors like high-unionized ammonia concentrations, low DO conditions, predation, poor handling during sampling, and disease. While it is desirable to predict fish mortality from simulated water quality conditions, the problems of identifying the actual causes of fish mortality makes this approach impractical (Nath, 1996). The fish mortality rate (death in Eq. (2)) is expressed using a logistic equation (Pearl and Reed, 1920):

\[
\frac{dP}{dt} = \gamma(P_a-P)P \tag{3}
\]

where: \( \gamma = \text{mortality rate (d}^{-1}) \); \( P_a = \text{management allowable fish population} \); \( P = \text{fish population at time, t (fish, ha}^{-1}) \). The management allowable population is calibrated based on survival data recorded at fish harvest.

4.3. Feed intake and preference

Tilapias show a preference for phytoplankton, which disappears under phytoplankton-limited conditions (Schroeder, 1978). In nature, tilapias have a predominantly herbivorous diet consisting of phytoplankton, vegetable debris, zooplankton, and benthic organisms (Philippart and Ruwet, 1982). Under waste-fed conditions common in integrated aquaculture/agriculture systems, supplementary feed is applied to ponds to augment the natural food supply. Therefore, both natural and supplemental feed may be available to the fish. In the model, it has been assumed that tilapias prefer natural feed (phytoplankton and detritus) to artificial feed in the following order: phytoplankton>detritus>supplemental feed (Schroeder, 1978; Spataru, 1978; Philippart and Ruwet, 1982). While this preference series may be true for waste-fed tilapia ponds, it may not hold for systems
where fish are fed high quality feeds. Apart from feed quality, fish physiology and learning behavior may also affect fish preference for certain feed types. Therefore, the feed preference series adopted here should be viewed as a simplification of the complex behavioral and physiological factors that influence feed preference by fish in aquaculture ponds. The feed intake for feed resource \( i \) is dependent on the total feed intake rate, a preference factor (analogous to a half saturation constant), and the feed concentration. Feed intake rate for the \( i \)th feed resource can be generally presented as:

\[
r_i = \frac{c_i}{Ks_i + c_i}
\]  
(4)

where \( r_i \) = intake rate for feed \( i \) (g d\(^{-1}\)); and \( R \) is the total feed intake rate (g d\(^{-1}\)) (Nath, 1996), \( Ks_i \)=half saturation constant for uptake for the \( i \)th feed resource (mg l\(^{-1}\)), and \( c_i \)=concentration of the \( i \)th feed resource (mg l\(^{-1}\)). The \( Ks_i \) value for feed uptake captures the effects of feed preference by fish and feed concentrations on the feed intake rate. Based on Eq. (4), fish in the model are allowed to initially feed on phytoplankton by setting the \( Ks \) for phytoplankton uptake by fish to a lower value relative to the \( Ks \) for detritus. When the fish cannot meet their daily requirements, they supplement their total feed requirements by consuming detritus and artificial feed. This approach requires the determination of feed uptake coefficients and feed uptake factors that are used to calculate feed intake rates for each of the three feed resources. The feed uptake coefficient is defined as the Monod coefficient for the feed resource expressing feed uptake as a function of its concentration and corresponding half saturation constant. For example, the phytoplankton uptake coefficient (\( PUC \)) is expressed as:

\[
PUC = \frac{Chla}{Ks_{Chla} + Chla}
\]  
(5)

where: \( Chla \)=measure of phytoplankton biomass density (chlorophyll \( a \), mg l\(^{-1}\)); \( Ks_{Chla} \)=half saturation constant for phytoplankton uptake (mg l\(^{-1}\)). A similar equation can be developed for the detritus uptake coefficient (\( DUC \)) as follows:

\[
DUC = \frac{D_{detritus}}{Ks_{detritus} + D_{detritus}}
\]  
(6)

where: \( D_{detritus} \)=the concentration of detritus in the water column (mg l\(^{-1}\)) and \( Ks_{detritus} \)=half saturation constant for detritus uptake (mg l\(^{-1}\)).

