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Mangroves as filters of shrimp pond effluent: predictions and biogeochemical research needs

A. I. Robertson[1] & M. J. Phillips[2]
[1] *Australian Institute of Marine Science, PMB No 3, Townsville MC Qld 4810, Australia*
[2] *Division of Food and Agricultural Engineering, Asian Institute of Technology, GPO Box 2754, Bangkok 10501, Thailand*

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Abstract

Preliminary estimates of the ratio of mangrove forest: shrimp pond area necessary to remove nutrients from shrimp pond effluent are made using budgets of nitrogen and phosphorus output for semi-intensive and intensive shrimp ponds combined with estimates of total net primary production in *Rhizophora*-dominated mangrove forests in tropical coastal areas. If effluent is delivered directly to mangrove forest plots, it is estimated that, depending on shrimp pond management, between 2 and 22 hectares of forest are required to filter the nitrogen and phosphorus loads from effluent produced by a 1 hectare pond. While such ratios may apply to small scale, integrated shrimp aquaculture - mangrove forestry farming systems, the variability in mangrove hydrodynamics makes it difficult to apply such ratios at a regional scale. Before mangroves can be used to strip shrimp pond effluent more research is required on the effects that high ammonia and particulate organic matter loads in pond effluent have on nutrient transformations in mangrove sediments and on forest growth.

Introduction

Mangrove systems trap, transform and export materials. Whether mangroves are sources or sinks of materials depends on several factors, induding the element or substance under consideration, soil and vegetation types, geomorphic setting, hydrodynamics and the time frame of observation (Woodroffe, 1992; Wolanski *et al.*, 1992: Alongi *et al.*, 1992; Boto, 1992). Mangroves form an important part of the 'aquatic continuum' (Billen *et al.*, 1991) from river catchments to the sea, acting as filters for land-derived materials before they enter the ocean (e.g. Nedwell, 1974, 1975; Clough *et al.*, 1983), but it is difficult to generalize about the assimilative capacity of particular mangroves with respect to added nutrients.

Mangrove forests and their waterways have long been used as convenient sites for the disposal of sewage and wastewater in tropical countries. There have been several past attempts to evaluate the ability of mangroves to assimilate nutrients derived from sewage (Nedwell, 1975: Clough *et al.*, 1983; Boto, 1992), but recent losses of mangroves in many developing countries necessitates an assessment of an emerging serious threat to tropical mangroves - shrimp aquaculture. There has been a rapid expansion of marine and brackish water shrimp aquaculture in many tropical countries, particularly in South East Asia (Chua & Tech, 1990). Shrimp aquaculture presents an interesting challenge to coastal managers and scientists because most shrimp ponds are constructed in or adjacent to mangrove systems; large areas of mangroves have been lost to this form of development (Hatcher *et al.*, 1989; Barg, 1992). The more intensive shrimp aquaculture practices produce large quantities of effluent with high particle loads and ammonia concentrations (Lin, 1993), which in many cases pollutes the water used for future shrimp production. There are many problems associated with using mangrove areas for shrimp ponds, including direct loss of mangrove forests with subsequent loss of dependent capture fisheries, and development of acid-sulphate soils (e.g. Ong, 1982; Hong & San, 1993). In this paper, we ask whether retention of existing mangrove forests around sites of shrimp aquaculture production, or planting of new mangroves in mixed shrimp-forest farming models, can ameliorate the impacts of effluent derived from intensively managed shrimp ponds.

First, we provide a brief review of the range of shrimp farming practices currently used in coastal region of southeast Asia, particularly regarding potential effluent loads. Secondly, we review briefly some recent advances in mangrove biogeochemistry that are relevant to the issue of mangroves as effluent filters. Finally some predictions are made about the area of mangroves required to remove nutrients from the effluent produced by different shrimp pond farming practices. Acknowledging that many factors influence the nutrient-stripping capabilities of mangrove systems, we suggest research that is required to help aid management decisions about the balance between mangroves and shrimp ponds in tropical coastal regions.

Table 1. Nutrient and particle concentrations and bacterial cell densities and production in effluent water from intensively-managed shrimp ponds. For comparison, the range of values in pristine tidal and estuarine mangrove waters are also provided. Data from Kanit & Putha (1992), Lin (1993), Alongi *et al.* (1992), Robertson *et al.* (1993) and unpubl. data.

