A design and management approach for horizontally integrated aquaculture systems

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Abstract

This thesis presents an assessment concerning the potential of horizontally integrated aquaculture, with outcomes assessed from a systems-based perspective. A literature review concerning the negative impacts and the limitations of current wastewater management approaches demonstrated that improved strategies are required. Horizontally integrated aquaculture was proposed, where the productive reuse of aquaculture wastewater ameliorates associated negative impacts. A definition for horizontally integrated aquaculture is presented and management strategies that conform to this definition reviewed.

The development and application of the ADEPT bioeconomic model to assess the potential of a constructed wetland and trout fishery to treat wastewater from a commercial smolt unit in Scotland is described. The model outputs were tested against observations from commercial facilities operating under comparable conditions to those envisaged for horizontally integrated systems. Findings demonstrated that the modelling approach adopted was generally effective in predicting the composition of wastewater outputs from the farm and the effect of the selected treatment strategies. The model was applied to two further case studies. One assessed the potential of treating wastewater from shrimp farms in Thailand using a constructed mangrove wetland; the second evaluated the possible advantages of a rational design approach to lagoon-based wastewater treatment and reuse, as opposed to a conventional design and traditional practices developed in peri-urban Calcutta.

Chapter One

Introduction

Overview

Rapid expansion of the aquaculture sector in many countries has often outpaced the introduction of legislation and monitoring programmes to regulate wastewater discharges, and such a pattern of development has frequently been linked with negative environmental impacts. When regulatory controls are implemented, producers may consider them rigid and draconian, whilst environmental activists and advocates of the precautionary principle may find them lenient and poorly policed. Practical aspects of managing and treating aquaculture wastewater make compliance with discharge standards problematic, time consuming and expensive, whilst the prospect of more stringent standards threatens to exceed developments in waste management and conventional treatment approaches. Therefore, a proactive strategy for wastewater management, in both emerging aquaculture sectors and established industries, that exploits the waste resource in complementary production systems, whilst ameliorating negative environmental, economic and social impacts is desirable. This thesis proposes that the strategy of horizontally integrated aquaculture embodies this concept. By invoking a systems-based approach, the potential of this strategy to ameliorate negative impacts of aquaculture wastewater and produce products with value, whilst allaying the concerns of stakeholders, especially consumers and regulators is assessed.

As a background to discussing horizontally integrated aquaculture, the range of negative environmental, economic and social impacts associated with aquaculture wastewater, together with the strategies that have been employed to mitigate these impacts are discussed in Chapter 1. In Chapter 2 a definition for horizontally integrated aquaculture is derived and studies of the systems meeting the proposed criteria reviewed. Chapter 3 outlines the formulation, development and application of a bioeconomic model for horizontally integrated aquaculture systems; a case study concerning the integration of a smolt unit, constructed wetland and trout fishery is used to evaluate the model. Chapters 4 and 5 present further case studies assessing an integrated shrimp farm-mangrove system and a rational design approach to wastewater aquaculture, respectively. Although not falling within the proposed definition of horizontally integrated aquaculture, the case-study concerning wastewater aquaculture in peri-urban Calcutta was selected as this system constitutes one of the few large-scale ecologically based wastewater production and reuse systems from which it was thought insights concerning the management demands, social and economic implications and real world constraints to such systems could be gained. It was also hoped to test the generality of the modelling approach adopted. A Delphi investigation concerning future prospects for horizontally integrated aquaculture is described in Chapter 6. Findings from the review process, modelling exercises and Delphi investigation are drawn together in Chapter 7; recommendations to guide the sustainable development of horizontally integrated aquaculture systems are proposed and constraints requiring further investigation identified.

1.1. Introduction

This chapter introduces the concept that aquaculture appropriates a range of environmental goods and services from an ecosystem support area, or ecological footprint, and that the demand for these goods and services can exceed the carrying capacity of the ecosystem. Environmental, social and economic impacts associated with aquaculture wastewater discharges that exceed this carrying capacity are reviewed. The current status of wastewater treatment and management is discussed and constraints to developing more efficient and reliable approaches identified.

1.2. Aquaculture and its ecological footprint

Aquaculture appropriates a range of environmental goods and services (Beveridge, Phillips and Macintosh, 1997). The area of natural environment required to sustain the supply of these goods and services has been referred to as both the *ecosystem support area* and the *ecological footprint* (Kautsky, Berg, Folke, Larsson and Troell, 1997). Appropriated environmental goods include the physical area of land or water for site development; construction materials e.g. timber, stone, soil; water (containing oxygen) for filling ponds and replenishing that lost through evaporation or exchange; broodstock, seed and juveniles for stocking; areas of land or water to produce feed. Environmental services required by aquaculture consist mainly of processes metabolising and assimilating waste fractions and replenishing dissolved oxygen concentrations.

Ecological footprints associated with intensive aquaculture indicate that environmental goods and services are appropriated from relatively large areas (Folke and Kautsky, 1989; Larsson, Folke and Kautsky, 1994; Robertson and Phillips, 1995; Berg, Michélsen, Troell, Folke and Kautsky, 1996; Kautsky et al., 1997; Folke, Kautsky, Berg, Jansson and Troell, 1998). Shrimp (*Penaeus stylirostris* and *Penaeus vannamei*) production in semi-intensive ponds along the Caribbean coast of Colombia was estimated to require an ecosystem support area 35-190 times the surface area of the farm (Larsson et al., 1994). The mangrove nursery area required to supply postlarvae to 1 ha of shrimp ponds was estimated to be the largest component of the ecological footprint at 160 ha. Describing small-scale semi-intensive pond farming of tilapia (*Tilapia rendalli*, *Oreochromis mossambicus* and *Oreochromis niloticus*) in Zimbabwe, Berg et al. (1996) estimated that the area for phosphorus assimilation and oxygen production to support 1 m^2

of this culture system was 0.9 and 0.5 m^2, respectively, and could therefore be accommodated within the pond area. By comparison, it was estimated that tilapia production in 1 m^2 of intensively managed cages situated in Lake Kariba required an ecosystem area of 115 and 160 m^2 for phosphorus assimilation and oxygen production, respectively. At a further extreme, the intensive production of salmon in cages is largely dependent upon the supply of concentrated feed containing fishmeal and oils derived from marine capture fisheries. Folke (1988) estimated that the ecosystem area needed to sustain the supporting capture fisheries ranged between 40,000 and 50,000 times the cage area.

A key factor to be considered when invoking the principle of ecological footprints is the degree of connectivity that exists between the activity appropriating environmental goods and services and the supporting ecosystem area. Traded goods and services e.g. feed and fry, derived from other ecosystems can be purchased to augment inadequate local supplies. Environmental goods and services that cannot be supplemented i.e. the assimilation of nutrients, will therefore define the capacity of the localised ecosystem to sustain production in the aquaculture facility. Balancing the demand for these goods and services with supply is essential to avoid undesirable environmental impacts.

The supposition that increasing material and energy inputs to aquaculture leads to increased risk from negative environmental impacts has been presented (Folke, 1988; Kautsky and Folke, 1990). However, pond-based integrated agriculture-aquaculture and dike-pond systems receive comparatively large inputs of organic material, yet have little impact on the external environment (Edwards, 1993). Inputs of duck, pig and cattle manure to traditional dike-pond farms in the Zhujiang Delta, China are typically 75, 454 and 550 t ha^{-1} y^{-1}, resulting in fish yields of 7-10 t ha^{-1} y^{-1} (Korn, 1996). However, integrated production of sugarcane, mulberry, silk worms, vegetables and bananas, fertilized with mud and water from the fishpond, may increase production to 20-40 t ha^{-1} y^{-1}. Physically, this integrated system is relatively closed, with little discharge of pond water or by-products to the external environment, instead relying on *in situ* processes to assimilate waste and supply other environmental services; therefore, the openness of aquaculture systems is important when considering possible environmental impacts.

Production in semi-intensive and intensive aquaculture requires that natural supplies of environmental goods, particularly feed, be supplemented. However, as production intensifies beyond the productive and assimilative capacity of the standing water body, the exchange of water and therefore the volume of wastewater discharged tends to increase (Phillips, Beveridge and Clark, 1991). This leads to the appropriation of increased levels of environmental goods and the export of waste to the receiving environment. Large wastewater volumes discharged from intensive aquaculture typically contain low concentrations of waste compounds as compared with effluents from other industrial processes (Cripps and Kelly, 1996), and in overall mass flow terms, nutrients discharged from aquaculture make a relatively small contribution to the overall anthropological input to most aquatic ecosystems (Ackefors and Enell, 1990; Kronvang, Ærtebjerg, Grant, Kristensen, Hovmand and Kirkegaard, 1993). However, aquaculture wastewater may contain significantly higher concentrations of pollutants than would be expected in the receiving environment (Robertson and Phillips, 1995).

To appreciate the likely impact of aquaculture wastewater on an ecosystem key factors must be considered: the connectivity between the aquaculture system and ecosystem; the wastewater volume and quality; the hydrology of the receiving environment and the carrying capacity of the ecosystem. Hydrological characteristics of the receiving environment largely dictate the carrying capacity of the ecosystem, with open systems situated in dynamic high-energy sites that promote mixing and dispersion being less likely to exceed the carrying capacity of the ecosystem. However, continuous point source discharge of dilute aquaculture wastewater to low energy systems over an extended period may exceed the assimilative capacity of the environment close to the culture facility, thus producing localised environmental impacts.

Invoking the ecological footprint concept demonstrates that even wastewater discharged from well-planned and managed aquaculture operations appropriates a range of environmental goods and services, disrupting natural functioning of the system. Over time this may lead to shifts in the species assemblage of the receiving environment or may compromise the ecosystems resilience to natural shocks and perturbations. Therefore, unless a comprehensive and precise long-term monitoring programme is in place, such subtle changes will not be recorded. However, where carrying capacities of ecosystems have been exceeded by aquaculture wastewater discharges, negative impacts have been observed and pertinent examples are presented in the following sections.

1.3. Environmental impacts of aquaculture wastewater

Recent articles have focused public attention on the potential negative environmental impact of aquaculture, particularly the intensive production of salmon and shrimp (Holmes, 1996; Anon, 1997; Hecht, 1998; Naylor, Goldburg, Mooney, Beveridge, Clay, Folke, Kautsky, Lubchenco, Primavera and Williams, 1998; Naylor, Goldburg, Primavera, Kautsky, Beveridge, Clay, Folke, Lubchenco, Mooney and Troell, 2000). However, when assessing the impact of aquaculture a rational appraisal, based on the best available data, must be established to prevent the debate becoming distorted by stakeholder groups with markedly different agendas (Boyd, 1999).

Environmental impacts associated with aquaculture wastewater discharges include the alteration of physio-chemical parameters in the receiving environment, eutrophication, shifting trophic status and interactions, problems associated with the escape of culture organisms and the impact of pathogens released from the aquaculture facility on native species. The cause and effect of these environmental impacts are discussed in the following sections.

1.3.1. Physio-chemical impacts

The appropriation of water for aquaculture and the subsequent discharge of wastewater can have a significant impact on the local hydrology and chemical composition of water and sediments in the receiving environment. Major physical impacts described here include the modification of flow regimes and hydrological conditions in the receiving environment and the effect of sedimentation. Changes in the chemistry of the receiving environment are largely related to the release of nutrients and chemical agents used to treat disease. However, respiration within the aquaculture facility and biological and chemical oxygen demand associated with discharged waste also impact upon the water quality.

Appropriation of water resources

Beveridge and Phillips (1993) reviewed environmental impacts associated with tropical inland aquaculture and noted that water resource appropriation may have several adverse consequences: altering channel morphology and sedimentation patterns, reducing access to spawning and nursery areas, creating barriers to migratory fish, changing thermal regimes and modifying biological communities in the receiving environment. Groundwater abstraction for tropical coastal aquaculture has been implicated in causing subsidence and saline intrusion, whilst the discharge of saline wastewater has caused the salinisation of surface water and land resources (Phillips, Kwei Lin and Beveridge, 1993; Tran, Le and Brennan, 1999). Water resource appropriation for sub-tropical and temperate freshwater aquaculture is likely to have impacts similar to those outlined by Beveridge and Phillips (1993), although water requirements for pond aquaculture are likely to be higher in more arid climates (Boyd and Gross, 2000). These authors also proposed some practical water conservation measures for pond aquaculture that included seepage control through employing good construction practices, limiting water exchange and providing storage for rain and runoff water. Jones (1990) noted that water abstraction for flow-through trout farms reduced flows in river stretches between the farm intake and outflow to such an extent that the movement of migratory fish could be hampered. Situations such as this could be avoided through improved site selection and restricting abstraction to well below the minimum recorded flow; supplementary aeration and water reuse also have potential for improving the efficiency of water use in aquaculture.

Sedimentation

Deposition of particulate matter entrained in aquaculture wastewater can be unsightly, but also causes additional problems. Siltation can smother invertebrates and macrophytes, and increase substrate embeddedness, reducing interstitial water flow and restricting the access of certain fish species to substrates suitable for spawning. Sedimentation is more likely where the dilution capacity of the receiving environment is limited (Jones, 1990). Within the receiving environment, particular habitats are more vulnerable to siltation i.e. the deeper reaches of rivers and streams downstream of commercial aquaculture operations and directly beneath cage farms. Excessive sedimentation can inundate the habitat, altering the sediment composition, restricting colonisation and eventually leading to anoxia that may contribute to eliminating pollution tolerant species.

Pollutants

In addition to the physical impacts outlined above, several potential pollutants are discharged to the receiving environment entrained in aquaculture wastewater (Beveridge, Phillips and Clarke, 1991). Primarily, these waste fractions arise from uneaten feed, excreta and faecal material and chemical treatments used to maintain water quality and eradicate disease (Beveridge and Phillips, 1993).

Ammonia discharged from fish farms and released during the degradation of proteinatious waste can be toxic to invertebrates, particularly in lentic habitats. Streams and rivers in Scotland are particularly vulnerable as they generally have high water quality, characterised by diverse communities of benthic invertebrates, many of which are intolerant of pollution (NCC, 1990). However, several studies show that nutrients released from aquaculture contribute only a small proportion to the overall anthropogenic input to aquatic ecosystems (Ackefors and Enell, 1990; Foy and Rosell, 1991; Kronvang et al. 1993; Páez-Osuna, Guerrero-Galván and Ruiz-Fernández, 1998). Compared with other nutrient sources, shrimp aquaculture in the coastal states of Mexico contributes only 1.5% to total nitrogen, and 0.9% to total phosphorus inputs to the marine environment (Páez-Osuna et al., 1998). However, these relatively small inputs have been implicated in causing adverse local environmental impacts. Reports of eutrophication and shifts in species assemblages and interactions associated with wastewater discharged from aquaculture are reviewed later.

Organic and inorganic fertiliser employed in semi-intensive pond aquaculture may significantly increase nutrient levels in the culture water. However, the minimal exchange of water in these systems limits the release of nutrients to the receiving environment. The use of chemicals in aquaculture, including inorganic fertiliser, has been reviewed in a number of studies (Beveridge et al., 1991; Beveridge and Phillips, 1993; Phillips et al., 1993; Bergheim and Åsgård, 1996). Lime is used extensively to condition pond sediments between production cycles, but impacts on the external environment are likely to be limited. More problematic may be the use of highly toxic and persistent chemicals such as chlorinated hydrocarbons and organotins to eradicate predators, competitors and disease vectors (Phillips et al., 1993). Piscicides and molluscicides derived from plant extracts e.g.

rotenone, saponin and nicotine are widely used in tropical aquaculture (Baird, 1994) and Phillips et al. (1993) suggest that the environmental impact of using these products is expected to be limited as they readily biodegrade. However, the non-specific nature of these compounds and their possible impact on the health of workers during preparation and application represent serious concerns (Baird, 1994).