The feed (phytoplankton, detritus, artificial) uptake factor is defined as the fraction of feed consumed by the fish for each of the feed resources after accounting for the effects of feed concentration and preference. Since it has been assumed that tilapia prefer phytoplankton to other feed resources, the phytoplankton uptake coefficient is always equal to the phytoplankton uptake factor (\( PUF=PUC \)). Therefore, the phytoplankton uptake rate (\( r_{PUC} \), g d\(^{-1}\)) is equal to the feed intake rate multiplied by \( PUF \):

\[
r_{PUC} = R.PUF
\]  
(7)

The \( PUF \) is always less than one, although it can approach one if the phytoplankton concentration is much greater than the half saturation constant for uptake. Therefore the phytoplankton uptake by fish (\( r_{PUC} \)) will always be less than the total feed intake rate (\( R \)). The balance; \( R(1-PUF) \), will have to be met through fish feeding from the detritus and artificial feed pools. In the case of the present model the next preferred feed is assumed to be detritus and the detritus uptake factor (\( DUF \)) is calculated as:

\[
DUF = (1-PUF)DUC
\]  
(8)

The detritus uptake factor is then used to calculate the detritus feed intake rate (\( r_{Detritus} \), g d\(^{-1}\)) as:

\[
r_{Detritus} = R.DUF
\]  
(9)

If artificial feed is available, it will constitute the balance of the total diet, \( R(1-PUF-DUF) \). Therefore, the artificial feed uptake factor (\( AFF \)) is:

\[
AFF = 1-(PUF + DUF)
\]  
(10)

and the artificial feed factor by the feed uptake rate (\( r_{AFF} \), g d\(^{-1}\)):

\[
r_{AFF} = R.AFF
\]  
(11)

### 4.4. Feed quality

Fish growth in integrated aquaculture/agriculture systems can be affected by low feed quality. The effect of feed quality on growth is incorporated in the model based on the work of Urabe and Watanabe (1992) and the definition of a balanced diet as presented by Sterner and Hessen (1994). The model for the effect of feed quality on fish growth is expressed as:

\[
q = \frac{N:C_{F}}{Q_{c-E}^{*}} \quad \text{if} \quad Q_{c-E}^{*} > N:C_{F}
\]  
(12)

or

\[
q = \frac{N:C_{F}}{Q_{c-E}^{*}} \quad \text{if} \quad Q_{c-E}^{*} \leq N:C_{F}
\]  
(12)

where: \( Q_{c-E}^{*} = K_{c}(N:C_{C}) \) is the critical food nitrogen to carbon ratio below which fish production would be limited by nitrogen, \( N:C_{F} \)=nitrogen to carbon mass ratio in food (g N/ g C), \( K_{c} \)=carbon gross growth efficiency (g feed C incorporated in fish biomass/g feed C consumed), \( N:C_{C} \)=nitrogen to carbon mass ratio in fish (g N/g C). The parameter \( K_{c} \) is determined by calculating the ratio of fish biomass growth rate to food intake rate, all expressed in terms of carbon (Lucas, 1996). Feed N:C ratios in excess of the critical feed nutrient requirements results in decreased fish growth rates due to the meta-
bolic costs associated with the breakdown of excess nitrogenous products from the fish. However, low feed N:C ratios due to low protein feed and inadequate N fertilization are more common in waste-fed aquaculture where herbivorous and detritivorous fish such as tilapia are reared (Bowen, 1987) and the effects of high feed N:C ratios on fish growth have been ignored.

4.5. Phytoplankton biomass

The phytoplankton mass balance calculations incorporated in the IAAS model account for phytoplankton growth, grazing by fish, respiratory losses, settling, death and sporadic effluent losses through irrigation water. The phytoplankton sub-model is linked to: (i) the fish growth sub-model through fish grazing, (ii) organic matter dynamics through dead phytoplankton and, (iii) nitrogen dynamics through nitrogen uptake and mineralization of dead phytoplankton. Phytoplankton growth rate is a function of phytoplankton biomass, solar radiation, temperature, and nutrient concentrations. Nitrogen is considered to be the only limiting nutrient for phytoplankton growth based on the fact that nitrogen is the primary nutrient limiting productivity in tropical aquatic and aquaculture systems (Moss, 1969; Melack et al., 1982; Knud-Hansen et al., 1991). Phytoplankton settling and death are expressed as first order processes with respect to phytoplankton biomass. The temperature limitation coefficient is expressed as a “skewed normal” function (Svirezhev et al., 1984; Nath, 1996).

4.6. Water quality

The major parameters affecting water quality in the aquaculture module are organic matter, dissolved oxygen (DO), and total ammonia nitrogen (TAN). Organic matter decomposition results in the depletion of DO due to microbial respiration and nitrification. Decomposition of organic matter also releases nitrogen which undergoes ammonification to form TAN. TAN levels above the tolerance threshold for the fish and DO below the tolerance threshold for fish affect fish growth rates (Boyd, 1995).