Variate	Intensive shrimp-pond effluent water	Pristine mangrove waterways
Salinity (‰)	10.0–35.0	0–37
NH_3 (μM)	1.97–73.15	0.10–1.42
NO_3 (μM)	0.05–1.54	0–11.75
PO_4 (μM)	0.53–4.21	0–5.26
TSS (mg l^{-1})	119–225	67–3312
Chl*a* (μg l^{-1})	20–250	0.2–5.07
Bacterial numbers (cells ml^{-1})	8.8–25.7$\times 10^6$*	0.85– 4.70$\times 10^6$
Bacterial production (μg C l^{-1} d^{-1})	39–87*	0–18

* Based on semi-intensive ponds (Moriarty, 1986)

Shrimp aquaculture systems and effluent production

Shrimp aquaculture practices vary from traditional extensive culture systems practiced for centuries (Kungvankij, 1985) to the super-intensive culturing systems currently being used in Thailand. In traditional extensive systems, relatively large ponds (up to 50 ha, but usually less than 5 ha) are constructed in intertidal (usually mangrove) regions. Wild shrimp fry enter the ponds either during tidal water exchange, or are intentionally gathered from the wild and stocked directly in to ponds. Growth of individual shrimp in these systems relies on natural pond fertility, since fertilizers and feeds are not generally used. Retention time of water in the ponds varies, but may be up to two months. Yields of shrimp from such systems are low, in the range 100-300 kg.ha^{-1}.y^{-1} (Kungvankij, 1985). Since there is no supplementary feeding in these systems, their contribution to total nutrient loading in coastal waters is low. For instance, maximum concentrations of ammonia, nitrate and phosphate of 6.2, 1.0 and 1.8 μM recorded in extensive shrimp ponds in Dat Mui District of southern Vietnam (Hong & An, 1992) were not greatly different to those in adjacent mangrove creeks, or other pristine mangrove systems (see Table 1).

During the past two decades, improvements in extensive shrimp culture systems have led to increased shrimp production. Among improvements are liming of ponds prior to stocking to condition the soil, application of organic or inorganic fertilizers to enhance natural food production, increased stocking densities due to pumping, and removal of pests and predators through chemical techniques.

Semi-intensive pond systems have achieved higher yields of shrimp (14 tonnes ha^{-1} y^{-1}) through better control of pond size and construction to facilitate water exchange, higher stocking rates (often using hatchery reared post-larvae, particularly *Penaeus monodon*), application of fertilizers and regular supplementary feeding (often trash fish) and water exchange in ponds. Stocking density of postlarval and juvenile shrimp in semi-intensive systems range from 2–5 per m^2.

In the last decade there has been a rapid shift to intensive culture systems in many parts of southeast Asia (e.g. Chua & Tech, 1990). These systems combine high initial stocking densities (>30 per m^2) and feeding rates with pond management techniques that aim to maintain a water-column algal bloom beneficial to the behaviour and growth of *Penaeus monodon*, the preferred culture species. Depending on local conditions, feeding with pelletized food takes place at least once per day and there is regular water exchange (up to 30% of pond volume per day) near the end of the grow-out period. Yields from such systems are regularly 10 tonnes ha^{-1} y^{-1} (Lin, 1993).

Recently, 'super'-intensive culture systems have been used in Thailand. Mainly in response to very poor water quality in intake waters for ponds, and subsequent disease risks, these systems do not rely on high rates of water turnover. Rather, by treatments of the ponds with lime before and during the grow-out period, the pH of the pond water is maintained at a constant level. Together with fertilization, this procedure helps to maintain a constant water-column algal bloom in the ponds. Post-larval shrimp are stocked at densities >70 per m^2 and pelletized food is fed to prawns several times per day. Careful placement and constant use of mechanical stirrers ensures that waste products accumulate in the centres of ponds, and that aerobic conditions are maintained in a region bordering the pond walls where shrimp feed. Claims of yields of 20 t ha^{-1} y^{-1} have been made for such systems.