Disinfectants, e.g. sodium hypochlorite, formalin and benzalkonium chloride, used in production facilities, particularly hatcheries, may be discharged in aquaculture wastewater; however, the consequences of this practice have not been investigated (Phillips et al., 1993). Environmental impacts associated with using algicides such as copper sulphate in shrimp ponds have not been studied, but impacts are likely to depend on the application rate and degree to which water is exchanged between the culture facility and receiving environment. Copper-based antifoulants used to treat nets on salmon cages have been associated with elevated concentrations of copper in sediments close to culture facilities, although, no harmful impacts were reported (Bergheim and Åsgård, 1996).

The majority of other chemicals employed in aquaculture are used to treat disease and combat parasites i.e. antibiotics, parasiticides and fungicides. However, despite widespread use in tropical aquaculture, information concerning the extent and impact of these chemicals is limited. Burka, Hammell, Horsberg, Johnson, Rainnie and Speare (1997) presented a comprehensive review of the current status of chemotherapeutant use in salmonid aquaculture; compared with mammalian therapeutants the range of treatments is limited, and application is restricted to anaesthesia and anti-infective agents preventing parasitic and microbial diseases.

In many cases a large proportion of the medication is not absorbed and retained by the culture organisms, but passes out of the aquaculture system to the receiving environment (Weston, 1996). Antibacterial medications are often delivered as feed additives, however, diseased fish generally have a suppressed appetite, decreasing the proportion of medicated feed consumed. The discharge of therapeutants from fish farming operations is increased further as the majority of active agents ingested are not absorbed in the gut but expelled, unmodified, in faeces. It has been estimated that due to the factors outlined above, 62-95% of antibacterial agents such as oxytetracycline and oxolinic acid may be discharged from culture facilities (Weston, 1996).

The fate and potential impact of therapeutants entrained in aquaculture wastewater from commercial facilities has been investigated in a number of studies (Pouliquen, Le Bris and Pinault, 1993; Smith, Donlon, Coyne and Cazabon, 1994; Davies, McHenery and Rae, 1997). Residues from therapeutants used in aquaculture may pose a serious threat to non-target invertebrate communities in the receiving environment as many chemical treatments used in aquaculture are designed to eradicate invertebrate pathogens. Ivermectin (22,23-dihydroavermectin B1) used to treat sea-lice (*Caligus elongatus* and *Lepeophtheirus salmonis*) infestations at salmon cage sites has been implicated in eradicating communities of benthic invertebrates and damaging shellfish fisheries. However, research suggests that using this agent in aquaculture is unlikely to cause acute direct toxic effects in invertebrates and that bivalves in the receiving environment will not bioaccumulate detectable concentrations (Davies et al., 1997).

Several studies have recorded an increase in the occurrence of antibiotic-resistant bacteria following the widespread introduction of antibacterial agents in aquaculture. Weston (1996) provides a comprehensive review regarding the stimulation of antibacterial resistance through antibiotic use in aquaculture. Treatment using oxolinic acid, oxytetracycline and potentiated sulphonamide on rainbow trout (*Oncorhyncchus mykiss*) farms resulted in elevated numbers of bacteria resistant to these agents being released to the receiving environment (Austin, 1985). Oxytetracycline use at cage farms culturing salmonids was associated with an increased abundance of resistant bacteria in sediments underlying the farms (see Weston, 1996) and cross-resistance between oxytetracycline and potentiated sulphonamide in sedimentary bacteria (Gray, Weston and Herwig, 1994; cited in Weston, 1996). Microcosm studies confirmed that adding oxytetracycline-medicated feed increased both the number of oxytetracycline-resistant bacteria and bacteria resistant to potentiated sulphonomide (Gray and Herwig, 1994; cited in Weston, 1996).

Brown (1989) highlighted the potential problem regarding the prophylactic use of antibiotics in aquaculture, and suggested that their unregulated low-level application in penaeid shrimp hatcheries in Ecuador may have contributed to the development of antibiotic-resistant bacteria. Of particular importance from a human perspective is that this practice may lead to the development of antibiotic-resistance in aquatic bacteria i.e. *Vibrio cholerae*, *Vibrio parahaemolyticus*, *Vibrio vulnificus* and *Vibrio alginolyticus*, which have been identified as human pathogens.

In general, the use of chemicals in temperate aquaculture is closely regulated to ensure operator safety, protect the environment and safeguard the quality of the product. In tropical developing countries where aquaculture production has increased dramatically, the rate of expansion in the industry has frequently overwhelmed attempts to regulate and monitor the industry. Consequently, problems associated with chemical use in aquaculture may be more acute in developing countries with poorly defined regulatory frameworks. However, as institutional arrangements evolve, the operators of aquaculture facilities are likely to require appropriate management strategies and treatment technologies to limit the release of chemicals in wastewater.

Dissolved oxygen

Aquaculture wastewater has elevated biological and chemical oxygen demands (Kelly and Karpinski, 1994), therefore directly following discharge, dissolved oxygen (DO) levels, already depressed by respiration within farms, can be reduced further. Boaventura, Pedro, Coimbra and Lencastre (1997) reported that mean DO concentration in the Coura River, 250 m downstream of a rainbow trout farm in northern Portugal producing 500 t y^{-1}, were 9.6 mg l^{-1}, as compared with 10.7 mg l^{-1} upstream of the farm. However, mean biochemical oxygen demand (BOD) levels in the river were 5.6 mg l^{-1} at distances of 250 m and 1000 m downstream of the farm, as compared with 1.6 mg l^{-1} upstream. Furthermore, BOD levels only returned to background levels 12 km downstream of the farm and the maximum recommended BOD level for salmonid waters (3 mg l^{-1}; EEC, 1978; cited by these authors) was exceeded in a 1 km stretch of the river. Cornel and Whoriskey (1993) noted oxygen depletion in the water column (0-9 m depth) close to rainbow trout cages in Lac du Passage, Quebec. Typical monitoring data show that at a control site ~400 m from the farm, DO levels of ~12 mg l^{-1} were recorded at 4 m, however, adjacent to the cages DO levels were ~6 mg l^{-1} at this depth. Direct impacts of decreased oxygen levels on the native biota were not recorded; the authors raised concern that the stock may have been stressed, although no supporting evidence was presented.

Compared with oxygen depletion rates in other sea lochs, Gillibrand, Turrell, Moore and Adams (1996) attributed elevated levels in the bottom waters of Loch Ailort, a sea loch on the west coast of Scotland, to the microbial breakdown of organic matter. These authors estimated that 50% of the total particulate organic carbon supply to the loch could be ascribed to two cage-based salmon farms. Increased sediment oxygen consumption was also observed under salmon cages in the Bay of Fundy, Canada (Hargrave, Phillips, Doucette, White, Milligan, Wildish and Cranston, 1997) and beneath tilapia cages in Lake Kariba, Zimbabwe (Troell and Berg, 1997).

Problems associated with oxygen consumption within aquaculture facilities, and the subsequent compensation for biological and chemical oxygen demand, can be compounded by secondary oxygen consumption (Bailey-Watts, 1994). Decomposition of 1 g of fish feed has been estimated to consume 1.65 g of oxygen. However, phosphorus contained in the feed can sustain 10-40 g of plant production, which will consume 2-6 g of oxygen when decomposed. Despite the negative implications, nutrients released from aquaculture generally represent a small proportion of the overall input to aquatic ecosystems. Depressed oxygen levels associated both directly and indirectly with aquaculture wastewater would be detrimental to native species assemblages, however the magnitude of impact would be influenced by the sensitivity of organisms to reduced DO concentrations, prevailing seasonal and diurnal oxygen regimes, and the re-aeration capacity of the receiving environment.

1.3.2. Eutrophication

The Council of the European Communities (1991) defines eutrophication as "the enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned." Aquaculture wastewater may contribute to eutrophication; the discharge of organic and inorganic matter from commercial aquaculture has been implicated in stimulating the production of phytoplankton adjacent to cage farms (Bonsdorff, Blomqvist, Mattila and Norkko, 1997). Increased phytoplankton abundance and biomass are frequently the first evidence of eutrophication, making phytoplankton communities important indicators for monitoring environmental impacts associated with aquaculture.

Arzul, Clément and Pinier (1996) demonstrated that seawater used to rear turbot (*Psetta maxima*) in tanks enhanced the growth of the marine diatom, *Chaetoceros gracile* and dinoflagellate, *Gymnodinium* cf. *Nagasakiense*, during *in vitro* studies. However, dinoflagellate (*Alexandrium minutum*) production was inhibited, suggesting that wastewater from turbot farms may significantly change the assemblage of phytoplankton in the receiving environment. The response of phytoplankton populations to discharges from aquaculture depends on factors such as wastewater composition, initial phytoplankton community, ambient conditions and nutrient status of the receiving environment.

Phytoplankton bioassays conducted with *Selenastrum capricornutum* and *Oscillatoria redekei* revealed that bioavailable phosphorus concentrations in water passing through six land-based rainbow trout and Atlantic salmon (*Salmo salar*) farms increased significantly (Massik and Costello, 1995). This implies that the wastewater discharged is likely to enhance phytoplankton production in receiving environments. However, in some freshwaters there will be little enhanced phytoplankton growth, even in response to large wastewater inputs, as other growth factors such as dissolved humic matter, carbon, trace elements and vitamins, will limit production (NCC, 1990).

Concern in the United Kingdom is growing over the rising incidence of toxic algal blooms (Pearce, 1996) and the possible association, in some instances, with aquaculture wastewater (Anon, 1999). This association was debated following suggestions by Folke, Kautsky and Troell (1994) that modifying the composition of commercial fish feed may stimulate the production of toxic algal species (Black, Gowen, Rosenthal, Roth, Stechy and Taylor, 1997; Folke, Kautsky and Troell, 1997). The discussion focused on the fact that lowering the phosphorus content of feed may increase the ratio of nitrogen to phosphorus in the wastewater, contributing to the selection of toxic species.

Carr (1988, cited in NCC, 1990) found that below a rainbow trout farm on the River Hull in England, the growth potential for algae (measured by a *Selenastrum* bioassay) and growth of periphyton on polyethylene sheets increased when compared with upstream controls. In streams directly below three trout farms, Selong and Helfrich (1998) recorded a significantly higher standing crop of periphyton on rocks. The downstream standing crop of periphyton also showed a positive correlation with feed loading rates at the farms and a negative correlation with overhead cover.

Whilst monitoring natural populations of periphyton (*Cladophora glomerata*) in the sheltered bays of south-west Aland in the northern Baltic Sea, Ruokolahti (1988) recorded a significantly higher biomass at sites close to cage farms culturing rainbow trout as compared to reference sites. However, Stirling and Dey (1990) compared the development of periphyton communities on artificial substrates positioned below 75 floating cages containing 135 t of rainbow trout, and at a control site 1.61 km south-west of the cage site in Loch Fad. Of twenty-two periphyton species recorded, only one species had a significantly different annual geometric mean abundance between the two sites. The absence of demonstrable differences between the farm and control site were attributed to the high nutrient status of the loch causing phytoplankton and periphyton production to be light, not nutrient limited. The situation was compounded by high turbidity levels associated with farm waste; wind induced turbulence was also believed to regulate algae growth within the loch.

Increased growth of epiphytes below trout farms was implicated in smothering *Ranunculus penicillatus*, causing growth to be retarded (Carr, 1988; cited in NCC, 1990). In addition, the possible decline of the rarest plant in Britain (*Crassula aquatica*) in Loch Shiel has been attributed to increased rural development and possibly the effects of cage fish farms (NCC, 1990). It is hypothesised that an increase in available nutrients leads to the selection of macrophyte species with the largest assimilative capacity, causing diversity to decline, whilst weedy species capable of rapid seasonal growth and producing large numbers of propogules become dominant. Changes in the composition, abundance and distribution of macrophyte communities can also result in the movement of sediments previously consolidated by the vegetation, altering the hydrology of the stream or river.

Nutrients released from aquaculture have been associated with increased primary production in the receiving environment and Table 1.1 presents a review of the accounts described above. However, studies regarding the extent of the problem are limited. The role of aquaculture wastewater in eutrophication will depend upon the nature of the receiving environment, including background nutrient status and water chemistry, hydrology and carrying capacity. However, when nutrients are limiting primary production, hydrological conditions restrict water exchange and the capacity for nutrient assimilation in the receiving environment is low, nutrients released from aquaculture may make a significant contribution to the process of eutrophication.

1.3.3. Shifting trophic status and interactions

Enrichment of a water body may cause a shift in trophic status resulting in an increased abundance of pollutant tolerant species, whilst overall diversity decreases. The impact of aquaculture wastewater on benthic invertebrate communities, zooplankton assemblages and resident fish populations is described in the following sections and summarised in Table 1.1.

Benthic invertebrates

Loch, West and Perlmutter (1996) and Selong and Helfrich (1998) monitored the impact of trout farming in North America on communities of macro-invertebrates. Loch et al. (1996) recorded a significantly lower diversity of pollution intolerant taxa i.e. Ephemeroptera, Plecoptera and Trichoptera (EPT) in a stream below three trout farms in western North Carolina, as compared with upstream control stations. Taxa richness 1.5 km downstream had increased compared with that recorded at the farms, however, it remained significantly below that observed at upstream controls. Selong and Helfrich (1998) recorded similar observations in streams receiving wastewater from five commercial trout farms in Virginia: the abundance and diversity of sensitive taxa decreasing downstream, whilst pollution tolerant taxa increased.

Beneath salmon cages in Bliss Harbour, Bay of Fundy, Canada, a decrease in the sediment meiobenthic taxa diversity was recorded and found to be associated with increasing levels of organic enrichment (Duplisea and Hargrave, 1996). Similar modifications in the assemblage of benthic species beneath other cage farms and mussel rafts have been recorded (Karakassis, Hatziyanni, Tsapakis and Plaiti, 1999; Stenton-Dozey, Jackson and Busby, 1999), however, observable impacts commonly extend little beyond 100 m from the culture facility (Costa-Pierce, 1996). The extent of environmental impacts from organic enrichment caused by aquaculture is dependent on hydrological factors e.g. flushing, resuspension and dispersion. De Grave, Moore and Burnell (1998) suggest that the highly dissipative nature of a site in Dungarvan Bay, Ireland, used to culture oysters (*Crassostrea gigas*) on trestles meant that no evidence of organic enrichment or changes in the benthic community was found.

Zooplankton communities

Eutrophication frequently changes the abundance, species composition and size structure of natural zooplankton communities (Blancher, 1984; cited in NCC, 1990). Several zooplankton species have specific food requirements; therefore changes in phytoplankton communities may induce a trophic cascade. Eutrophication may lead to oligotrophic communities characterised by copepods being replaced with communities dominated by large numbers of small herbivorous zooplankton (Allan, 1976), leading in turn to changes in the phytoplankton community. The grazing pressure exerted on phytoplankton populations may be altered, resulting in the selection of certain species types or size ranges, ultimately changing the standing biomass and structure of the phytoplankton community. Furthermore, the proliferation of small herbivorous zooplankton may result in phytoplankton species resistant to grazing e.g. cyanobacteria, becoming dominant and forming dense standing crops.

Zooplanktivorous copepods and fish will be influenced by changes in the composition of zooplankton assemblages. Fish populations may depend on the availability of a specific food source at a specific time of year; larval fish are particularly vulnerable. Recruitment can be severely limited if suitable zooplankton species or size classes are unavailable for periods as short as two weeks, especially soon after hatching (Hommer, 1985; cited in NCC, 1990). The availability of zooplankton can be an ultimate factor for bringing fish into breeding condition e.g. Arctic charr (*Salvelius alpinus*).

The migration of zooplankton and deposition of faecal pellets can play a significant role in the movement and fate of nutrients in the water column (Anderson et al., 1988; cited in NCC, 1990; Ferrante and Parker, 1977). Reducing the abundance of copepods can decrease the rate of nutrient transfer to sediments in faecal pellets, while increased numbers of small cladocerans that recycle faecal material can maintain nutrient levels in the water column, increasing the possibility of eutrophication (Lehman, 1980).

Fish assemblages

Oberdorff and Porcher (1994) investigated the impact of wastewater from farms in Brittany, France using the Index of Biotic Integrity (IBI), a fish-based index that relies on the assemblage of fish communities changing in a characteristic manner, dependent upon stream quality. Compared with upstream communities, the abundance and biomass of fish species downstream from nine fish farms generally increased, with pollution tolerant rudd (*Rutilus rutilus*) and exotic rainbow trout appearing; however, the pollution sensitive goby (*Cottus gobio*) was no longer present.