The major sources of organic matter in a waste-fed aquaculture pond are applied feed and organic fertilizers. The organic matter mass balance calculations for the water column account for losses that occur during organic matter decomposition, pond effluent discharge during harvest and/or irrigation, and fish consumption. Organic matter is produced within the pond through phytoplankton death, fish death, and fecal production. In addition, the pond gains organic matter through organic fertilization and pond influent. The simulation of organic matter decomposition in the pond water column and sediment is based on the multi-G model of Westrich and Berner (1984). The multi-G approach is similar to the approach adopted in the modeling of terrestrial crop organic matter described earlier. The decomposition of organic matter can be expressed as:

\[ r_{gi} = -k_i T_c G_i \]  

where: \( r_{gi}= \) rate of decomposition for the \( i \)th group organic matter group (kg ha\(^{-1}\) d\(^{-1}\)); \( G_i = \) concentration of organic matter in the \( i \)th group (kg ha\(^{-1}\)); \( G_{tm}= \) the total concentration of organic matter (kg ha\(^{-1}\)); \( k_i= \) decay rate constant of the \( i \)th organic matter group as a function of the substrate C:N ratio (d\(^{-1}\)); \( T_c= \) temperature multiplier. The concentrations of different groups or organic matter fractions are determined from proximate analyses for different feeds or fertilizers being applied to the pond. Literature values (Gohl, 1981) can be used if the proximate analyses are not available. These data are input to the IAAS model by the user.

Dissolved oxygen concentrations are simulated using the DO sub-model that follows the procedures described by Piedrahita (1990). In this sub-model, the water column DO mass balance calculations include photosynthetic DO production, DO exchange with the atmosphere, consumption through biological (fish, phytoplankton and microbial populations) respiration in the water column and sediment oxygen demand. The demand for oxygen in the sediment is primarily due to the oxidation of TAN through nitrification, and of methane and hydrogen sulfide by a wide range of bacteria. Only nitrification oxygen demand in the sediment is modeled explicitly. The remaining oxygen sinks (methane and hydrogen sulfide oxidation, macrobenthos respiration) are combined into a single sediment respiration term that is calibrated from measured sediment respiration rates (Boyd and Pippopinyo, 1994). Sediment respiration rate is expressed as a function of temperature and water column oxygen concentrations (Walker and Snodgrass, 1986):

\[ r_{o_s} = -k_s O_2 \frac{O_2}{K_{O_2} + O_2} \]  

where \( r_{o_s}= \) sediment respiration rate (mg O\(_2\) l\(^{-1}\) d\(^{-1}\)); \( k_s= \) specific sediment respiration rate (d\(^{-1}\)); \( O_2= \) water column oxygen concentration (mg O\(_2\) l\(^{-1}\)), and \( K_{O_2}= \) half saturation constant for oxygen (mg O\(_2\) l\(^{-1}\)).

Water column and sediment TAN is produced from the ammonification of organic nitrogen which occurs during the decomposition of organic matter. The nitrogen transformations are simulated using first order models adopted in other fish pond ecosystem models (Piedrahita, 1990; Kochba et al., 1994). The rate of nitrogen mineralization is calculated as a first order reaction from the organic matter decomposition rate (Eq. (13)).
In addition to organic matter mineralization, the pond nitrogen mass balance also includes nitrogen fertilization, influent and effluent discharge, diffusion between sediment and water column, denitrification and fish excretion. Fish nitrogen excretion is modeled explicitly using mass balance calculations by expressing the fish bioenergetic model in terms of nitrogen. Any excess nitrogen not utilized by the fish is assumed to be excreted as TAN. The simplified equation for calculating fish TAN excretion is:

\[ r_{\text{TAN}} = r_F + r_{\text{FC}} + r_E \]  

(16)

where \( r_{\text{TAN}} \) = TAN production rate, \( r_F = \) TAN excreted during fasting catabolism, \( r_{\text{FC}} = \) TAN excreted during feeding catabolism and \( r_E = \) TAN excreted in excess of TAN requirements for growth and metabolism. All units in Eq. (16) are in kg ha\(^{-1}\) d\(^{-1}\). It is assumed that all the nitrogen in excess of the fish requirements for growth and respiration is excreted as TAN. In addition, the nitrogen model includes diffusion terms for nitrate nitrogen (\( \text{NO}_3^- - \text{N} \)) and TAN. The diffusion of \( \text{NO}_3^- - \text{N} \) and TAN is obtained from (Blackburn and Blackburn, 1992):

\[
\text{Diffusion rate} = \frac{(\text{Concentration difference})}{D \cdot \text{Depth}}
\]