As the level of intensity in shrimp farming systems increases, so do the concentrations of nutrients in effluent water and the total load of nutrients entering coastal waterways (Table 1). The largest increases are in the concentrations of ammonia (by 1 to 2 orders of magnitude) and in the particulate loads in the forrn of chlorophyll *a* and bacterial cell densities (Table 1). This is not surprising, considering that algal blooms are maintained within the ponds. With the high levels of nutrients available from the breakdown of added food pellets and fertilizer, these blooms are not nutrient limited, and the decomposition of organic matter and the excretion by shrimp and other organisms in the pond maintain the high ammonia concentrations. Much of the food added to intensive ponds supports the growth of water-column bacteria; bacterial numbers and production are an order of magnitude greater than in mangrove waterways (Table 1).

Not all excess nutrients in semi-intensive and intensive shrimp ponds are in the dissolved or suspended particulate phases. Most accumulate in pond sediments (see Table 4). The high organic content of pond sediments supports high bacterial cell densities and production (Moriarty, 1986). There are no published records of the nutrient concentrations in pond sediments at harvest-time in shrimp ponds, but our budgeting exercises (see later) indicate that sediments must be a major site of nutrient accumulation.

The usual management practice following harvest in semi-intensive and intensive shrimp ponds is to remove the organic-rich sediments prior to initiation of the next production cycle. Pond sediments are either stored in settlement area, spread on supra-tidal waste dumps or placed in adjacent waterways.

Biogeochemistry of mangrove systems: potential for nutrient filtering

There have been two recent reviews of nutrient cycles in mangrove-dominated systems (Alongi *et al.*, 1992; Boto, 1992) and we wish to highlight some of the most recent research findings that are relevant to the preliminary models of the nutrient stripping capabilities of mangrove systems which we construct later in the paper.

Nutrient requirements for growth of Rhizophora *forests*

There have been previous estimates of the nutrient requirements to support primary production in *Rhizophora*-dominated forests (e.g. Clough *et al.*, 1983; Boto & Wellington, 1988). As Clough (1992) has pointed out, no study has yet measured total net primary production in a natural or managed mangrove forest. While estimates of total net above-ground production (litter fall + wood accumulation) are available for some mangrove forests, there are no published estimates of root production, or estimates of release of dissolved organic material from living mangrove roots.

Estimates of wood accumulation in *Rhizophora* forests vary from 6–45 t ha^{-1} y^{-1}, and average 20 t ha^{-1} y^{-1} (see review by Clough, 1992). Litter fall in the same forests averages 8 t ha^{-1} y^{-1} (Table 2).

Root biomass is very high in mangrove forests (Komiyama *et al.*, 1987; Gong & Ong, 1990) when compared to terrestrial forests (Vogt *et al.*, 1986). A recent study of fine root turnover rates in *Rhizophora* spp. forests of northeastern Australia (Robertson and Dixon, 1993 and manuscript in preparation) has shown that fine root production is equivalent to the total above-ground net production. There may therefore be 20+8 = 28 t ha^{-1} y^{-1} of fine root production in typical *Rhizophora* forests (Table 2). By converting dry mass production to nitrogen (N) and phosphorus (P) equivalents (from Clough *et al.*, 1983; Spain & Holt, 1980; Gong & Ong, 1990), it is possible to estimate N and P requirements to support total net forest production of 219 kg N ha^{-1} y^{-1} and 20 kg P ha^{-1} y^{-1}, respectively (Table 2).

These estimates are likely to be very conservative if used to calculate the ratio of forest:shrimp pond necessary to strip nutrients from aquaculture effluent because they are based on an average of natural and managed forests. Plantations of *Rhizophora* can be managed to

Table 2. Average values for annual production of different components of humid, tropical *Rhizophora*-dominated forests in terms of dry mass, nitrogen and phosphorus (for sources, see text).

Component of production	Dry mass Production ($t\ ha^{-1}\ y^{-1}$)	Tissue Nutrients N (%)	P (%)	Nutrients required for primary production ($kg\ N\ ha^{-1}\ y^{-1}$)	($kg\ P\ ha^{-1}\ y^{-1}$)
Litterfall	8	1.00	0.10	80	8
Wood accumulation	20	0.12	0.03	24	6
Root production	28	0.41	0.02	115	6
TOTALS	56			219	20

increase tree growth rates (Haron, 1981), and mixed shrimp pond plantation farming system (e.g. Hong & San, 1993) may have greater production potentials than presently realised. In addition, in response to fertilization, *Rhizophora* trees not only increase their growth rates, but also exhibit increased nutrient concentrations in their tissues, thus enhancing nutrient removal from the environment (Clough *et al.*, 1983).