Selong and Helfrich (1998) employed a modified IBI, accounting for the characteristically low diversity of fish species in Appalachian headwater streams, to assess the impact of wastewater from fish farms in Virginia on the assemblage of fish species in receiving streams. Species diversity consistently increased downstream of the farms, with increasing primary production downstream implicated in expanding the trophic base and encouraging diversity in the fish species present (Selong and Helfrich, 1998). Oligotrophic rivers and lochs in temperate environments, such as those found in Scotland, frequently play host to communities of salmonids. However, application of the IBI in assessing the impact of wastewater from aquaculture in these environments may be limited. The general lack of species diversity in oligotrophic streams requires a modified version of the IBI to be employed, otherwise a disproportionate influence to the abundance of particular taxa is given (Selong and Helfrich, 1998).

1.3.4. Escapees

With all animal production systems there is the potential for escapees, and when considering aquaculture it is likely that the majority of escapees will be entrained in discharged wastewater. Environmental impacts associated with the unintentional introduction of non-native species from aquaculture facilities have been reviewed extensively (Welcomme, 1988; Beveridge, Ross and Kelly, 1994; Arthington and Bluhdorn, 1996; Bardach, 1997; Beardmore, Mair and Lewis, 1997). Beveridge and Phillips (1993) proposed five distinct impacts associated with introductions from aquaculture: direct disruption to the receiving environment; disruption to the host community, through predation or competition; genetic degradation of local stocks; the introduction of diseases and pathogens; socioeconomic impacts. A more detailed assessment of these impacts is beyond the scope of the current thesis, although the references quoted should provide useful further information where required.

1.3.5. Summary of ecological impacts

Negative ecological impacts associated with commercial aquaculture wastewater are summarised in Table 1.1. Despite the widespread nature of aquaculture, the cost and difficulty in monitoring environmental impacts means there are relatively few accounts describing the frequency with which aquaculture operations cause negative environmental impacts. However, where the carrying capacity of the receiving environment is exceeded it may be anticipated that one or more of the impacts described above will be evident.

1.4. Resource-use conflicts

Despite environmental impacts associated with aquaculture operations, the lack of value ascribed to environmental goods and services means costs associated with discharging aquaculture wastewater are usually only considered when functions serving society are effected, or other users of the aquatic resource inconvenienced. Therefore, to evaluate the costs of discharging aquaculture wastewater more comprehensively, the following sections assess the risk of self-pollution, impacts on other aquatic resource users, disruptions to ecosystem functionality and impacts on non-use values.

1.4.1. Self-pollution

Wastewater from land-based aquaculture is frequently discharged to streams and rivers supplying downstream commercial aquaculture operations, whilst waste discharged from cage farms may be conveyed to other facilities by currents and tides. However, in the case of cage aquaculture and pump-ashore facilities there is a danger that wastewater discharged to the receiving environment may contaminate water subsequently used to supply the culture facility.

Problems of self-pollution have been associated with shrimp farming in the inter-tidal area surrounding the Mundel-Dutch Canal lagoon system in the northwestern province of Sri Lanka. Farms were discharging wastewater containing high concentrations of ammonia, nitrite, nitrate and metals to the same areas used to supply their own, or neighbouring culture systems. The problem was compounded as mixing in the lagoon

Table 1.1: Ecological impacts associated with wastewater discharges from commercial aquaculture

Genera	Impact	Setting	Reference
Phytoplankton	- modified growth potential for marine diatoms and dinoflagllates	- wastewater from turbot rearing tanks tested in bioassays	Arzul et al. (1996)
	- *Selenastrum capricornutum* and *Osillatoria redekei* bioassays demonstrated an increase in bioavailable phosphorus	- wastewater from six land-based farms culturing trout and salmon	Massik and Costello (1995)
Periphyton	- enhanced growth on polyethylene sheets	- study conducted downstream of a trout farm on the River Hull, England	Carr (1988; cited in NCC, 1990)
	- increased standing stock on rocks	- observations made in streams directly below trout farms in Virginia, USA	Selong and Helfrich (1998)
	- significantly higher biomass of *Cladophora glomerata* in sheltered bays	- recorder in sheltered bays adjacent to cage farms culturing trout in the northern Baltic Sea	Ruokolahti (1988)
Macrophytes	- increased epiphytic growth implicated in smothering *Ranunculus penicillatus*	- observed downstream from trout farms on the River Hull, England	Carr (1988; cited in NCC, 1990)
	- decline of the rarest British plant *Crassula aquatica*	- attributed to possible effects of increased cage aquaculture in Loch Shiel, Scotland	NCC (1990)
Benthic invertebrates	- decreased diversity of pollution sensitive species (EPT), with taxa richness remaining significantly lower 1.5 km from the farm	- downstream of trout farms in North Carolina, USA	Loch et al. (1996)
	- decreased abundance and diversity of pollution sensitive taxa (EPT) with an increase in pollution tolerant species	- downstream of trout farms in Virginia, USA	Selong and Helfrich (1998)
	- decreased taxa diversity with increasing sediment organic enrichment under the salmon cages	- samples collected from beneath salmon cages in Bliss Harbour, Bay of Fundy, Canada	Duplisea and Hargrave (1996)
Fish	- increase in abundance and biomass with pollution tolerant *Rutilus rutilus* and exotic *O. mykiss* appearing, whilst the pollution sensitive goby *Cottus gobio* was excluded	- observations made in streams receiving wastewater from farms in Brittany, France	Oberdorff and Porcher (1994)
	- increased species diversity downstream attributed to increased primary production increasing the trophic base	- recorded in Appalachian headwater streams in Virginia receiving wastewater from trout farms	Selong and Helfrich (1998)

is limited due to the shallow bathymetry and a sandbar that prevents exchange with the marine environment. Furthermore, destruction of mangroves and salt marshes to accommodate new shrimp farms created elevated sediment loads and decreased the capacity of the environment to buffer floods. These factors resulted in the shrimp farms extracting water from the lagoon, which had a high biochemical oxygen demand and contained concentrations of ammonia, nitrite, nitrate, sulphides and suspended solids considered unacceptable for shrimp farming. This caused disease outbreaks, encouraged parasite infestations, increased the fouling of gill structures with suspended solids, retarded growth rates and resulted in the production of poor quality shrimp (Corea, Johnstone, Jayasinghe, Ekaratne and Jayawardene, 1998). Similarly, wastewater discharged from large shrimp farms in Indonesia caused water quality problems in adjacent small-scale *tambak* shrimp farms that were unable to regulate the exchange of water between ponds (Muluk and Bailey, 1996).

Increased oxygen consumption by microbial communities in sediments below cage farms may lead to anoxic conditions and the evolution of methane and hydrogen sulphide. The release of gaseous methane and hydrogen sulphide has been implicated in causing gill disease at salmon farms (Black, Ezzi, Kiemer and Wallace, 1994). Furthermore, respiration by benthic invertebrates and microbial communities in sediments receiving organic inputs from cage farms may depress oxygen concentrations in the water column. Reduced oxygen concentrations in the water column have been observed beneath salmon cages in Scotland (Lumb, 1989; Gillibrand et al., 1996) and upwelling of this anoxic bottom water poses a serious threat to the health of cultured fish and has been implicated in causing fish kills (Beveridge, 1996).

Fishermen and environmentalists have hypothesised that nutrient discharges from salmon aquaculture have stimulated toxic algae blooms, which have disrupted shellfish fisheries on the west coast of Scotland. However, the regulatory authorities that impose restrictions on the fisheries attributed the blooms to natural phenomenon (Anon, 1999). Consumption of shellfish that have accumulated toxins from algae has caused Paralytic Shellfish Poisoning, which can be fatal. Toxins excreted by cyanobacteria may also damage the liver and nervous system of animals consuming the water; dogs and cattle have died after drinking contaminated water or licking toxic scum from their coats, and birds and fish have been killed. Although the blooms often collapse suddenly, the toxins can persist in the environment, possibly entering the aquatic food web, bioaccumulating in herbivorous water fleas and potentially biomagnifying in secondary consumers.

1.4.2. Restricted amenity

Discharging aquaculture wastewater can affect users of the aquatic resource, other than aquaculture operators. Deteriorating water quality in the Mundel-Dutch Canal lagoon system, which has been attributed to the increase in shrimp farming, has been blamed for the observed decline in the capture fisheries of the lagoon. This has caused resentment in local fishing communities, which has manifested itself as poaching (Corea et al., 1998). A survey of local community members revealed that a number of people complained of skin diseases attributed to poor water quality. Although no direct link has been established, concerns of this type may strengthen opposition to shrimp farming in the local population.

Saline wastewater discharged from shrimp farms in Songkhla, Thailand has been implicated in causing the death of livestock drinking from canals (Primavera, 1997). The author further reports that following the salinisation of surface water caused by the discharge of wastewater from shrimp farms in Nellore district, India, women were forced to spend longer collecting drinking water from distant sources. Yields on farms cultivating rice have also been affected by saline water released from neighbouring shrimp farms (Tran et al., 1999); however, it appears that this situation has arisen from leaching during the dry season and bunds being breached during the growing season. Tran et al. (1999)

noted that shrimp farming may increase the sediment load to local canals and rivers and Phillips et al. (1993) also reported that sediment in wastewater from shrimp farms in Thailand and Sri Lanka caused irrigation canals to become silted.

Aquaculture wastewater may encourage nuisance growths of macrophytes that interfere with recreational activities, for example, angling, swimming and boating. In Scotland, possible disease transmission from cage farms to native fish stocks has been blamed for causing a decline in the number of Atlantic salmon returning to recreational capture fisheries (NCC, 1990). Estimates have demonstrated that declines in recreational fisheries may have severe implications for rural economies that receive income from visiting anglers. The introduction of disease with the signal crayfish (*Pacifasticus leniusculus*) reduced native crayfish (*Austropotamobius pallipes*) populations in Europe and caused the wild crayfish capture fishery to decline. Furthermore, reduced indigenous crayfish populations allowed aquatic macrophytes to proliferate, resulting in the elimination of habitat for game fish (Thompson, 1990).

1.4.3. Reduced functionality

Burbridge (1994) described natural wetland functions that produce a wide array of environmental goods and services that sustain economic activities and societal systems. However, wastewater discharged from commercial aquaculture operations may have a detrimental effect on the functional integrity of wetlands, disrupting the supply of environmental good and services. Considering the discharge of wastewater from shrimp farms, Robertson and Phillips (1995) suggested that the assimilative capacity of a mangrove ecosystem may be exceeded and that excessive loads of ammonia and organic matter could lead to anaerobic sediments, resulting in tree mortality.

Elimination of mangrove trees may result in the loss of a number of functions. Reduced production of mangrove biomass will be accompanied by a decrease in the assimilation and cycling of nutrients, potentially leading to nutrient export from the mangrove ecosystem, and, consequently, adverse impacts on the receiving environment. Decline of the mangrove root system could decrease sediment stability, leading to erosion, which could increase the risk of saline water intruding inland. The loss of mangrove habitat may also represent a reduction in the extent of nursery areas for juvenile fish and shrimp, and contribute to a general decrease in biodiversity.

1.4.4. Impacts on option and non-use values

Reduction in quality of the aquatic environment can influence the value an individual attributes to preserving the resource to allow the individual, other individuals and future generations the option of using the resource at a later date (Muir, Brugere, Young and Stewart, 1999). The impact of an activity on this *option value* may be estimated by assessing the willingness-to-pay (WTP) of an individual to preserve the environment. Folke et al. (1994) extrapolated marginal costs of [1]SEK 50-100 and SEK 20-30 kg^{-1} for nitrogen and phosphorus removal, respectively, from sewage in Sweden, to represent the WTP of Swedish society to limit nutrient discharges from salmon aquaculture. Based on a comparison of waste production presented as person equivalents, it was estimated in 1994 that the cost to society of eliminating nitrogen and phosphorus discharges originating from salmon aquaculture equated to SEK 4-4.5 kg^{-1} of production. These authors also calculated that internalising this cost increased production costs for salmon to SEK 31-31.5 kg^{-1}, and although the 1994 farm gate price for salmon was not given, such a cost increase would severely reduce profits and possibly threaten the viability of salmon farming.

Environments also have non-use values. The intrinsic or *existence value* of environments is unrelated to humans and their present, or potential, direct or indirect use of the resource (Turner, 1991; Muir et al., 1999). For example, people that are unlikely ever

[1]1US$ = ~ 6 SEK. Folke et al. (1994).

to visit a region may attribute value to its existence and feel a sense of loss were the ecosystem to be degraded through the discharge of aquaculture wastewater. Degradation of the environment would also reduce the value ascribed to passing the asset onto future generations, termed the *bequest value*. Therefore, although changes in non-use values of environments due to aquaculture wastewater discharges have not been described, they may be expected to be negative.

1.4.5. The cost of discharging aquaculture wastewater

The previous sections present a review of the environmental impacts, cases of self-pollution, restricted amenity and functionality and decreased non-use values associated with discharging aquaculture wastewater. Table 1.2 summarises the range of environmental impacts and stakeholder conflicts identified.

Growing recognition that aquaculture wastewater may be responsible for a wide range of environmental costs and stakeholder conflicts has lead to increased legislation and discharge monitoring in many countries. Therefore, unless managers of aquaculture facilities adopt new strategies to reduce the volume and concentration of wastewater produced, they may find themselves liable to taxes, fines and ultimately closure. The perception of consumers is also important, and where concern exists over the impacts associated with aquaculture, the demand, and subsequently value, of products from the aquaculture sector may decline.

1.5. Strategies for managing aquaculture wastewater

Operators of aquaculture facilities have been presented with a range of strategies for reducing pollutant concentrations in discharges from their facilities (NCC, 1990; Cripps and Kelly, 1995). Phillips et al. (1993) suggest that appropriate site selection, improved

Table 1.2: Negative impacts associated with aquaculture wastewater

Impact	Consequence	Key features	Reference
Physical	- modified hydrology	- reduced flow rates, modified channel morphology and flow regimes and salinisation of surface waters in coastal areas	Beveridge and Phillips (1993); Phillips et al. (1993); Tran et al. (1999)
	- sedimentation	- reduced interstitial water flow, increased embeddedness and sediment anoxia	
Chemical	- pollutants	- excreted ammonia toxic to invertebrates; waste nutrients lead to hypernutrification; therapeutants and their residues affect non-target organisms leading to antibiotic resistant strains	NCC (1990); Phillips et al. (1993); Weston (1996); Davies et al. (1997)
	- reduced oxygen concentrations	- on-farm respiration reduces oxygen levels leading to exclusion of sensitive species; biological and chemical oxygen demand further depletes oxygen levels	Gillibrand et al. (1996); Kelly and Karpinski (1994)
Nutrient enrichment	- eutrophication	- increased phytoplankton and periphyton production near cages, including possible stimulation of toxic algae blooms; increased epiphyte growth downstream of land-based farms; elevated respiration during decomposition	Anon (1999); NCC (1990); Bonsdorff et al. (1997); Selong and Helfrich (1998)
Shifting trophic status	- modified species assemblages	- elimination of pollution sensitive invertebrates and fish; increased abundance and biomass of tolerant species and eutrophication leading to trophic cascades	Oberdprff and Porcher (1994); Loch et al. (1996); Selong and Helfrich (1998)
Escapees	- predation, competition and ecological impacts	- ecosystem disruption through foraging and consumption of native flora and fauna; escapees breed with resident populations leading to genetic degradation	Welcomme (1988); Arthington and Bluhdorn (1996); Bardach (1997)
Disease and parasites	- loss of native species	- viruses, bacteria and parasites infest native populations, exotic parasites may also devastate non-resistant indigenous populations	Arthington and Bluhdorn (1996); McAllister and Bebak (1997)
Self-pollution	- decreased production and product quality	- upwelling of anoxic water causing fish-kills in cages; reduced water quality leading to disease outbreaks and stimulation of toxic algae blooms	Lumb (1989); Black et al. (1994); Corea et al. (1998)
Restricted amenity	- decline in capture fisheries and water quality	- competition and disease can damage capture fisheries; sedimentation and plant growth restricts water flow in navigation and irrigation canals; reduced water quality affects access of livestock and humans to water causing social unrest	NCC (1990); Phillips et al. (1993); Primavera (1997); Tran et al. (1999)
Reduced functionality	- loss of ecological functions	- discharged wastewater can degrade ecosystems leading to habitat loss, decreased diversity, restricted storage capacity for nutrients and water and disruption to flows of environmental goods and services	Burbridge (1994); Robertson and Phillips (1995)
Impacts on option and non-use values	- reduced perception of aquatic resources	- degraded aquatic environments and stakeholder conflicts lead to a negative perception of aquaculture, with reduced values attributed to the ecosystem	Turner (1991); Folke et al. (1994); Muir et al. (1999)

pond management, wastewater treatment and adopting effective planning and monitoring may reduce the environmental impact associated with shrimp farm wastewater. However, strategies proposed by researchers and commercial enterprises with a vested interest in uptake are sometimes constrained by concerns regarding reliability, practical limitations, possible risks and financial demands. The following sections present a review of strategies to limit pollutant discharges. Potential advantages and constraints are described; findings are summarised in Table 1.3.