(17)

where: Diffusion rate = diffusion rate for nitrogen species (kg m\(^{-2}\) d\(^{-1}\)), Porosity = void volume fraction (dimensionless), Diffusion coefficient (m\(^2\) d\(^{-1}\)), Concentration difference = difference in concentration between water and sediments (kg m\(^{-3}\)), and Depth = sediment depth (m). Porosity values are calculated from published bulk density and organic matter particle density values as follows:

\[
\text{Porosity} = 1 - \frac{\text{Bulk density}}{\text{Particle density}}
\]

(18)

where: Bulk density and Particle density have units of kg m\(^{-3}\). Mean values for sediment bulk and particle density from a pond sediment characterization study by Munsiri and Boyd (1995) are used in the model as default values. It is assumed that all the pore spaces in the sediment are occupied by water, therefore the volume of pore space in the sediment is equal to sediment pore water volume.

4.7. Pond sediment organic matter and nitrogen dynamics

The mass balance calculations for pond sediment include settling, resuspension, and decomposition. Organic matter settling and resuspension are represented as first order processes, while the multi-G model (Eqs. (13) and (14)) is also used to simulate decomposition of organic matter in the sediment. The sediment is divided into two zones: the upper 1 mm layer whose oxygen status is dynamic and dependent on the oxygen concentration in the water column, and an anaerobic layer at sediment depths greater than 1 mm. The first order decay constants for the different organic matter pools in the aerobic layer are similar to those adopted in the water column. However, for the anaerobic layer, lower (about 10–20% of the aerobic values, Prats and Llavador, 1994; Reddy and Patrick, 1984) decay rate constants are used to reflect the overall lower efficiency of other electron acceptors in the oxidation of organic matter.

Sediment nitrogen simulations are generally similar to those of the water column. However, in addition to diffusion, leaching, and nitrification, sediment pore water TAN is adsorbed by the sediment exchange complex. In this model, unidirectional movement of TAN to the exchange complex is assumed to occur during TAN adsorption. It is further assumed that equilibrium between porewater and adsorbed TAN is reached instantaneously. The maximum amount of TAN adsorbed by sediments is determined by a potential upper limit for TAN adsorption calculated from sediment cation exchange capacity (CEC) values (Mehrani and Tanji, 1974). The potential upper limit for TAN adsorption calculated from CEC values is only approximate, since not all exchange sites on the sediment are occupied by TAN as other cations compete for the exchange sites. TAN adsorption only occurs when the potential upper limit for TAN adsorption is greater than the TAN adsorbed by the sediment, i.e. when the exchange sites are not completely occupied by TAN. The TAN adsorption rate is defined as the rate at which pore water TAN is attracted to the sediment through physical and chemical processes (Tchobanoglous and Schroeder, 1987). TAN adsorption rate can be obtained from the following equation (Deizman and Mostaghimi, 1991):

\[
\text{TANadsorption} = (\text{PorewaterTAN}) (\text{SpecificTANadsorption})(\text{AdsorbedTAN})
\]

(19)

where: TANadsorption = rate of TAN adsorption by the sediments (kg ha\(^{-1}\) d\(^{-1}\)), PorewaterTAN = TAN in the pore water (kg ha\(^{-1}\)), SpecificTANadsorption = specific rate of TAN adsorption (d\(^{-1}\)) and AdsorbedTAN = Fraction of TAN adsorbed (Dimensionless). The fraction of TAN adsorbed by the sediment is expressed as (van Raaphorst and Malschaert, 1996):

\[
\text{AdsorbedTAN} = \frac{K_D}{(K_D + 1)}
\]

(20)

where: \( K_D \) = TAN distribution coefficient (dimensionless), defined as the ratio of sediment to pore water TAN.

In addition to the processes occurring in the sediment accumulated during the growing season, the model also incorporates processes occurring in the mineral compo-
ponent of the soil. There is a lack of information on the extent to which the mineral soil participates in pond sediment and water column processes. However, in the current pond sediment sub-module, the model boundaries are determined by considering the sediment sampling depth for the parameters of interest. In the data used here for model testing, a 5-cm core sampling depth was used to determine the nutrient and organic matter of pond sediments (Berkman, 1995). The volume of mineral soil involved in sediment nitrogen and organic matter processes is, therefore, defined as the difference between the sampling depth and the depth of accumulated organic matter multiplied by the pond area. Using this definition, the organic matter layer is assumed to be a distinct and separate layer from the mineral layer, an assumption validated by Munsiri and Boyd (1995). The nitrogen processes occurring in the mineral sediment are similar to those occurring in the organic matter layer. These nitrogen processes include Ficksian diffusion, leaching through water infiltration losses, denitrification of nitrate, and total ammonia adsorption by the soil exchange complex.