On the other hand, canopy production is lost from trees as litter fall, and litter may or may not be retained within forests, depending on the degree of tidal flushing and leaf burial by crabs (Robertson *et al.*, 1992). Inclusion of litter fall in total net production estimates may overestimate the nutrient stripping capabilities of *Rhizophora* forests.

Finally, it is worth noting that mangrove forests can only be used as nutrient filters if mass accumulated during growth is harvested (for timber, firewood or charcoal), or at least retained or recycled within mangrove sediments (as is the case with fine root production).

Sediment bacteria

Bacteria in mangrove sediments are more productive than in temperate marine sediments, and they play an essential role as sinks for dissolved and particulate nutrients (Alongi, 1988a; Alongi, 1993a). Bacterial productivities ranging from 0.2 to 5.1, averaging 1.5 g $C\ m^{-2}\ d^{-1}$, have been recorded in *Rhizophora* forests in tropical Australia (Alongi, 1988a) and on mudflats adjacent to mangrove forests (Moriarty, 1986; Alongi, 1988b, 1991). These highly productive populations are able to sequester much of the dissolved nutrients available in sediment porewaters and in overlying waters. Flux chamber experiments conducted by Stanley *et al.* (1987) and Boto *et al.* (1989) in northern Australia, revealed negligible DON (as amino acids) and DOC flux in or out of the sediment, despite very large concentration differences between porewaters (high levels) and overlying waters (low levels). However, when sediments were poisoned to kill bacteria, 27–69 mg $N\ m^{-2}\ d^{-1}$ and 0.4–2.4 g $C\ m^{-2}\ d^{-1}$ was liberated from sediments to the water. It was estimated that such fluxes support between 9 and 38% of the nitrogen and carbon requirements for sediment bacterial production. Alongi (1991) performed similar flux studies on bare mud banks adjacent to mangrove forests in the Fly Delta of Papua New Guinea. He found that in all cases there were fluxes of nutrients (DOC, dissolved inorganic nitrogen and phosphorus, dissolved organic nitrogen and phosphorus) into the sediments from overlying waters, indicating that sediments and their microbiota are sinks for dissolved material.

Work in mangrove forests and adjacent mudflats shows a significant positive correlation between sediment bacterial growth rates and dissolved organic phosphorus in porewaters and total phosphorus (Alongi, 1991; Alongi, 1993b). However, the difficulty of conducting nutrient enrichment experiments with intertidal sedimentary bacteria (Alongi, 1993a) precludes definitive tests of nutrient limitation of bacterial growth, although this is likely.

Sewage or aquaculture effluent added to mangrove areas may be sequestered and processed by sediment bacteria (Nedwell, 1974; Moriarty, 1986; Alongi, 1993a). However, processing efficiency tends to decrease with increasing input. High organic loadings usually shift balanced aerobic-anaerobic systems to complete anaerobiosis. Anaerobic systems are less efficient in nutrient recycling. Little work has been done on anaerobic bacterial processes in mangrove sediments. Rates of sulphate reduction in tropical mangroves are similar to those in temperate intertidal sediments, but lower than in saltmarsh soils (e.g. Kristensen *et al.*, 1991), owing to moderate redox levels and high rates of aeration of soil by burrowing crabs. However, rapid turnover of the NH_4 pool in some mangrove sediments indicates subsurface activity of other (non sulphate-reducing) anaerobes (Blackburn *et al.*, 1987).

Few estimates of denitrification are available for mangrove sediments. Nitrate discharged from a sewage treatment plant in Fiji mangroves, and not taken up by phytoplankton, decreased by 30% by the time it passed through the estuary, with estimated reduction rates of 26.2 - 87.6 mg N m^{-2} d^{-1} (Nedwell, 1975). By comparison, in an unpolluted mangrove system in tropical Australia, the mean rate of denitrification was 0.18 mg N m^{-2} d^{-1} (Alongi *et al.*, 1992).