1.5.1. Feed technology and management

The development of new feed types has played a significant role in reducing pollution associated with aquaculture (Cho and Bureau, 1997). Unlike traditional milled pellets that are pressed together, extruded pellets are expanded and fused together, fully gelatinising the starch component to form a matrix. This makes the pellet more digestible and reduces the amount of faeces produced. Seymour and Johnsen (1990; cited in Seymour and Bergheim, 1991) found extruded feed gave a food conversion ratio of 1.05 as compared with 1.12-1.25 for pressed pellets. Furthermore, it was estimated that for every tonne of fish produced, solid waste levels between 320-450 kg for pressed pellets could be reduced to 250 kg with extruded pellets.

In addition to increasing digestibility, extrusion results in a decreased surface area liable to attack when immersed, making extruded pellets more water-stable than milled pellets. Seymour and Johnsen (1990; cited in Seymour and Bergheim, 1991) found that commercial extruded pellets immersed for 24 hours remained 84% intact with respect to dry weight, whilst two types of commercial pressed pellets of the same size remained only 50% intact after 17 and 53 minutes. Extruded pellets can contain increased lipid levels, giving them a higher energy density. Energy dense diets reduce the feed required per unit production as increasing lipid concentration leads to protein sparing. This reduces nutrient discharges as the major source of phosphorus and nitrogen pollution from fish farms is protein catabolism (Henriksson, 1989 and Bergheim et al., 1990; cited in Seymour and Bergheim, 1991).

Crampton (1987) noted that "phosphorus outputs from fish farms are determined by the level of phosphorus in the feed and by the feed conversion ratios achieved". Therefore, reducing the concentration contained in feed represents another strategy to limit phosphorus discharges. Wiesmann, Scheid and Pfeffer (1988) found that reducing the phosphorus content of semi-purified trout diets from 10 g kg^{-1} to 4 g kg^{-1} caused no significant reduction in growth rate or feed conversion.

Stuart (1953) and Rimen and Power (1978) (cited in Jørgensen and Jobling, 1992) observed that cultured brown trout and Atlantic salmon only attack moving particles, ignoring those coming to rest on the bottom. The extrusion process can be employed to produce pellets that float or sink slowly, giving fish an extended period to ingest the pellet before it comes to rest on the base of the culture vessel, or in cage culture, passes out of the enclosure. In cages, however, increased buoyancy may result in greater feed loss through the sides. Beveridge (1996) notes that fine mesh screens around the tops of cages can minimise feed losses due to wind and wave action, but increased fouling and reduced water exchange may constrain the uptake of this approach. Ensuring feed is introduced at the cage centre would also limit losses through the cage sides. However, restricting feed delivery to part of the cage may increase competition, causing reduced and uneven growth.

The feeding regime adopted at a farm is the approach with the greatest potential to maximise feed utilisation. Seymour (1985 and 1989a; cited in Seymour and Bergheim, 1991) found that fish experience cycles of appetite associated with emptying and filling of the stomach. Employing video equipment, Kadri, Metcalfe, Huntingford and Thorpe (1991) observed that salmon in cages feed actively early in the morning and again in the afternoon; feed presented between these times was largely wasted. Therefore, two feeding events during these active periods were proposed to maximise the assimilation of feed and limit pollution. Fixed feeding times also promote behavioural responses that contribute to improved feed intake and conversion ratios (Seymour, 1989b; cited in Seymour and Bergheim, 1991). The efficiency of feed utilisation during the proposed feeding periods may be optimised through the use of demand feeders (Alanärä, 1992).

Rough handling and transport creates dust in feed bags and it has been estimated that the production of feed dust could amount to 10 t y^{-1} of dry waste on an average size cage farm using pressed pellets (Bergheim et al., 1990; cited in Seymour and Bergheim, 1991). The superior structure of extruded pellets as compared to pressed pellets means that lower quantities of dust are produced. Careful handling may contribute to reducing the production of dust and the operator could sieve the pellets prior to feeding to reduce the level of dust introduced to the water. Where feed dust remains a problem, machines are available that produce new pellets from dust and crushed pellets at the farm site (Seymour and Bergheim, 1991).

Henderson and Bromage (1987) proposed the use of low pollution diets as a highly effective strategy for reducing the environmental impact of aquaculture wastewater. However, limitations to feed constituents and the efficiency of biological production systems indicate that in addition to advancements in feed formulation and delivery, strategies designed to minimise the impact of faecal material and excreted byproducts derived from metabolic processes will also need to be developed. The following sections outline strategies for the design and management of aquaculture facilities aimed at minimising the production of waste products and optimising the performance of wastewater treatment.

1.5.2. Facility design and operation

Improving the design of farms e.g. pipe layout, reducing the length of pipe and number of bends, may reduce the break-up of particulate matter, enabling a larger proportion to be retained during wastewater treatment. Within certain culture facilities, such as ponds, remedial measures to reduce waste discharges may be integrated into the general management of the farm. Permitting macrophytes to colonise channels between ponds could encourage nutrient removal, whilst promoting vegetated pond margins will facilitate nutrient uptake and stabilise embankments, reducing erosion and eliminating scour which can be an important source of suspended solids in pond-based aquaculture (Funge-Smith and Briggs, 1998). Modification of the receiving environment may ameliorate the negative effects of aquaculture wastewater; altering the river or stream morphology could promote mixing and aeration, reducing sediment deposition and compensating for depressed oxygen levels. Such modifications are already undertaken to disrupt the lifecycle of intermediate parasite hosts e.g. weed removal to eradicate the intermediate snail (*Lymnaea* spp.) host of digenean (*Diplostomum* spp.) eye flukes; steps to reduce environmental impacts of wastewater could be incorporated with this work. Furthermore, where appropriate, adopting more proactive strategies, such as building weirs, constructing upstream groynes and installing gravel riffles could enhance the habitat value of the receiving environment.

Wastewater discharged during and directly following feeding, water used to flush culture units and clean screens, and water released after grading and harvesting frequently contain a significant proportion of waste discharged from aquaculture (Briggs and Funge-Smith, 1994; Shireman and Cichra, 1994; Schwartz and Boyd, 1994). Preferential treatment of wastewater released at these times may significantly increase the efficiency of treatment. Lystad and Selvik (1991) and Selvik and Lystad (1991) investigated the potential of *in situ* sludge separation in culture units. Particulate waste produced during salmon farming was concentrated using a circular current in the culture tank, with particles transported to a central sludge cone located in the base of the vessel. The collection of particulate matter using this technique accounted for the retention of 18% of nitrogen and 81% of phosphorus discharged. The greater retention of phosphorus was explained as it is largely bound to faeces, whereas most nitrogenous waste is excreted as soluble ammonia.

Employing oxygenation in aquaculture facilities can reduce the volume of water required, thus improving the efficiency of existing treatment processes or permitting the use of treatment techniques with a limited hydraulic capacity. During an extensive review of the environmental impact of aquaculture in Scotland, the potential of using aeration to reduce water consumption and improve the efficiency of filtration, whilst increasing the capacity of the receiving environment to dilute discharged wastewater, was identified (NCC, 1990). However, as the majority of aquaculture facilities are able to meet existing discharge standards and further intensification of sites is constrained by limitations in disease management and wastewater treatment, the adoption of this technology remains limited.

1.5.3. Wastewater treatment at land-based farms

Settlement

Traditionally, settlement ponds have been employed to reduce concentrations of suspended solids (SS) in aquaculture wastewater discharged from fish farms in the United Kingdom. Solbé (1982) reported that in 1980, 24 of the 141 freshwater farms surveyed in the United Kingdom reported treating wastewater prior to discharge, with all strategies incorporating settlement. A more recent survey of 150 land-based freshwater fish farms in Scotland revealed that 55 were using settlement, whilst 7 were using filtration devices (Allcock and Buchanan, 1994).

Hennessy (1991) assessed the efficiency of earthen settlement ponds in treating wastewater from two freshwater Atlantic salmon farms. Following treatment, the author found that total phosphorus (TP) concentrations increased by 24-170% and total ammonia nitrogen (TAN) concentrations increased by 22-193% as compared with untreated wastewater. Increased TP concentrations were ascribed to the degradation of accumulated sediments within the settlement ponds, while increased TAN concentrations indicated that processes in the ponds were contributing to ammonification. The removal of SS and BOD was variable and limited.

Settlement ponds are poor at controlling the discharge of BOD from temperate aquaculture as ~60% is associated with soluble waste fractions and pond retention times, and water temperatures are too low to facilitate biodegradation. In addition, TAN in aquaculture wastewater is water-soluble, making removal in settlement ponds ineffective. Hennessy (1991) concluded that sludge accumulation was the primary cause of ineffective settlement ponds, emphasising the importance of frequent sludge removal to maintain optimal performance.

Teichert-Coddington, Rouse, Potts and Boyd (1999) demonstrated that settlement could be applied successfully to treating water discharged from shrimp farms during harvest. However, this was only used to treat the final 20 cm of water discharged from the pond during harvesting; this strategy was adopted as the proportion of nutrients and solids is greatest in the final 10-20% of water discharged. Following a retention period of 6 hours, the concentration of SS decreased by 88%, BOD was reduced by 63% and TP and total nitrogen (TN) decreased by 55% and 31%, respectively. Allcock and Buchanan (1994) outlined constraints limiting the effectiveness of settlement ponds in typical situations:

- infrequent removal of deposited sludge, leading to *in situ* decomposition resulting in deterioration in wastewater quality through the release of soluble pollutants;
- inadequate sludge treatment and inappropriate methods of disposal following removal;

- the omission of duplicate ponds to which wastewater may be diverted during maintenance and sludge removal;
- poorly designed and constructed influent structures, resulting in uneven wastewater distribution and increased turbulence, reducing the efficiency of settling and promoting the resuspension of solids;
- incorrectly designed and constructed outflow structures, resulting in resuspension causing deterioration in wastewater quality.

Limiting flow velocities in settlement ponds is considered the most important factor in increasing treatment efficiency (Henderson and Bromage, 1988). Reviewing the performance of 16 settlement ponds for aquaculture wastewater, these authors found that where inlet concentrations are below 10 mg l^{-1}, resuspension of settled solids through turbulent scouring and effective solids removal are problematic. In addition, reducing the concentration of solids below ~6 mg l^{-1} is difficult. The removal efficiency for SS was related to the inlet concentration thus:

$$SS_i - SS_o = 0.975SS_i - 6.26$$

where, $SS_{i,o}$ = SS concentration (mg l^{-1}) in inflow and outflow, respectively.

Employing good design practice may optimise settlement efficiency and avoid resuspension. Henderson and Bromage (1988) recommend that the hydrology of the settlement pond should be engineered to produce flow rates below 4 m min^{-1} with increasing hydraulic loading up to, and including, the maximum permitted discharge of the site. These authors found that resuspension in aquaculture settlement ponds was correlated with a function of mean fluid velocity, and that a flow rate of 4 m min^{-1} limited resuspension whilst maintaining a relatively high loading rate. To further enhance SS removal the authors recommend maintaining flow rates <1 m min^{-1}, however, practical constraints may limit this.

Inflow and outflow channels should be wide enough to avoid rapid flows, thereby ensuring the even distribution of flow throughout the settlement chamber, maximising settlement, preventing short-circuiting and avoiding channelisation which leads to resuspension. Sloping sides and the inclusion of a slight slope towards the outflow promote settlement and including a drain at the lowest point enables the use of a sludge pump for the frequent removal of settled solids. Finally, adequate provision for the removal and de-watering of accumulated sludge should be made based on average SS concentrations and removal efficiency.

Mechanical filtration

Filtration can be an effective treatment strategy for aquaculture wastewater as a significant proportion of phosphorus, organic matter and BOD are associated with suspended particles (Cripps, 1991). Several commercial filters are available, with the simplest devices containing stationary screens, for example, the *Triangel Filter*[2] (Mäkinen, Lindgren and Eskelinen, 1988). Reviewing performance figures for this filter type from various studies indicates removal efficiencies ranging from 27-40% for TN, 18-83% for TP, 4-90% for SS and 17-85% for BOD (see NCC, 1990 and Hennessy, 1991). *Triangel Filter* performance may be adversely affected by the breakup of solids prior to filtration or algal growths clogging the screens (NCC, 1990).

Self-cleaning axial flow rotary screens such as the *Unik Filter*[3] offer an advantage over stationary screens where fouling is a problem (Cripps and Kelly, 1996). Treating wastewater from Atlantic salmon tanks using a *Unik Filter* fitted with two submersible

[2]Triangel Filters manufactured by Hydrotech AB, Industrigatan 1, 235 32 Vellinge, Sweden
[3]Unik Filters manufactured by Unik Filtersystem A/S, N-5200 Os, Norway

microsieves, with mesh sizes of 60 μm and either 150 or 350 μm, resulted in mean removal rates for SS, TP and TN of 68, 63 and 17 per cent, respectively. TAN concentrations were unchanged (Bergheim, Sanni, Indrevik and Hølland, 1993). The soluble nature of TAN makes removal by filtration impractical, therefore, the removal performance of mechanical filters for other dissolved complexes may also be limited. Despite this, *Unik Filters* have been used extensively in Norway to treat wastewater from smolt units, and several hatcheries were reported to filter both influent and effluent water (Bergheim, Tyvold and Seymour, 1991).

Radial flow rotary screens are drum shaped and wastewater entering the open end is filtered through a cylindrical mesh screen. Matter retained on the inner surface of the drum is then washed intermittently into a trough and the resulting backwash water flows to a settlement chamber for de-watering (Cripps and Kelly, 1996). *Hydrotech*[4] produce a range of commercially available drum-shaped filters that have found widespread application and studies have demonstrated that this type of filter, fitted with a 60 μm pore size screen, may remove between 67-97% of SS, 21-86% of TP and 4-89% of TN (Ulgenes, 1992b, cited in Cripps and Kelly, 1996). Other types of mechanical filter for treating aquaculture wastewater include chain type rotary screens and vibratory screens (Cripps, 1991), although these are not commonly used. Belt filters have been installed at a number of aquaculture sites in the past 3-4 years (Muir, personal communication 2001), but performance data are absent from the literature.

The potential of mechanical filtration as a treatment option depends largely on the proportion of waste compounds contained in the particulate fraction, and the characteristics of the particles. Cripps (1995) describes the distribution of particles in wastewater from a salmonid hatchery. Analysis of particles in the wastewater divided into seven size fractions using membranes with pore sizes of 200, 100, 85, 65, 47, 25 and 5 μm, revealed that no

[4]Hydrotech Drumfilters manufactured by Hydrotech AB, Industrigatan 1, 235 32 Vellinge, Sweden

single size group contained a disproportionate concentration of phosphorus, nitrogen or particulate volume or number. This suggested that mechanical filters fitted with 60 μm screens may retain only approximately half the particulate matter discharged. However, using screens with a smaller pore size would restrict the volume of wastewater filtered for equivalent surface areas.