5. Model Calibration

Calibration values for parameters in the agriculture module were available from published reports because the model has been validated for a wide range of environmental conditions (Spitters et al., 1989). Therefore, only the aquaculture model was calibrated. The calibration procedure involved running the model using inputs from observed data, comparing model output against observations and making appropriate adjustments to parameter values until a general fit between model output and measured values was achieved. Given the scope and structure of the model, the calibration process was carried out sequentially for each sub-module in the aquaculture component model (Table 1).

Data inputs from the Pond Dynamics/Collaborative Research Support Program (PD/A CRSP) database (Berkman, 1995) were used to provide initial model values for state variables, site characteristics (elevation and latitude), and environmental variables (wind speed, solar radiation, air and water temperature). The PD/A CRSP database contains a variety of fish production and environmental data obtained during the conduct of different fish culture experiments conducted at five sites around the world. Data from the PD/A CRSP experiments conducted at the Rwawase Fish Culture Station, Butare, Rwanda (2.4' S and 29.45' E) were chosen for calibrating the model (Table 2). The site is located at an elevation of 1625 m and has a humid tropical climate with average annual temperatures between 14 and 28°C and a mean monthly humidity range of 59–83% (Bowman and Claire, 1996). The water supply to the station had a pH range of 6.5–7.0, and alkalinity of 17 mg l^{-1} as CaCO₃. The soils at the site are acidic (pH=4.5–4.8), and organic matter and cation exchange capacity range between 0.7 and 5.1% and between 4.5 and 17.6%, respectively. The dataset chosen for calibrating the model consisted of observations collected during the conduct of an experiment on fertilization of tilapia pond using plant wastes and chicken manure at 1000 kg ha⁻¹ wk⁻¹ and 500 kg ha⁻¹ wk⁻¹ dry weight, respectively. This dataset was chosen because it is the only dataset in the PD/A CRSP database where plant wastes were used as a pond fertilizer. Considering that plant wastes from crop systems provide a crucial link for integration, calibration of the model using this dataset provided realistic parameter estimates for the range of inputs normally used in waste-fed aquaculture systems.

6. Conclusions

The integration of aquaculture and agriculture activities is of primary interest for aquaculturists and agricultural ecologists as a means of sustaining system productivity and mitigating the negative environmental impacts of aquaculture effluents. The role of integration in sustaining system productivity and reducing the negative environmental impacts of aquaculture can only be enhanced if more information about the integrated system, module interactions and processes and mechanisms controlling the functioning of the integrated system is known (Fig. 1).

IAAS represents an important first step in formalizing a model that includes the most important components of an integrated aquaculture/agriculture system, and that quantifies the complex interactions between the different components of the system. For example, the inclusion of pond sediments as part of the mass balance calculations in the model is particularly important because of the role that sediments can play in pond water quality as well as in the recycling of nutrients to the agriculture component. Because the model has been developed in a modular format, individual sub-modules and sub-models can be modified independently without affecting the integrity of the model. The inclusion of processes and interactions between the main system components and the adoption of the modular format to the model provides a useful tool for exploring the effects of different management scenarios on the ecological performance, productivity, and environmental effects of different types of integrated aquaculture/agriculture systems.

Acknowledgements

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Table 2
Parameter calibration values for the aquaculture module obtained using fish production and pond data Rwanda (PD/A CRSP Workplan 4, Experiment 3) as model input