Mangrove sediment chemistry

When soluble reactive phosphorus is added to soils, it is usually rapidly immobilized by adsorption reactions depending to a large extent on the soil clay mineralogy, iron content and redox status (Holford & Patrick, 1979). Additions of 400 kg P ha^{-1} (in the form of superphosphate) over 1 year to mangrove sediments in northern Australia increased the concentration of plant-available phosphorus by a factor of 10 (Boto & Wellington, 1988). However, highly reduced mangrove soils have far less capacity to absorb phosphorus, owing to the greater solubility of iron sesquioxides (Boto, 1992). Adsorption maxima for mangrove soils may range from 250–700 μg P g^{-1} dry weight (Clough *et al.*, 1983; Boto, 1992) and are lower in soils which have naturally high P concentrations, or which have received high nutrient effluent for years.

Total nitrogen levels in pristine mangrove sediments range from 600–2000 μg N g^{-1} dry weight, mainly in the form of organic nitrogen. Ammonium is the main forrn of inorganic nitrogen in the mostly anaerobic mangrove soils. The availability of cation exchange sites in mangrove soils is often low because of the generally high levels of sodium in sediments. The result is that most ammonium occurs in the porewater dissolved phase in concentrations ranging from 0–760 μM (Clough *et al.*, 1983; Alongi *et al.*, 1992). The loss of porewater ammonium from mangrove soils depends on the degree of soil permeability (low loss with heavy clays), the degree of tidal flushing (higher losses with higher flushing), the degree of denitrification occurring in the sediments and the rate of uptake by plants (Clough *et al.*, 1983; Alongi *et al.*, 1992). Boto (1992) queried whether mangrove sediments could be viewed as long term sinks for nitrogen added from nutrient pollution, if most effluent was delivered in the form of ammonia rather than nitrate. In the case of high nitrate concentrations, Nedwell (1975) has argued that denitrification could significantly reduce the nitrate levels in treated sewage effluent applied to mangrove soils.

Effluent loads from coastal shrimp ponds

Enough data are available to prepare some preliminary budgets for nutrient loads in the effluent derived from different types of shrimp ponds (Tables 3 and 4). Because there are no data available for DOC in shrimp pond effluent we have restricted our budgeting exercise to nitrogen (N) and phosphorus (P).

Based on recent interviews with farmers and managers in Tra Vinh Province of southern Vietnam, we were able to construct approximate N and P budgets for a semi-intensive shrimp pond managed in the way detailed in Table 3. Unfortunately, no data were available on the nutrient concentration in pond inflow or effluent water. For the purpose of this exercise we assumed that N and P concentrations in intake water were similar to those for dissolved inorganic materials measured in creeks and improved-extensive culture systems of nearby Minh Hai Province. Hong & An (1992) reported concentrations of 0.174 mg N l^{-1} and 0.054 mg P l^{-1} for creek water, with pond concentrations being approximately twice those in creek water. These are likely to be underestimates for semi-intensive ponds, but are the best figures available at present. The budget (Table 3) shows that the major input of N and P to the ponds is via trash fish used as shrimp feed. There appears to be a food conversion ratio (food input/shrimp yield) of 10. The harvest of shrimp is a minor part of the N and P budgets (Table 3). The total annual effluent load of N from a 1 ha semi-intensive pond is estimated to be 565 kgN, of which dissolved inorganic N in pond discharge water is estimated to be 23% of the total (Table 3). The remaining N would be in the particulate form, and presumably in the pond sediments that would be removed at the end of each culture period (i.e. 0.5 y). Total annual P load in effluent is estimated to be 70 kgP, of which 57% is estimated to be in the dissolved inorganic form in discharge waters.

Our budgets for intensive shrimp ponds in north-eastern Thailand are likely to be more reliable, as they are based on a 5 month detailed study of pond inputs

Table 3. Nitrogen and phosphorus budgets for semi-intensive shrimp (*Penaeus merguiensis* and *P. monodon*) ponds. Based on information collected from 3 sources in Tra Vinh Province, Vietnam, August 1993 (A. Robertson and M. Phillips, unpubl. data). Remainder component of outputs is the difference between total inputs and (shrimp harvest + discharge water) outputs.