The performance of mechanical filters also depends on the configuration of screens employed and wastewater characteristics; Cripps and Kelly (1996) recommend *in situ* testing during the commissioning phase to select the most appropriate screen configuration. Varying operating conditions, wastewater characteristics and screen configurations used in studies reviewed here makes drawing comparisons between different filter makes and models unwise. This review suggests that mechanical filters are effective for the removal of SS, BOD and TP, however, performance is limited by particle breakup prior to filtration, clogging and an inability to remove dissolved substances and fine particles. Mechanical filters usually produce small volumes of backwash water in relation to the wastewater volume treated (Seymour and Bergheim, 1991; Cripps, 1994), and de-watering backwash water to produce a manageable volume of concentrated sludge requires investment in equipment and constant monitoring.

Bergheim, Rønhovde and Mundal (1997) reported that the average volume of backwash water required by 4 *Hydrotech* drumfilters receiving 30-35 m^3 min^{-1} of wastewater from a Norwegian smolt farm equated to less than 1 l m^{-3} i.e. <0.1% of treated water. The sludge was made more manageable by filtering the backwash water through a single *Hydrotech* filter, and the resulting sludge settled using a conical tank. It was subsequently decanted to a second tank where lime was used to raise the pH to 12, eliminating odour, arresting putrefaction and killing pathogens. Stabilised sludge was transferred to a third tank for 2-4 weeks storage before being removed by local farmers, mixed with livestock manure and stored prior to land application. The proportion of solids, nitrogen and phosphorus in wastewater discharged from the smolt unit and applied to the land was 50, 20 and 50 per cent, respectively, suggesting that efficiency could be improved. The negligible potassium content of the sludge would limit its value as a fertilizer. Fixed costs associated with developing the system were considered high, however, this was because drumfilters with additional capacity to meet future farm expansion were installed. Annual costs amounted to 5% of total production costs, although this was expected to decrease as the farm expanded.

1.5.4. Wastewater treatment at cage farms

The potential of sludge collection systems suspended below fish cages has been reported by a number of authors (Beveridge, 1996; Costa-Pierce, 1996; Cripps and Kelly, 1996). Garmann et al. (1984; cited in Seymour and Bergheim, 1991) placed a trap below rainbow trout cages; the trap had an effective horizontal surface area of 50 m^2, equating to approximately 40% of the surface area of the cages. Over a four-month period the trap collected 75-80% of particles released from the cages, appearing to represent an effective treatment strategy; however, limitations had been identified. The sludge collection system worked well in lakes and sheltered coastal areas, but was vulnerable in more exposed conditions (Cripps and Kelly, 1996). Kasper et al. (1988; cited in Seymour and Bergheim, 1991) estimated that 10% of phosphorus and 65% of nitrogen in feed pass into the receiving aquatic environment as soluble compounds, and so these fractions are not removed by particle collection.

Dispersal of waste that settles under cage facilities has been attempted using mechanical mixers (Beveridge, 1996). Used at a cage site to re-suspend and disperse settled material this method resulted in a reduction of 60-75% of the 40 cm thick layer of accumulated waste (Braaten, Ervik and Boje, 1983; cited in Beveridge, 1996). This may be a beneficial management strategy, but although dispersal can limit localised impacts, the problems may only have been relocated. An alternative strategy is to fallow cage sites, this allows recovery of impacted benthic environments and may assist in limiting the proliferation of disease organisms (Beveridge, 1996). However, recovery of sediments may take several years and the limited availability of suitable sites for cage aquaculture restricts the practicality of this option.

The development of offshore sea cages at sites exposed to strong waves and currents appears promising as these conditions promote water exchange and the dispersion and dilution of waste products. An increased distance from other farms is also attractive, reducing the risk of local nutrient enrichment and disease transfer between sites. However, the development of offshore cage farms is restricted by the availability of suitable technologies to cope with extreme hydrological conditions, and by the practical constraints of servicing exposed locations.

1.5.5. Water reuse in aquaculture

Water reuse in aquaculture depends on the use of advanced treatment techniques, such as biofiltration, ultraviolet steralisation and ozonation, so that the culture water may be recycled (Muir, 1994). Frequently, water reuse is employed in an attempt to retain heat energy, enabling the maintenance of the optimal culture temperature, leading to increased growth rates and improved feed conversion ratios (Seymour and Bergheim, 1991). Farms employing water reuse are effective at limiting pollution, although several constraints to managing water quality and disposing of wastewater produced in water reuse systems have been identified.

The effectiveness of wastewater treatment in these systems is constrained by a number of factors; the accumulation of TAN, phosphorus and small particles in the culture water represent particular areas of concern. Chen, Timmons, Aneshansley and Bisogni (1993) recorded the size profile of particles in water reuse facilities and found that over

95% of particles had a diameter below 20 μm and that this fraction accounted for 40-70% of SS by weight. The implication is that conventional mechanical filters will be unable to remove a large proportion of these smaller particles, leading to the deterioration of water quality. Employing advanced wastewater treatment techniques such as contact media filters may facilitate the removal of small particles, but may lead to increased management demands associated with maintaining and monitoring the treatment unit. Irrespective of which treatment strategy is employed, removal of solid waste from the culture water will produce sludge that will require the provision of de-watering and storage facilities and ultimately need disposal. On-site sludge management options are limited because nutrients could leach to local watercourses; sludge containing pathogens would represent a health hazard; odour or flies could cause a nuisance; the cost and physical area required to establish de-watering and drying beds may be prohibitive. Sludge disposal is frequently contracted out, however, this results in risks associated with nutrient and pathogen management being transferred off-site. Furthermore, the large capital investment required to establish aquaculture facilities employing water reuse has restricted its adoption.

1.6. Future directions

Concern regarding the impact of wastewater from aquaculture, including nutrient enrichment and alterations in the biotic community in the receiving water, has been addressed in several fundamental ways (NCC, 1990). Effluent loading rates have been significantly reduced through careful feed management, the formulation of energy-dense and low pollution diets, and through more efficient sludge collectors and mechanical filters. In addition, fish farm managers have developed *best management practices*, e.g. the environmental management system proposed by Gavine, Rennis and Windmill (1996).

In the UK, the range of developments and initiatives mentioned above has resulted in discharge-consents for freshwater aquaculture being attained more frequently. However,

Table 1.3: Strategies for reducing waste loadings from commercial aquaculture

Management area	Strategy	Constraints	Reference
Feed formulation	- constituents selected to match nutritional demands; processing techniques used to maximise digestibility; increased energy levels limit protein catabolism	- quality feed constituents are expensive and limited in availability; excess feed lipid levels can affect product quality	Wiesmann et al. (1988); Seymour and Bergheim (1991); Cho and Bureau (1997)
Feeding strategy	- feeding regime tailored to behaviour; feed delivery automated; careful handling limits dust production	- disease can seriously restrict feeding, automation requires monitoring and investment	Kadri et al. (1991); Seymour and Bergheim (1991)
Facility layout and management	- pipes configured to limit particulate break-up by reducing length and number of bends; oxygenation to reduce water volume requiring treatment	- pipe layout constrained by physical and hydrological conditions; oxygenation increases operating costs	NCC (1990)
Modification of the receiving environment	- stream and river channels modified to promote mixing and aeration, limiting sediment deposition and compensating for depressed oxygen levels	- modifying receiving environments may create separate problems; environmental impacts may only move downstream	
Wastewater collection	- preferential collection and treatment of wastewater produced after feeding, cleaning, grading and harvesting and *in situ* sludge separation	- requires increased attention to management and facility design; culture water produced during routine operation may still require treatment	Lystad and Selvik (1991); Selvik and Lystad (1991); Schwartz and Boyd (1994)
Settlement	- settlement provides a simple approach to wastewater treatment; design parameters developed; effective at removing suspended particles and associated waste fractions	- settlement ineffective at removing soluble waste and small particles; prohibitive land area required; sludge requires regular removal and disposal	Henderson and Bromage (1988); Hennessy (1991); Allcock and Buchanan (1994); Teichert-Coddington et al. (1999)
Mechanical treatment	- screens configured to wastewater characteristics; treatment efficiency predictable and a range of commercial systems available	- frequent monitoring and maintenance required; sludge requires de-watering, storage and disposal; capital and operating costs represent a financial burden	Makinen et al. (1988); Cripps (1991); Bergheim et al. (1991); Bergheim et al. (1993); Cripps (1995); Bergheim et al. (1997)
Water reuse	- culture water treated and reused to retain heat energy; optimal conditions maintained for species cultured; facilitates aquaculture where access to water is limited	- systems require constant monitoring; sludge produced requires de-watering, storage and disposal; capital and operating costs high	Chen et al. (1993); Muir (1994)

Doughty and McPhail (1995) caution that this compliance should "be considered in context of the low sampling frequency employed: large daily variations in losses of total phosphorus and total nitrogen (of which ammonia is the largest fraction) from fish farms have been noted". In many situations, nutrient discharges, particularly nitrogen and phosphorus, from freshwater aquaculture are unregulated and represent a potential cause for concern. Nutrient discharges from industry, including agriculture and aquaculture, are increasingly controlled by guidelines and legislation, with non-compliance leading to fines and the imposition of pollution taxes.

Management strategies with high potential to minimise aquaculture waste discharges can be identified by adopting a systematic approach to assessing each element of the production process which influences the efficiency of feed utilisation and/or the effectiveness of managing the waste produced. One such approach is *Life cycle analysis*, which has been widely applied in developing environmental impact management strategies for industrial processes. As part of an integrated approach to aquaculture wastewater management, Alanärä, Bergheim, Cripps, Eliassen and Kristiansen (1994) applied a similar assessment to Atlantic salmon culture. The proposed strategy included demand feeders to optimise feed utilisation, wastewater treatment using rotary filters with screen sizes tailored to particle size distribution, and sludge disposal to infiltration basins. However, constraints to current practices employed in managing aquaculture wastewater, such as those described in Table 1.3, suggest that improving, or 'fine-tuning' existing management strategies may be insufficient to meet more stringent discharge standards.

Where treatment is employed, the effectiveness of settlement ponds and mechanical filtration in removing dissolved nutrients and fine particulate matter is limited (Hennessy, 1991; Cripps and Kelly, 1996). Mechanical filters represent a significant burden upon the time of the operator as they are susceptible to shock loadings, mechanical failure, wear and tear and interruptions to the supply of electricity, and therefore require constant monitoring and regular maintenance. Settlement ponds and mechanical filters produce concentrated sludge that requires regular disposal, an activity that may represent a significant cost to the operator and may lead to the formation of "*h*ampered *e*ffluent *a*ccumulation *p*rocesses (HEAP) traps, where former point source pollution is ultimately converted into non-point source pollution" (Günther, 1997). Furthermore, even aquaculture facilities employing water reuse continue to produce backwash water and sludge requiring disposal. The absence of economically viable techniques capable of removing the dissolved nitrogen and phosphorus from the large volumes of wastewater that characterise discharges from aquaculture represents a significant constraint to achieving stricter discharge standards.

In addition to meeting the requirements of regulatory bodies, aquaculture producers must be aware of concerns expressed by the public regarding the origins of products they consume and their environmental credentials. Wasteful producers could find themselves at a disadvantage while operators employing environmentally friendly options may be able to obtain a premium for their products. Young, Brugere and Muir (1999) suggest that market perceptions and environmental attributes will increasingly drive future growth within the aquaculture industry. A proactive approach to managing problems associated with discharging aquaculture wastewater could therefore confer a significant advantage on operators.

The prospect of stricter discharge consents, pollution taxes and increased preference of consumers for environmentally friendly goods justifies research into improved treatment methods. In addition to public concern regarding the appropriation of environmental goods and services by aquaculture, there is a growing recognition that nutrients entrained in aquaculture wastewater represent a potentially valuable resource. In developing countries where discharge standards have yet to be implemented for rapidly expanding aquaculture sectors, there may be little incentive to treat aquaculture wastewater, even with simple strategies such as settlement, despite the risk of negative environmental impacts. However, where tangible benefits from increased production are achieved through the productive reuse of aquaculture wastewater, operators in developing countries would be more willing to adopt such strategies. Treatment options that facilitate the reuse of wastewater to culture aquatic species could be attractive to both consumers and operators in developed and developing countries. This rationale forms the basis from which *horizontally integrated aquaculture* has developed.

Chapter Two

Horizontally integrated aquaculture systems

2.1. Introduction

This chapter describes the origins of using waste-flows as production enhancing inputs for aquaculture and develops the concept of generating valuable products whilst facilitating processes to improve water quality. A definition for the practice of *horizontally integrated aquaculture* is proposed and the potential of employing unexploited resources and assets from primary aquaculture activities to integrate aquatic organisms in one or more secondary production units is described. Such a strategy would capitalise on existing knowledge, management structures, infrastructure and markets. Aquaculture practices that fit the definition are characterised, and experiences of constraints and benefits with each are described. It is shown that despite apparent technical viability, extensive research and trials concerning horizontally integrated aquaculture, few commercial enterprises have been established. Therefore, a systems perspective is proposed to assess the potential of horizontally integrated aquaculture in commercial settings.

2.2. Productive reuse and treatment of waste through aquaculture

Limited access to nutrient sources and the expense of inorganic fertiliser and feed has led to food processing by-products, animal manure and human waste being used to stimulate production in fishponds (Edwards, 1980; Little and Muir, 1987; Wohlfarth and Hulata 1987; De Pauw and Salomoni, 1991; Lin, Teichert-Coddington, Green and Veverica, 1997). Employing unexploited resources derived from within farming systems or from the immediate area to intensify production in fishponds has been termed *integrated aquaculture* (Little and Muir, 1987). Examples of these practices are described in the remainder of this section.

2.2.1. Manure

The poor resource base of most small-scale farms in developing countries means that unexploited nutrient sources e.g. crop by-products, terrestrial weeds, aquatic plants and manure represent important production enhancing inputs to fishponds (Edwards, Demaine, Innes-Taylor and Turongruang, 1996). Little and Edwards (1999) reviewed the alternative strategies that have evolved to integrate the production of livestock and aquaculture. Manure from cattle, buffalo, sheep, pigs and poultry has been employed to enhance production in aquaculture systems (Teichert-Coddington, Behrends and Smitherman, 1990; Edwards, Pacharaprakiti and Yomjinda, 1994; Maclean, Brown, Ang and Jauncey, 1994; Garg, 1996). Zhu, Yang, Wan, Hua and Mathias (1990) studied the response of ponds stocked with silver carp (*Hypophthalmichthys molitrix*), bighead carp (*Aristichthys nobilis*), common carp and crucian carp (*Carassius carassius*) to various applications of fermented pig manure. Manure rates of 31-48 kg ha^{-1} d^{-1} produced a net fish yield of 10.2 kg ha^{-1} d^{-1}; control ponds receiving equivalent inputs of inorganic fertiliser produced 4.3 kg ha^{-1} d^{-1}. These authors proposed that higher fish production observed in manured ponds, as compared to ponds receiving equivalent nutrient inputs in the form of inorganic fertiliser, was attributable to increased heterotrophic production. This demonstrates that both autotrophic and heterotrophic pathways are important in optimising fish production in ponds.

The integrated rearing of poultry above fishponds has been widely advocated (Johnson and Avault, 1982; Msiska and Cantrell, 1985; Knud-Hansen, Batterson and McNabb, 1993; Christensen, 1993; Christensen, 1994; Njoku and Ejiogu, 1999). It has been practised in Asia for several centuries, and has been shown to confer several advantages. Fermentation, evaporation and coagulation are reduced, preserving the nutritional quality of faeces; feed residues are consumed directly by fish; costs associated with collecting, storing and transporting manure are eliminated, and as compared with separate culture units, less land is required (Barash, Plavnik and Moav, 1982). These authors found that production in fishponds receiving direct inputs of excreta from ducks was equivalent to that in ponds receiving equivalent inputs of dry poultry manure and supplementary feed. However, despite the apparent benefits of integrating poultry farming with aquaculture, constraints have been identified (Little, 1995). These include limited access to inputs e.g. capital, labour and feed, and lack of skills amongst potential operators to successfully manage such systems. Small-scale farmers integrating the production of fish and ducks may also encounter problems obtaining inputs and selling products, especially if competing with large-scale producers (Edwards, 1998).