<table>
<thead>
<tr>
<th>Parameter/constant</th>
<th>Calibrated</th>
<th>Literature range</th>
<th>System</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>O2 consumed per oxidized 3 g g^-1 organic matter</td>
<td>1.08–3.00 g g^-1</td>
<td>marine sediments; fishponds</td>
<td>Jorgensen, 1976; Schroeder, 1978</td>
<td></td>
</tr>
<tr>
<td>Ks for phytoplankton uptake rate</td>
<td>1–6 mg C l^-1</td>
<td>fish ponds; laboratory</td>
<td>Nath, 1996; Perschbacher and Lorio, 1993; Northcott et al., 1991</td>
<td></td>
</tr>
<tr>
<td>Denitrification rate k</td>
<td>10 d^-1</td>
<td>fish tanks/ponds; marine sediments</td>
<td>Neori, 1996; Acosta-Nassar et al., 1994; Vanderborght et al., 1977</td>
<td></td>
</tr>
<tr>
<td>TAN diffusion rate k</td>
<td>5.0E^-10 m^2 d^-1</td>
<td>fish ponds</td>
<td>Hargreaves, 1995; Schroeder, 1978; Acosta-Nassar et al., 1994</td>
<td></td>
</tr>
<tr>
<td>NO3 diffusion rate k</td>
<td>0.00015 m^-2 d^-2</td>
<td>marine sediments; fish ponds</td>
<td>Acosta-Nassar et al., 1994</td>
<td></td>
</tr>
<tr>
<td>Ks for nitrogen uptake</td>
<td>0.00015–0.00062 m^-2 d^-1</td>
<td>laboratory systems</td>
<td>Prats and Llavador, 1994</td>
<td></td>
</tr>
<tr>
<td>TAN partition coefficient</td>
<td>10 g g^-1</td>
<td>fish ponds; agricultural soils</td>
<td>Hargreaves, 1995; Mehrani and Tanji, 1974</td>
<td></td>
</tr>
<tr>
<td>Water column respiration 1 d^-1</td>
<td>0.03–2.57 g O2 m^-3 d^-1</td>
<td>fish ponds</td>
<td>Teichert-Coddington and Carlos, 1993</td>
<td></td>
</tr>
<tr>
<td>Sediment respiration rate k</td>
<td>0.5 d^-1</td>
<td>freshwater sediments Walker and Snodgrass, 1986</td>
<td></td>
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</tr>
<tr>
<td>Lower sediment N mineralization rate k</td>
<td>0.0001–0.05 d^-1</td>
<td>lake sediments; fish ponds</td>
<td>Koecha et al., 1994</td>
<td></td>
</tr>
<tr>
<td>Lower sediment N mineralization rate k</td>
<td>0.00001–0.003 d^-1</td>
<td>lake sediments; wetlands</td>
<td>Reddy and Patrick, 1984</td>
<td></td>
</tr>
<tr>
<td>Phytoplankton respiration 0.005 d^-1</td>
<td>0.005–0.05 d^-1</td>
<td>laboratory</td>
<td>Lee et al., 1991; Prats and Llavador, 1994</td>
<td></td>
</tr>
<tr>
<td>TAN adsorption capacity</td>
<td>0.00012 g g^-1</td>
<td>pond sediments</td>
<td>PD/A CRSP data</td>
<td></td>
</tr>
<tr>
<td>Ks for detritus uptake</td>
<td>1.4 d^-1</td>
<td>fish ponds</td>
<td>Nath, 1996</td>
<td></td>
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<tr>
<td>Non-algal light extinction 8.42 m^-1</td>
<td>40– 80 gC l^-1</td>
<td>fish ponds</td>
<td>PD/A CRSP data</td>
<td></td>
</tr>
<tr>
<td>Chlorophyll a per phyto-</td>
<td>0.005–0.025 g g^-1</td>
<td>fish ponds</td>
<td>Lee et al., 1991</td>
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<tr>
<td>plankton biomass ratio</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Phytoplankton death rate k</td>
<td>0.01–0.09 d^-1</td>
<td>freshwater sediments</td>
<td>Hagiwara and Mitsch, 1994</td>
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<tr>
<td>Anaerobic layer mineralization rate k</td>
<td>0.0001–0.0008 d^-1</td>
<td>freshwater</td>
<td>Jorgensen, 1979</td>
<td></td>
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<tr>
<td>Sediment oxygen demand k</td>
<td>1.4 d^-1</td>
<td>freshwater sediments</td>
<td>Walker and Snodgrass, 1986</td>
<td></td>
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<tr>
<td>Phytoplankton digestibility</td>
<td>0.72 d^-1</td>
<td>freshwater</td>
<td>Popma, 1982; Boyd, 1995</td>
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<tr>
<td>Phytoplankton respiration 0.05 d^-1</td>
<td>0.005–0.05 d^-1</td>
<td>freshwater</td>
<td>Lee et al., 1991; DiToro et al., 1971</td>
<td></td>
</tr>
<tr>
<td>Nitrification rate k</td>
<td>0.05 d^-1</td>
<td>—</td>
<td>Prats and Llavador, 1991</td>
<td></td>
</tr>
</tbody>
</table>

References