Pond management		
Stocking density	4 juv m^{-2} (2 g FW m^{-2})	
Culture period	0.5 y	
Size of pond	1.0 ha	
Depth of pond	1.0 m	
Feed input (trash fish)	10,000 kg FW ha^{-1} y^{-1}	
Yield of shrimp	1,000 kg FW ha^{-1} y^{-1}	
Water turnover	10% per day	
Fertilizer	No	
Nutrient concentrations		
N content of shrimp	2.94% FW	
P content of shrimp	0.19% FW	
N content of feed (trash fish)	5% FW	
P content of feed (trash fish)	0.5% FW	
Budgets (for 1 ha pond)		
	N (kg ha^{-1} y^{-1})	P (kg ha^{-1} y^{-1})
Inputs		
Feed	500	50
Pond inflow	64	20
Shrimp juveniles	1	trace
Total input	565	70
Outputs		
Shrimp harvest	29	2
Pond discharge water	128	40
Remainder (sediment)	408	28
Total output	565	70

and outputs (Table 4). The inputs of N and P to these ponds via feed pellets are 3 and 8 times higher than inputs of N and P to semi-intensive ponds from shrimp feed. Total annual effluent output from 1 ha of intensive ponds is 1570 kgN and 433 kgP, most of which (87% for N, 91% for P) is in the form of pond sediments at the end of the harvest periods (Table 4). Total effluent loads produced by intensive shrimp ponds are 3 and 8 times greater than semi-intensive ponds for N and P, respectively (Tables 3 and 4).

Area of mangrove needed to remove N and P from effluent

Using estimates of the N and P required for primary production in an average, humid tropical *Rhizophora* forest (Table 2) as a guide, we have attempted to provide a rough 'rule-of-thumb' for the area of mangrove forest that would be necessary to remove N and P from the effluent of shrimp ponds (Table 5). For semi-intensive ponds 2 to 3 ha of forest would be required for each 1 ha pond while, for intensive ponds, up to 22 ha of *Rhizophora* forest might be needed.

Table 4. Nitrogen and phosphorus budgets for intensive shrimp (*Penaeus mondon*) ponds. Based on a study of five ponds in Chantaburi Province, Thailand, January May 1993 (M. Phillips, unpubl. data). Remainder component of outputs is the difference between total inputs and (shrimp harvest + discharge water) outputs.

Pond management		
Stocking density	52 post larvae m^{-2}	
Culture period	122 d	
Size of pond	0.4 ha	
Yield of shrimp	13.79 t FW ha^{-1} y^{-1}	
Feed input (pellets)	26.42 t FW ha^{-1} y^{-1}	
Fertilizer	Yes (see below)	
Nutrient concentrations		
N content of shrimp	2.94% FW	
P content of shrimp	0.19% FW	
N content feed	7.07% FW	
P content feed	1.64% FW	
Pond Budgets (normalized to 1.0 ha pond size)		
	N (kg ha^{-1} y^{-1})	P (kg ha^{-1} y^{-1})
Inputs		
Feed	1868	433
Inflow water	43	8
Fertilizer	62	17
'Tea seed cake'	1	1
Shrimp stock	1	trace
Total input	1975	459
Outputs		
Shrimp harvest	405	26
Pond discharge water	199	39
Remainder (sediment)	1371	394
Total output	1975	459

Assumptions and implications for management and research

To make these first-order estimates we assumed that 1) effluent is delivered directly and evenly dispersed in the mangrove forest; 2) all N and P is available for plant uptake; 3) plant uptake is the only sink for effluent N and P, and; 4) the average values for *Rhizophora* forest production are appropriate for a general model. Of these assumptions, the last is the least critical. High variation in mangrove primary production is caused mainly by the degree of aridity (which controls soil wetness and salinity) and air temperature (Clough, 1992). Shrimp aquaculture is centred on the humid tropics (Chua & Tech, 1990) and our estimates of average *Rhizophora* production come from such regions.

Not all N and P in pond effluent will be available for primary production. Depending on the nature of the receiving sediments, a proportion of effluent P would be immobilized as iron, calcium or aluminium phosphates (see earlier). Likely, some P would already be adsorbed on sediments on pond bottoms, before removal. If some effluent P becomes bound in the sediments, our estimates of forest area required for nutrient filtering are overestimates. The most serious (perhaps simplistic) assumption concerns delivery of effluent to mangrove forests. As Clough *et al.* (1983) and Boto (1992) have pointed out, delivery of effluent to mangrove forests, rather than mangrove creeks appears to be the most preferable option owing to the

Table 5. First-order estimates of the ratio of *Rhizophora* mangrove forest area to shrimp pond area that would be required to remove nitrogen and phosphorus loads produced during the operation of semi-intensive and intensive shrimp pond operation. Based on figures in Tables 2-4. See text for assumptions and drawbacks with these estimates.