Within the dike-pond farming system that evolved over several hundred years in the Zhujiang Delta, Guangdong Province, South China, excreta from pigs was used extensively to fertilise ponds (Ruddle and Christensen, 1993). Ruddle and Zhong (1988) suggested that in many cases, manure production was the primary reason for keeping pigs. However, in recent years the extent of land under dike-pond cultivation has declined and farmers have removed dikes to increase available pond area for stocking intensive market-orientated monocultures, producing high-value products such as eels, prawns and terrapins destined for export markets (Wong Chor Yee, 1999). Factors leading to these changes have been the prospect of greater economic returns, and the decline in the value of silk, an important output of the traditional dike-pond system.

Concern has also been expressed regarding the role that pigs may play in facilitating contact between avian and human influenza viruses, possibly leading to influenza pandemics. It has therefore been recommended that co-location of pigs and poultry in integrated farming systems should be avoided, and that the promotion of this management strategy should be discontinued (Scholtissek and Naylor 1988). Replying to this commentary, Edwards, Kwei Lin, Macintosh, Leong Wee, Little and Innes-Taylor (1988) stated that pigs and ducks had been raised together in traditional Asian farming systems for centuries without apparent risks, and that promoting integrated aquaculture would be unlikely to increase the risk posed by new influenza strains. Furthermore, problems with management and marketing had constrained the integration of more than one species of livestock or poultry making the dangers of wider multi-species transfers less likely. In response, Naylor and Scholtissek (1988) called for a proposal from the Consultative Group on International Agricultural Research advocating "fully integrated crop/livestock/fish farming" to be qualified. Skladany (1996) chronicles the evolution of the resulting debate and highlights the potential of following the research, publications, projects and controversies of innovators to inform sociologists studying aquaculture development. The discourse described is based largely on a hypothetical risk, however, Muratori, de Oliveira, Ribeiro, Leite, Costa and da Silva (2000) showed that 11.2% of fish sampled (n = 540) from ponds fertilised with pig manure harboured *Edwardsiella tarda*. This causes tropical fish septicaemic disease and is considered an emergent foodborne health hazard for humans. Therefore, risks associated with using animal manure in aquaculture require consideration, and precautionary measures such as pre-treatment to kill pathogens and depuration may be advisable.

2.2.2. Human waste

A comprehensive review of domestic waste-based aquaculture is presented elsewhere (Edwards, 1992). Practices from 21 countries were reviewed and classified according to the approach used to introduce faecal material to the aquaculture system i.e. overhung latrines, cartage of urban nightsoil, faecally polluted surface water and sewage water.

Informal wastewater aquaculture systems receiving faecally polluted surface water were described from five Asian countries: Indonesia, Japan, China, Sri Lanka and Taiwan. Surface water containing human waste is commonly used in Indonesia to culture fish; around Bogor, grey water from drains that also received human excreta was used to culture tilapia. Indonesian aquaculturists developed the culture of common carp in bamboo cages positioned in canals containing faecally polluted surface water. Traditionally, no supplementary feed was given to carp cultured in these cages as benthic organisms e.g. chirnomids, proliferated in the nutrient rich water. However, intensification has resulted in supplementary feeds being applied to enhance production.

In addition to informal practices in Asia where aquaculture is undertaken in faecally polluted surface water, Edwards (1992) described the deliberate use of sewage water in formal systems to produce fish in four Asian countries: China, Indonesia, Israel and India. Kundu (1994) described the origins and current status of wastewater aquaculture in peri-urban Calcutta, India. Ponds managed for wastewater aquaculture covered an area of 3,000 ha, consisting of ~154 fisheries or *bheries*. The extent of the fisheries had developed to receive 550,000 m^3 d^{-1} of untreated Calcutta wastewater. Wastewater from the city was pumped into a series of secondary canals that conveyed it to the reuse system that, in addition to facilitating wastewater aquaculture, supported an area of wastewater agriculture. Production from the Calcutta system was estimated at 13,000 t y^{-1} of Indian major carp and tilapia; harvested fish sold in nearby urban markets represented ~16% of total fish sales in the municipality (Mara, Edwards, Clark and Mills, 1993).

Wastewater aquaculture has been extensively practiced in the Thanh Tri District of Vietnam. During the 1960's a canal was constructed to transport wastewater away from urban areas; fishponds were subsequently constructed adjacent to the canal and wastewater from the canal pumped into them. The reticulated nature of the canal network means that fishponds further from the urban centre do not receive nutrient rich wastewater. However, water entering these ponds is rich in biota. Fish species cultured include silver carp, rohu (*Labeo rohita*) and tilapia and yields ranged from 5-8 t ha^{-1} y^{-1}. Production from fishponds receiving wastewater represented an important source of relatively cheap food for poor communities; the annual production of 4,500 t was equivalent to 40-50% of the total supply of fish to Hanoi (Edwards, 1996).

Aquaculture can play an important role in converting unexploited nutrient resources in wastewater to valuable products. However, concerns about the capacity of aquatic ecosystems to attenuate pathogen numbers, compensate for biological oxygen demand and assimilate nutrients has favoured the use of conventional treatment ponds and lagoons to treat domestic, agricultural and industrial wastewater (Mara, 1997). Based on conventional treatment approaches, a rational design had been proposed by Mara et al. (1993) predicting that following retention in an anaerobic lagoon for 1 day and a facultative lagoon for 5 days, discharged wastewater may be used in fishponds to produce fish that are microbiologically safe for human consumption. The authors estimated that fish production in treated wastewater would average 13 t ha^{-1} y^{-1} and a financial assessment of the proposed system indicated commercial viability.

Despite the apparent benefits of traditional wastewater aquaculture practices and the potential of integrating aquaculture with formal lagoon based treatment facilities, research has tended to focus on the engineering of experimental and pilot scale wastewater aquaculture systems with the aim of maximising production (Edwards, 1992). Edwards and Sinchumpasak (1981) described culturing tilapia (*O. niloticus*) fed with microalgae from high rate stabilisation ponds used to treat sewage. Following a retention period of three days, wastewater from the 200 m^3 stabilisation pond, with a mean phytoplankton concentration of 94 mg l^{-1}, was transferred to 4 m^3 concrete fishponds. Edwards, Sinchumpasak and Tabucanon (1981b) estimated that tilapia yields from this system could approach 20 t ha^{-1} y^{-1}, although production was dependent on large inputs of phytoplankton from the stabilisation pond. Phytoplankton production in the stabilisation pond was 15.7 g m^{-2} d^{-1}, from which an annual production rate of 57 t ha^{-1} was extrapolated. However, despite the promise of this system, samples from fishponds contravened WHO standards for faecal coliform concentrations in wastewater for use in aquaculture or irrigating crops to be eaten after cooking (Edwards, Sinchumpasak, Labhsetwar and Tabucanon, 1981a). The experience of these authors demonstrated that even where highly productive waste-reuse systems are developed, other factors such as possible health risks from pathogens and pollutants, and consumer acceptance, must be considered.

2.2.3. Intermediaries in integrated aquaculture

When undertaking aquaculture in domestic wastewater, the presence of faecal coliforms and other pathogens e.g. protazoa, trematodes, helminths and viruses, and the threat of contamination with heavy metals, detergents and carcinogens severely restrict consumer acceptance. Depuration offers a potential solution, but requires that the cultured species be held in clean water for several days, which may be costly and difficult to manage and control. The potential of culturing organisms using waste and then feeding these as intermediate products to species destined for human consumption has also received attention (Edwards, 1990). Such intermediaries could prevent the transmission of disease agents and allay fears of potential consumers.

Numerous species have been investigated as intermediaries in integrated farming systems e.g. microalgae, aquatic macrophytes, zooplankton, invertebrates and fish (Groeneweg and Schluter, 1981; Schluter and Groeneweg, 1981; Gnudi, Caputo and Salomoni, 1991). In addition to assimilating nutrients, organisms produced in integrated systems may contribute significantly to the nutrition of animals to which they are fed. Chironomid larvae cultured using waste organic matter i.e. palm oil mill effluent, were shown to possess high levels of essential amino acids and polyunsaturated fatty acids as compared with larvae grown in algae cultures (Habib, Yusoff, Phang, Ang and Mohamed, 1997). The cultivation of chironomid larvae in Hong Kong using chicken manure was reported to have been an important source of live-feed for aquarium fish and fry of carnivorous species e.g. *Clarias fuscus* and *Ophiocephalus* spp. (Shaw and Mark, 1980).

Turner, Sibbald and Hemens (1986a) investigated the possibility of culturing tilapia in marine ponds fertilised with domestic wastewater. It was proposed that fish biomass produced could be used as a protein supplement in diets for shrimp (Turner, Sibbald and Hemens, 1986b). Despite appearing relatively inefficient, with 10 t of fish biomass being converted to 1.05 t of shrimp, the economics of the system were favourable, and although wastewater entering the ponds contained *Escherichia coli*, none were detected in the flesh of either tilapia or prawns. Heavy metal accumulation was not significant and levels of chlorinated hydrocarbons were within permissible levels (Turner, Sibbald and Hemens, 1986c).

2.3. Opportunities for horizontally integrated aquaculture

The culture of aquatic species is accompanied by the generation of waste products (Beveridge et al., 1997). Depending on the perspective from which these waste products are viewed, they could represent a constraint to sustainability, degrading the external environment and leading to negative feedback mechanisms. However, they may also represent an opportunity, as an unexploited resource for input to integrated production systems, assimilating waste nutrients and reducing discharges to the receiving environment. Furthermore, products from the integrated culture unit could be marketed to generate additional income. A wider range of benefits may also be conferred on the environment, stakeholders and operators of aquaculture facilities. These aspects constitute the central tenet underpinning horizontally integrated aquaculture.

Reducing waste discharges through horizontal integration will contribute to environmental protection and reduce the risk of negative feedback mechanisms, which could disrupt the supply of environmental goods and services upon which commercial aquaculture ultimately depends (Beveridge et al., 1997). Assimilating nutrients into marketable products, as opposed to removing them using conventional treatment techniques, avoids the production of sludge, the inappropriate disposal of which could transform a point source of pollution to a diffuse source (Gunther, 1997). The assimilation of nutrients using horizontally integrated aquaculture will reduce the loss of non-renewable resources e.g. phosphorus, in the unidirectional flow of matter entrained within the hydrological cycle.

Technical innovation and husbandry techniques e.g. low pollution feed, automated and demand-driven feed delivery units, reduced stocking densities, vaccines and wastewater treatment, may be employed to reduce the demand for environmental goods and services that aquaculture places upon the environment (Beveridge et al., 1997). However, adoption of modified management strategies is usually limited due to financial demands. Maddox and Kingsley (1989) state: "if any new activity such as waste treatment, does not reduce costs and/or increase income, it is unlikely to gain acceptance". The cultivation of valuable species in integrated units may address this constraint.

Culturing aquatic species in wastewater from aquaculture operations has been proposed previously and the associated treatment effect, productivity and technical demands, based on results from laboratory investigations and pilot-scale studies, have been described. Costa-Pierce (1996) describes using wastewater from shrimp ponds and cage facilities to rear bivalves and diverting waste from settling ponds and cage collectors to assist in creating wetlands as *ecological aquaculture*. Buschmann (1996) refers to cultivating seaweed in wastewater from commercial offshore aquaculture facilities as *integrated farming*, although this latter phrase has close associations with integrating agricultural activities, especially livestock production, with aquaculture. However, no formal definition has been derived to describe adequately the wide array of systems that have evolved from traditional practices and developed through scientific investigation. In the following section, existing descriptions of key components of horizontally integrated aquaculture are presented and used to formulate a tentative definition.

2.4. Defining horizontally integrated aquaculture

As outlined in Section 2.1, the practice of employing unexploited resources to intensify production in fishponds has been termed *integrated aquaculture* (Little and Muir, 1987). However, the definition proposed here aims specifically to describe the use of unexploited resources derived from aquaculture activities as production enhancing inputs to secondary aquaculture activities. *Aquaculture* is defined by FAO (1995) as "the farming of aquatic organisms, including fish, molluscs, crustaceans and aquatic plants. Farming implies some form of intervention in the rearing process to enhance production, such as the regular stocking, feeding, protection from predators, etc. Farming also implies individual or corporate ownership of the stock being cultivated. For statistical purposes, aquatic organisms that are harvested by an individual or corporate body that has owned them throughout their rearing period contribute to aquaculture, whilst aquatic organisms that are exploitable by the public as common property resources, with or without appropriate licenses, are the harvest of fisheries". Defining aquaculture is important for two reasons: firstly it identifies those activities from which unexploited resources may be derived and secondly it defines those culture practices that may be considered for integration. The FAO definition also appears to exclude groups other than individuals and corporate bodies from engaging in aquaculture, however, households, families, communities, cooperatives and governments may have ownership over cultured stock. This is of particular importance where horizontally integrated aquaculture is proposed as a suitable mechanism for empowering local communities, permitting them renewed access to productive coastal resources, but where equitable management is likely to depend upon community-based organisations that retain ownership or at least the right to exploit aquatic plants and animals cultured in integrated systems. Implications of allowing local communities access to extensive horizontally integrated aquaculture systems are discussed further in Chapter 4.

From a business perspective, Thompson (1993b) states that "horizontal integration occurs when a firm acquires or merges with a major competitor, or at least another firm operating at the same stage in the added value chain". However, *horizontal integration* is used here to describe using unexploited resources and assets from existing aquaculture operations to culture products at a similar stage in the added value chain. It implies parity between the aquaculture activities undertaken, although in most cases, horizontally integrated systems will consist of a primary activity receiving the majority of external inputs and generating a substantial part of the output value. Therefore, *horizontally integrated aquaculture* may be defined as:

> "the use of unexploited resources derived from primary aquaculture activities to facilitate the integration of secondary aquaculture practices."

A distinction may also be drawn between culture practices that are undertaken in conjunction with an aquaculture activity primarily to supply inputs (*vertical integration*) and culture systems that develop to exploit surplus resources (*horizontal integration*). However, situations may arise where products from horizontally integrated aquaculture systems facilitate vertical integration.

2.5. Developing a systems perspective

Research studies commonly focus on technical aspects of horizontally integrated aquaculture such as wastewater treatment performance, production rates, management demands and design parameters. However, operators of commercial aquaculture units deciding whether to adopt such strategies will need to consider a much wider range of issues[1]. Financial viability is likely to be foremost, and will depend on factors such as input costs, market demand for integrated products and the cost to benefit ratio of alternative wastewater treatment strategies. Regulatory controls may also hold sway, for example, planning restrictions and legislation against co-culturing species or outlawing the introduction of species suitable for integration may constrain development. Consumer acceptance and perceived public health risks will be influential, as will potential benefits such as increased environmental protection, enhanced resource-use efficiency and reduced stakeholder conflicts. Such factors also interact, operate at a range of scales and change with time, resulting in a complex setting in which operators must make decisions.

The "systems approach to management" for aquaculture proposed by Muir (1996) provides an overview of the essential elements requiring consideration and is used to guide the assessment of horizontally integrated aquaculture in this thesis. In the remainder of this chapter, to build on the definition of horizontally integrated aquaculture presented, studies investigating the practical application of this strategy are reviewed. A bioeconomic model is developed in Chapter 3 to further investigate technical and financial aspects of horizontal integration, and a case study concerning the integration of a constructed wetland and trout fishery with a smolt unit is presented. Chapters 4 and 5 present additional case studies developed to test the generality of the model, whilst in Chapter 6 the results of a Delphi investigation elicit constraints and opportunities to horizontally integrated aquaculture from stakeholders. Chapter 7 presents a synthesis of findings from this work;

prominent systems features are described and areas requiring future investigation, to develop the systems approach more fully, identified. However, before progressing further, the boundaries of the systems to be investigated must be delineated.