	Pond management type	
Element in effluent	Semi-intensive	Intensive
Nitrogen	2.4:1	7.2:1
Phosphorus	2.8:1	21.7:1

influence of high nutrient loads on aquatic food chains. In the integrated shrimp-forest farming models currently used in some parts of South East Asia (e.g. Hong & San, 1993), with their mixture of *Rhizophora* forest and semi-intensive shrimp culture, it may be possible to engineer delivery of effluent water and sediment from ponds to adjacent forests. Finally, the potential for high rates of denitrification in mangrove sediments (see earlier) imply that our estimates of removal of nitrogen from effluent by mangrove forests is likely an underestimate.

Can our rule-of-thumb on forest area:pond area (Table 5) be scaled up to the regional level (i.e. areas of 10^2 km^2) to provide guidelines for coastal managers who seek the correct balance of shrimp pond and mangrove forest areas that will allow sustainable shrimp farming? A major source of variability in estimating appropriate ratios of forest to pond area on larger scales will be the residence time of water in mangrove waterways, particularly if effluent is delivered directly to mangrove creeks, as is the case in many semi-intensive and intensive shrimp farming operations.

The residence time of water in mangrove systems is controlled by a number of synergistic factors (Fig. 1). Water will have long residence times in the upper reaches of intricately branched creek systems in very flat, wide mangrove forests, made longer by the dense root systems of *Rhizophora* forests (Fig. 1). In the dry season in northern Australia, water may be trapped for days to weeks in such tidal mangrove creeks (Wolanski & Ridd, 1986). Even in the wet season, when estuaries are flushed, the residence time in secondary creeks may be measured in days (Wolanski & Ridd, 1986). The area of forest required to strip nutrients from shrimp pond effluent will therefore increase with increasing complexity of mangrove waterways, since large loads of effluent may accumulate in these regions. Conversely, in short, unbranched creeks with a relatively narrow band of mangroves, residence time of water will be short, and effluent may be delivered rapidly to the coastal oceans if this is desirable. However, depending on coastal topography, material delivered from mangrove waterways may be trapped in embayments by coastal boundary layers (Wolanski & Ridd, 1990) and thus pollute inshore regions.

In our calculations, we have also assumed that there are no other sources of N and P loading other than shrimp pond effluent. Clearly, this is not usually the case in the coastal regions of south east Asia, where a cocktail of agricultural, industrial and urban waste is often delivered to nearshore habitats (e.g. Chua *et al.*, 1989).

We have based our arguments on budgets of N and P, but have not considered the organic carbon load in shrimp pond effluent. As far as we know there are no data on DOC concentrations in shrimp pond effluent. POC concentrations are likely to be high, given the high concentrations of chlorophyll a and bacteria in the effluent from intensive ponds (Table 1). This organic matter will have a low C:N ratio, and would therefore be likely to support rapid rates of bacterial turnover in mangrove sediments and waterways. This is turn may have major implications for oxygen levels in mangrove sediments and water, which are characterized by naturally low oxygen concentrations (Boto, 1992). In Table 6, we have summarized some of the questions that require further research if mangroves are to be used in stripping nutrients from effluent. As with an earlier review of mangroves and sewage, (Clough *et al.*, 1983), we stress the need for monitoring of existing shrimp pond effluent systems and the short- and long-term impacts they have had on sediments, fauna and forest production.

Conclusions

If the effluent from shrimp ponds could be delivered directly to mangrove forests, such as in small-scale, integrated shrimp-forest farming systems, we estimate that somewhere between 2 to 22 hectares of forest would be required to filter the nitrogen and phosphorus from pond waste, depending on the pond management regime. However, the probable impact of high concentrations of ammonia and particulate organic matter on mangrove sediments and plant life requires further

Table 6. Research requirements for sustainable management of mangroves to strip nutrients from shrimp pond effluent. We have separated mangrove plantations managed purely for timber or firewood production in small scale farming systems (= small scale plantations) and forests, natural or otherwise, which support a variety of system functions.