The FAO definition of aquaculture given earlier provides the basis for delineating both primary 'waste generating' and secondary 'waste processing' aquaculture activities. Horizontally integrated aquaculture therefore excludes using waste from aquaculture to enhance production in terrestrial agriculture. Furthermore, the FAO definition distinguishes between aquaculture and fisheries by ownership. Thus excluding the enhancement of capture fisheries from this discussion. However, employing unexploited resources to facilitate ranching may be included. Prospects for ranching are discussed in Section 2.5.6.

In addition to identifying farming practices and management structures that constitute aquaculture, assessment from a systems perspective requires the definition to be expanded. Spedding (1988) describes a *system* as "a group of interacting components, operating together for a common purpose, capable of reacting as a whole to external stimuli: it is unaffected directly by its own outputs and has a specified boundary based on the inclusion of all significant feedback". Muir (1996) presented a summary of components, or systems features, that interact within aquaculture systems; these included ethical, cultural, socio-political, economic, climatic, ago-ecological, production, farming, resource-use, mass-balance, ecological and biological factors. Furthermore, Muir (1996) described *aquaculture systems* as:

> "the interdependent elements of environment, physical structure, rearing process, resource use and socio-economic context which form the entity from which farmed aquatic organisms are produced".

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[1]An assessment of influential constraints and benefits that stakeholders, including operators of commercial aquaculture units, associated with horizontally integrated aquaculture is presented in Chapter 6

Within this definition, primary and secondary aquaculture activities that constitute horizontal integration may be considered sub-systems.

Having defined which farming activities constitute horizontally integrated aquaculture, the following section presents a summary of management strategies that meet the criteria; this may be considered the first step in developing a systems perspective. Subsequent steps include exploring the relationships between intra-system elements, leading to context and explanation, assessing functions and controls that govern systems behaviour, and describing implications and impacts of outcomes (Muir, 1996). The synthesis of these elements leads to the comprehension of a complete systems context, allowing areas for prioritisation and choice to be identified, and associated outcomes predicted.

2.6. Management strategies facilitating horizontal integration

Following the definition in the previous section, Figure 2.1 presents a framework in which culture systems that fulfil the criteria for horizontally integrated aquaculture are summarised. Primary aquaculture activities are divided into four types based upon the degree of connectivity existing between the culture system and the environment: static, water reuse, flow-through and open. Specific culture strategies that may be considered for integration with each of the primary aquaculture activities are presented and experiences with each are reviewed below. The framework in Figure 2.1 includes an indication of distance from the primary aquaculture activity and a notional boundary to the system in an attempt to facilitate the development of a spatial context in which to consider horizontally integrated aquaculture.

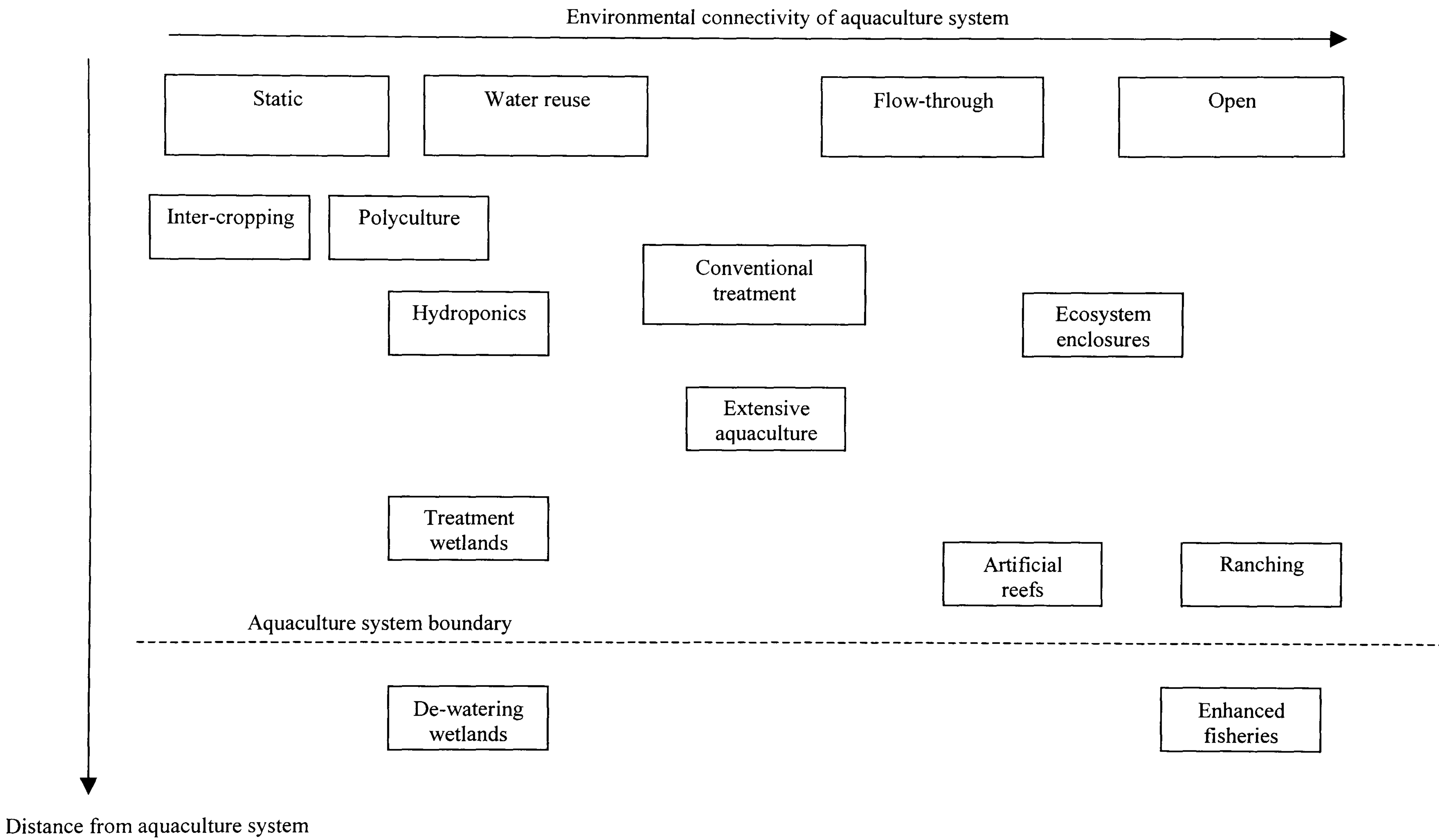

Figure 2.1: Aquaculture systems facilitating horizontal integration

2.6.1. Polyculture

Polyculture, the practice of stocking multiple species exhibiting different feeding preferences to exploit all niches within the pond, may be considered the most basic strategy facilitating horizontally integrated aquaculture. Polyculture practices employing indigenous fish species have developed predominantly in China and India, to optimise yields in static ponds where the production of diverse biota may be stimulated through introducing fertiliser, particularly organic waste. Traditional Chinese approaches are based on stocking 4-5 species with diverse feeding behaviours to exploit the various nutritional resources found in managed ponds. Typically these include species that feed preferentially on phytoplankton (silver carp), zooplankton (bighead carp), macrophytes (grass carp, *Ctenopharyngodon idella*), detritus (common carp and mud carp, *Cirrhina molitorella*) and snails (black carp, *Mylopharyngodon piceus*) (Ruddle and Zhong, 1988). There may also be synergistic effects of stocking different species together, for example, common carp consuming silver carp faeces, rich in partially digested phytoplankton, may attain a higher yield than in monoculture, whilst nutrient rich sediments put into suspension by foraging common carp suppress the development of filamentous algae and higher plants, and stimulate the production of phytoplankton, upon which the silver carp feed (Milstein, 1992). This author also noted that polyculture may contribute to improved water quality, with appropriate species used to regulate phytoplankton and macrophyte populations, contributing to more stable dissolved oxygen regimes, whilst benthic feeders prevent the accumulation of organic sediments, reducing oxygen consumption and ammonia production. However, despite the potential benefits, poorly managed or inappropriate polyculture practices could result in unstable phytoplankton populations, leading to oxygen depletion, blooms of undesirable algae or competition amongst species for food.

Production rates of 5-10 t ha^{-1} y^{-1} have been reported for ponds managed for polyculture in China (Ruddle and Zhong, 1988; Korn, 1996). However, accounts of recent initiatives, where bream (*Megalobrama bramula*) and tilapia (*Oreochromis* spp.) were stocked with 6 carp species to enhance production, reported yields of 7 t ha^{-1} y^{-1} (Ruddle and Christensen, 1993); this is still within the range given for traditional carp polycultures.

When compared to the Chinese system, the greater plasticity in feeding behaviour exhibited by fish used in Indian polycultures appears to have constrained productivity (Little and Muir, 1987). The feeding preferences of fish in the Chinese system tend to mean that particular species forage in different spatial zones, minimising competition for nutritional resources, facilitating optimal foraging and limiting energy expenditure during behavioural interactions. Furthermore, the diversity of fish species stocked in polycultures may introduce a degree of resilience to disease and market pressures. However, these potential benefits must be considered with respect to possible constraints, including restricted access to the required quantity and species of fry for stocking, extra handling costs during harvest, and poor market value and acceptability of some species (Little and Muir, 1987).

In geographical locations outside Asia, traditional polyculture practices have not developed, possibly due to the limited selection of indigenous fish that are cultured. However, in a number of situations, recent innovations have included culturing chinook salmon (*Oncorhynchus tshawytscha*) with Pacific oysters (Jones and Iwama, 1991), shrimp (*P. vannamei*) with clams (*Mercenaria mercenaria*) and oysters (*Crassostrea virginica*) (Hopkins, Hamilton, Sandifer and Browdy, 1993), and shimp (*Penaeus chinensis*) with hybrid tilapia (Wang, Li, Dong, Wang and Tian, 1998). Culturing bivalves in static water ponds may contribute to maintaining water quality, particularly in shrimp culture ponds where the inefficient feeding of shrimp frequently results in dense phytoplankton blooms.

Hopkins et al. (1993) found that DO concentrations in polyculture ponds were consistently higher than in a control pond stocked only with shrimp. Furthermore, shrimp production in polyculture and control ponds was similar at 10.6 and 11.5 t ha^{-1}, respectively; however, overall production in polyculture ponds, including bivalves, was 24.7 t ha^{-1}. High mortality rates for clams were associated with seeding juveniles directly into pond sediments; in contrast, the survival of oysters supported above the bottom of the pond was significantly higher, although the authors noted that the trays used might interfere with management procedures on commercial farms. Wang et al. (1998) investigated using tilapia to regulate the development of phytoplankton blooms in marine shrimp ponds and optimal stocking densities for shrimp (60,000 ha^{-1}) and tilapia (400 kg ha^{-1}) were reported. A possible problem with this strategy would be the unregulated reproduction of tilapia; regular netting could be employed to maintain the optimal biomass or a predatory species could be stocked to control the population (Little and Muir, 1987); however, the impact of a predator on shrimp culture must also be considered.

Within the definition of horizontally integrated aquaculture, aquatic plants may be considered as potential candidates for polyculture. Their capacity to assimilate inorganic nutrients also makes them potentially important as *in situ* biological filters. McCord and Loyacano (1978) reported that Chinese water chestnut (*Eleocharis dulcis*) cultivated on ponds stocked with channel catfish (*Ictalurus punctatus*), produced ~4,660 kg ha^{-1} of corms. The presence of water chestnuts was associated with significantly lower concentrations of phytoplankton, TAN and nitrate; nutrient removal during the growing season averaged 53.8, 3.4 and 18.6 mg m^{-2} d^{-1} for nitrogen, calcium and magnesium, respectively. The authors proposed that adopting this strategy could allow supplementary feed levels to be increased, enhancing fish production. However, despite its promise, possible disadvantages associated with the polyculture of macrophytes and fish include reduced light penetration into the pond restricting primary production in the water column, which could limit fish growth and restrict dissolved oxygen levels. Other problems include potential interference with routine pond management and the possibility that the macrophytes could harbour fish pathogens or their hosts.

An alternative strategy for integrating the production of plants in static water ponds has developed in association with traditional *chinampas* farming systems from Xochimilco-Chalco, Mexico, where 6-9 m long floating rafts of reed and cattail were covered with mud and used as nurseries for vegetables (Armillas, 1971). Recent studies have used rafts made from artificial materials e.g. polystyrene sheets to grow rice, flowers, vegetables, wheat and fibre producing plants. Preliminary results have demonstrated that this strategy is technically viable (Song, Ying, Wu, Jin, Zou, Zhu, Zhu, Li, and Li, 2000), however, the cost of construction materials may constrain development, and the physical presence of the planted beds may interfere with pond management activities. Fish culture in rice fields is a widespread and successful example of integrating the cultivation of aquatic plants with fish. Reviewing the evolution of this practice in Thailand, Little, Surintaraseree and Innes-Taylor (1996) provide a valuable account of stimuli affecting development, systems facilitating rice-fish culture and constraints and opportunities for wider adoption.

2.6.2. Inter-cropping

Polyculture may be used to optimise the use of space and feeding niches in aquaculture systems, however, inter-cropping facilitates the optimal temporal use of an area (Wenhua and Qingwen, 1999). The seasonal availability of water in many situations leads to activities such as fish production being suspended during the dry season, however, during this period alternative crops such as rice and vegetables may be grown in the bottom of ponds (Little and Muir, 1987). These strategies do not meet the definition of horizontally integrated aquaculture; however, valid strategies include culturing air-breathing catfish in ponds with decreasing water levels following the harvest of the main crop and maintaining and spawning broodstock in refugia prior to stocking the pond.

2.6.3. Horizontal integration in aquaculture systems employing water reuse

Adopting water reuse increases the carrying capacity of an aquaculture facility, reducing the volume of wastewater discharged and limiting environmental impacts. It also restricts the transmission of disease between farms, prevents stock escaping, increases the degree of stock control and facilitates feeding strategies that optimise feed conversion (Muir, 1996; Blancheton and Coves, 1993). Water reuse systems also conserve thermal energy in the process water, an important consideration in temperate climates where the cost of heating water is considerable. However, despite the potential benefits of water reuse in aquaculture, organic matter, inorganic nitrogen (especially TAN) and carbon dioxide tend to accumulate in the process water (Muir, 1982) and a lack of effective and economic treatment options has constrained adoption.

According to van Rijn (1996), incorporating biological processes in process water treatment, such as those that maintain water quality in fishponds, has considerable potential. However, earlier studies investigating this strategy, employing ponds and lagoons as biofilters to treat water for reuse in intensive aquaculture facilities, have not been successful. Mires and Amit (1992) employed an earthen reservoir stocked with common carp and tilapia to treat water for reuse in four concrete ponds being used to intensively culture tilapia; production in these ponds averaged 17 kg m^{-2} y^{-1}. High mortalities during the first year were attributed to the proliferation of bacteria (*Streptococcus* spp.). TAN and nitrite occasionally reached undesirable concentrations, possibly indicating that the 1:6 ratio of pond to reservoir volume was too low. However, reporting on the nitrogen dynamics in this combined intensive-extensive system, Diab, Kochba, Mires and Avnimelech (1992) conclude that although the ponds provided optimal conditions for nitrification, rapid flushing rates meant anoxic conditions in the reservoir inhibited the growth of nitrifying bacteria.

Neori, Krom, Ellner, Boyd, Popper, Rabinovitch, Davison, Dvir, Zuber, Ucko, Angel and Gordin (1996) cultivated seaweed (*Ulva lactuca*) to regulate water quality in an aquaculture system stocked with gilthead seabream (*Sparus aurata*) and employing water reuse; during the two year study, water quality parameters remained stable and within safe limits. Describing nutrient dynamics in the system, Krom, Ellner, van Rijn and Neori (1995) report that only 20-27% of nitrogen supplied as feed was present in wastewater discharged from the culture unit, reportedly less than half the nitrogen discharged from conventional marine aquaculture facilities. Despite the potential benefits, performance of the biofilter may be susceptible to variations in temperature, pH and oxygen concentrations (Dvir, van Rijn and Neori, 1999). Seaweed biomass harvested from the biofilter may be considered a valuable by-product, however, the capital required to establish the system and operating costs associated with intensive management and monitoring may constrain development.