Type of mangrove system	Research questions
Small scale plantations	• what loads of ammonia and particulate and dissolved organic matter (DOM and POM) cause complete anaerobiosis of mangrove sediments, and tree mortality? • what is the long term 'assimilative capacity' of sediments for phosphorus? • how do different sediment types influence the retention and transformation of effluent nitrogen? • what silviculture practices will maximize forest yields, and nutrient uptake rates?
Forests (on a regional scale)	• what loadings of ammonia, DOM and POM have significant negative effects on benthic micro-, meio- and macrofauna? • what are the indirect effects of the loss of burrowing fauna on system functions? • what combinations of effluent loads and hydrodynamic regimes cause anoxic conditions in mangrove waterways?

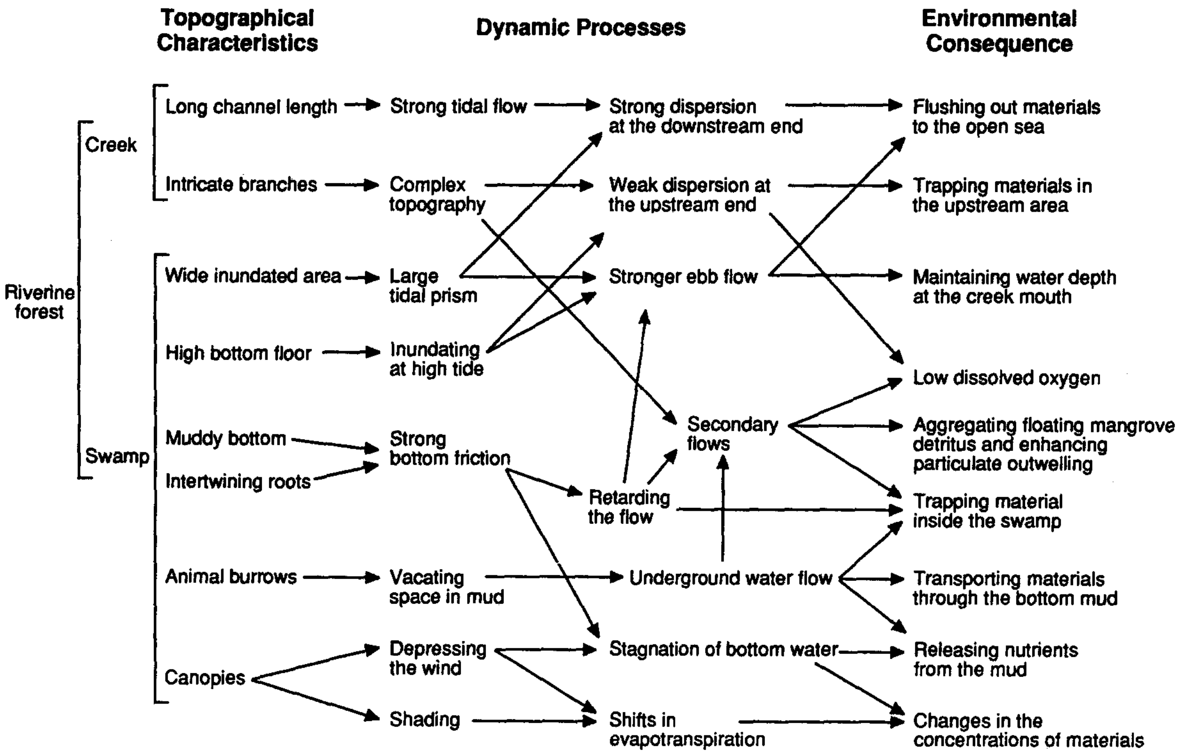

Fig. 1. How topographical features control physical processes and the trapping of materials in mangrove ecosystems (waterways plus forested swamp areas). Based on Wolanski *et al.* (1992).

research before it is known whether such practices are sustainable (Table 6).

Whether the ratios of forest area: shrimp pond area can be used on a more regional scale, to indicate the correct balance between areas devoted to shrimp aquaculture in mangrove ecosystems depends, to a large degree, on two factors: the hydrodynamics in mangrove waterways and the nutrient loading from other pollution sources. As with previous reviews of the role of mangroves in the tertiary treatment of sewage (Clough *et al.*, 1983; Boto, 1992), we believe that if mangroves are to be used to strip off aquaculture-derived nutrients, where possible, effluent should be delivered to higher intertidal mangrove areas, rather than directly to mangrove waterways. This would minimize the effects on food chains and fisheries in tidal waters.

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