Horizontal integration in freshwater systems employing water reuse is constrained to a large extent by the market value and demand for species that may be cultured. Freshwater bivalves and filter-feeding fish have a limited value in most countries where freshwater aquaculture is undertaken in systems employing water reuse. However, pilot-scale studies have been conducted regarding the technical viability and treatment performance of integrating the production of freshwater fish with the cultivation of aquatic plants, which may be valued for their ornamental or edible nature. Ng, Sim, Ong, Kho, Ho, Tay and Goh (1990) used giant pondweed (*Elodea densa*) to remove nitrogenous compounds from water being reused for aquaculture. Planted tanks reduced concentrations of nitrate, nitrite and TAN and removed 25% of total nitrogen (TN) from the influent water. During normal operation, turbidity also decreased. Combining the production of plants and fish represents a potential strategy for nitrogen removal, facilitating water reuse; however, the harvest of plants for sale disrupted the performance of the filter for 3-4 days

(Ng et al., 1990). Cropping plant culture tanks alternately would reduce the possibility of water quality problems. Redding, Todd and Midlen (1997) examined the effectiveness of three aquatic macrophytes, emergent watercress (*Rorippa nasturtium-aquaticum*), free-floating waterfern (*Azolla filiculoiides*) and submersed Canadian pondweed (*Elodea nuttallii*) to treat water for reuse in culturing tilapia (*O. niloticus*). Removal rates for TAN, nitrate and phosphorus were highest in tanks planted with watercress; however, rates of nitrate removal decreased throughout the study. The authors suggest periodic cropping may maintain growth rates, enhancing nutrient assimilation. Where watercress is harvested for sale on a regular basis this would stimulate nutrient uptake, whilst the revenue generated could offset any associated increase in management costs, although this would depend on the value of watercress harvested and the cost of labour employed.

Hydroponics

The hydroponic cultivation of macrophytes to treat water in aquaculture facilities employing water reuse, commonly referred to as *aquaponics*, has been proposed. Adler (1998) cultivated lettuce and basil in water being reused to culture rainbow trout; based on a feed conversion ratio of 1.2 it was estimated that 13-18 heads of lettuce could be cultivated on waste nutrients released from 1 kg of feed. Although Adler (1998) and others have demonstrated the apparent technical viability of aquaponics, constraints have been identified i.e. balancing nutrient levels in the system with the nutritional requirements of plants, preventing the accumulation of salts and controlling disease and pests of both plants and fish (van Rijn, 1996). Furthermore, waste-flows generated by mechanical and biological filters in aquaculture facilities employing water reuse require treatment. Jungersen (1991) found wastewater discharged from fish breeding facilities using water reuse contained high levels of nitrogen and phosphorus, and most micronutrients required by greenhouse crops. Based on a pilot-study, it was estimated that nitrogen discharged from a culture unit producing 100 t of eel (*Anguilla anguilla*) annually, could sustain the production of 690 t of tomatoes. Despite the potential of aquaponics in assimilating nutrients from the process water and wastewater from aquaculture systems employing water reuse into valuable by-products, this strategy does not meet the criteria for horizontally integrated aquaculture. Plants cultured in the studies described above are not truly aquatic and therefore their production does not fit with the definition of aquaculture; furthermore, producing edible and ornamental plants does not capitalise on existing management skills, infrastructure and markets: the wider objectives of horizontally integrated aquaculture. However, insights gained from studies involving aquaponics may assist in designing and managing systems facilitating horizontal integration.

2.6.4. Flow-through horizontally integrated aquaculture

Several authors have described the potential of developing horizontally integrated production units in wastewater discharged from flow-through aquaculture facilities. The most widely investigated species group for integration with marine aquaculture are macroalgae and filter-feeding bivalve molluscs, experiences with each are described below and the possibility of integrating both is discussed. Other species groups with potential for horizontal integration with both marine and freshwater aquaculture facilities are discussed in the following sections.

Macroalgae

Macroalgae has been proposed as a suitable species group for horizontally integration because they sequester dissolved nutrients, particularly TAN (Vandermeulen and Gordin, 1990; Subandar, Petrell and Harrison, 1993; Neori and Shpigel, 1999). Jiménez del Río, Ramazanov and García-Reina (1996) cultured *Ulva rigida* in wastewater from land-based seabream (*S. aurata*) culture units and results indicated that 153 m^2 of seaweed cultivation tanks would be required to assimilate discharged nitrogen whilst producing 1 t of fish. Furthermore, seaweed production, and concomitantly treatment efficiency, decreased during winter, whilst susceptibility to infections increased. Buschmann, Mora, Gómez, Böttger, Buitano, Retamales, Vergara and Gutierrez (1994) investigated using wastewater from an on-shore facility stocked with Pacific salmon (*Oncorhynchus kisutch*) and rainbow trout to cultivate the agarophytic red algae (*G. chilensis*) in raceways. The authors consider using wastewater from intensively managed salmonids to cultivate seaweed as an effective strategy in southern Chile. Seaweed production rates (48.9 kg m^{-2} y^{-1}) were reportedly higher than those recorded in traditional open systems or tank cultures in central and northern Chile and epiphytism did not constitute a major constraint; however, the agar content of the *G. chilensis* decreased during periods of high growth. Martinez and Buschmann (1996) found the agar content of *G. chilensis* cultured in aquaculture wastewater was lower than plants cultured in fresh seawater. These findings may have serious consequences for the financial viability of aquaculture operations that integrate the culture of fish and macroalgae as the value of seaweed is largely dependent on agar content (Anon, 1996). However, if macroalgae produced in integrated systems are used in the formulation of feed for herbivorous species e.g. abalone, the nutritional content, as opposed to the agar content, would also influence the market value.

There are constraints, both financial and practical, that could limit the application of macroalgae in removing nutrients from wastewater discharged from commercial aquaculture. Fouling with particulate matter and biofouling with colonising epiphytes has been reported widely as restricting the production of macroalgae in horizontally integrated units. Phang, Shaharuddin, Noraishah and Sasekumar (1996) found that production of *Gracilaria changii* cultivated in shrimp ponds and an irrigation canal at Morib, on the west coast of peninsular Malaysia, was severely effected as a result of being heavily infested with epiphytes (*Hypnea* sp.). These authors also reported that siltation, wave action and predation by rabbit-fish (*Siganus* spp.) adversely affected the production of *G. changii.*

The turbid nature of wastewater can restrict light penetration into the water column, thereby limiting primary production. Phang et al. (1996) suggested that turbidity in shrimp ponds and an irrigation canal restricted the production of *G. changii.* Faced with this problem, Briggs and Funge-Smith (1996) recommended pre-treatment of shrimp farm effluent using either settlement, or bio-filtration using bivalves, as a primary stage for integrated macroalgae cultivation. Light entering the water column is also rapidly attenuated in productive cultures as excessive plant growth causes self-shading. Problems of self-shading and bio-fouling can be limited through intensive management, for example by frequent harvesting and/or agitation by aeration (Briggs and Funge-Smith, 1996). However, the cost of labour required to manage horizontally integrated seaweed culture may represent a substantial burden to the operator, although recent developments e.g. culturing seaweed in net tubes, as opposed to attaching individual plants to a substrate, may reduce labour demands (Zertuche-Gonzalez, Garcia-Lepe, Pacheco-Ruiz, Chee-Barragan and Gendrop-Funes, 1999). Despite additional costs associated with the integrated cultivation of seaweed, a financial analysis undertaken by Buschmann, Troell, Kautsky and Kautsky (1996b) suggested that income generated through sales of *G. chilensis* may be significant, making such a venture profitable.

Filter-feeding bivalve molluscs

Turbidity, negatively affecting the growth of macroalgae, may be prevented using a settlement pond or lagoon to remove particulate matter before it passes into the macroalgae culture unit. An alternative approach, that develops the concept of horizontal integration a stage further, is to use filter-feeding organisms such as bivalves to remove suspended solids whilst at the same time assimilating nutrients contained within the particulate waste

(Helfrich, Zimmerman and Weigmann, 1995; Soto and Mena, 1999). However, Briggs and Funge-Smith (1996) recommend that the wastewater should be pre-settled prior to entering the integrated bivalve culture to reduce fouling with particulate matter.

Pilot-scale experiments conducted by Lin, Ruamthaveesub and Wanuchsoontorn (1993) investigated culturing the green mussel (*Perna viridis*) stocked on bamboo sticks at a rate of 200-250 and positioned 40-50 cm below the water surface in a drainage canal carrying wastewater from a commercial shrimp farm in the Upper Gulf of Thailand. Survival to harvest was 85%, mean individual mussel weight increased from 12 to 42 g and mean meat weight increased from 7 to 28 g during the 113 day grow-out period. The potential of another bivalve, the Sydney rock oyster (*Saccostrea commercialis*), to remove nutrients and SS from shrimp (*Penaeus japonicus*) culture wastewater was tested in Moreton Bay, Australia (Jones and Preston, 1999). Pumping wastewater through tanks containing oysters (mean individual wet weight 55g) at a density of 0.7 per litre, resulted in mean inflow concentrations of SS (130 mg l^{-1}), TN (1.4 mg l^{-1}) and TP (0.15 mg l^{-1}) being reduced by 51, 20 and 33 per cent, respectively. Oysters stocked at a density of 0.5 l^{-1} removed only 36, 12 and 17 per cent of SS, TN and TP, respectively. However, the flow-rate employed is not reported, therefore, potentially useful indicators such as removal rates per unit time may not be calculated.

Guerrero and González (1991) investigated culturing the palourde clam (*Ruditapes decussatus*) in wastewater from a turbot farm in Ria de Arosa, Spain. During a 6 month trial, two groups of fifty clams with a total biomass of 1.9 and 6.4 g suspended in wastewater discharged from the farm increased in weight to 512 and 1,200 g, respectively. The authors suggested that as only ~20-30% of nitrogen and phosphorus added as feed is assimilated in intensively managed farms, unassimilated nutrients and fish metabolites may stimulate phytoplankton blooms, contributing to good growth and survival; mortality rates were below 10%.

Shpigel, Gasith and Kimmel (1997) studied the particulate removal efficiency of equivalently sized (mean wet weight 5 g) oysters (*Crassostrea gigas*) and clams (*Ruditapes philippinarium*) stocked in 14.4 litre troughs receiving 40 l h^{-1} of settled wastewater from ponds stocked with seabream (*S. aurata*). At densities of 7 l^{-1}, the treatment efficiency for oysters measured against influent turbidity levels of 47 NTU and chlorophyll *a* concentrations of 42 μ g l^{-1}, was 87% and 93%, respectively; clams reduced turbidity and chlorophyll *a* by 68% and 83%, respectively. With the same total biomass and mean individual weights, these authors noted that particulate removal by oysters was consistently more efficient than by clams, and that this was consistent with findings from other studies.

Clam seed (*R. decussatus*) (mean live weight 0.16 g) stocked at a density of 2.5 kg m^{-3} in 560 l tanks receiving 1,120 l h^{-1} of wastewater from a turbot farm in Galicia, Spain, displayed instantaneous growth rates of 0.036 over 60 days, as compared with 0.001 in a control tank receiving the same volume of fresh seawater (Jara-Jara, Pazos, Abad, Garcia-Martin and Sanchez, 1997). During the study, the Condition Index (CI) of seed in the control tank was lower with a range of 9-12, while that of wastewater grown clams ranged between 13-19. The levels of all biochemical components measured e.g. total carbohydrates, total lipids and protein, were higher in wastewater grown clams, as was the energy content. The CI, biochemical component levels, energy content and survival of seed cultured in wastewater were also higher than that of seed reared under natural conditions.

The authors attributed better performance using wastewater to continuous submergence, permitting uninterrupted feeding, as compared with the diurnal tidal rhythm and fluctuating food supply in natural systems. However, integrating bivalve culture into a continuous stream of aquaculture wastewater may cause biochemical changes within the organisms, such as the accumulation of lipids (Jara-Jara et al., 1997), which may affect processing or consumer acceptance. Continuous submergence may also be accompanied by increased fouling with solids and colonising species. Therefore, integration of filter-feeding molluscs may be most appropriate for systems with a variable discharge of wastewater, allowing periodic exposure to the atmosphere, thereby limiting fouling and predation. However, in culture systems where water exchange is sometimes restricted, such as shrimp farms at night, it may be necessary to circulate and aerate the water separately in the integrated culture unit to provide DO and limit fouling (Briggs and Funge-Smith, 1996).

The possibility that shellfish may harbour bacteria and toxic algae, or accumulate chemicals used to treat diseases and remedy water quality problems, must be regarded as a potential human health hazard. This may require operators of integrated systems to avoid chemical treatments in favour of alternative management options and ensure that a period of depuration is observed. Shpigel and Blaylock (1991) raised concerns that horizontally integrated bivalve culture could increase dissolved nutrient loadings due to the excretion of waste products, although in terms of overall mass balance a net reduction in nutrient loadings would be expected. Therefore, where further reductions in nutrient loadings are desirable, macroalgae or plant cultivation provides a logical step in developing the integrated system, examples of which are given in the following section.

Combined bivalve mollusc-macroalgae

Based on a series of pilot studies, Shpigel, Neori, Popper and Gordin (1993) modelled the performance of a settlement pond and culture units containing bivalves (*C. gigas*, *Ruditapes semidecussatus*) and seaweed (*U. lactuca*) in removing particulate matter and dissolved nutrients from wastewater from a land-based seabream (*S. aurata*) culture unit. Fish, seaweed and bivalves were found to assimilate 63% of nitrogen introduced to the system as feed, deposition in faeces accounted for a further 33 % and 4% was discharged to the marine environment. Based on nitrogen modelling alone, 3 kg of bivalves and 7.8 kg of seaweed could be produced through the assimilation of nitrogen released while producing 1 kg of fish. Although bivalve and seaweed sales could generate income, viability may be limited by the cost of developing the area required. Prior to applying these research findings it would be desirable to evaluate the model further, considering the dynamics of other critical nutrients.

Cultivating hairy cockle (*Scapharca inaequivalvis*) and seaweed (*Gracilaria* spp.) in wastewater from a commercial shrimp farm in Kota Bharu, Malaysia reduced the concentration of ammonium, TN and TP by 61, 72 and 61 per cent, respectively (Enander and Hasselstrom, 1994). Wastewater was passed through two ponds in series, one (12 m long, 2.5 m wide, 0.3 m deep) stocked with 60 kg of cockles and one (12 m long, 1.5 m wide, 0.7 m deep) containing 5 kg of seaweed. Two sets of ponds were operated in parallel and each received 5.5 m^3 d^{-1} of wastewater. Mean TN and TP concentrations in bivalve ponds were reduced by 55% and 67%, respectively, whilst seaweed ponds reduced mean TN and TP concentrations by 36% and 10%, respectively. A mass balance model predicted that during one month, bivalve production (30 kg) would sequester 260 g of nitrogen and 9 g of phosphorus, whilst seaweed production (60 kg) would assimilate 5 g of nitrogen and 7 g of phosphorus. The authors noted that bivalves and seaweed are both potentially useful products for shrimp farmers, as they may either be recycled for shrimp feed or sold for supplementary income. Another suggested advantage is that diversifying species may disperse financial risks associated with farming shrimp.

Based on previous studies, Shpigel and Neori (1996) proposed integrating the production of abalone (*Haliotis tuberculata*), macroalgae (*U. lactuca*, *Gracilaria* spp.), Manila clam (*T. tuberculata*) and seabream (*S. aurata*). A modular design was proposed to allow operators to focus on particular components depending on the availability of labour and seed, environmental conditions, market prices and the status of individual components. Commenting on possible operational demands the authors note that greater flexibility may be accompanied by greater complexity, requiring more skilled labour and sophisticated management.

Neori, Ragg and Shpigel (1998) conducted a pilot-scale study investigating one such modular culture unit. Seaweed (*U. lactuca* or *Gracilaria conferta*) was cultivated in 1,500 l tanks receiving wastewater from one of two 600 l abalone (*H. tuberculata*) culture units. Seaweed biomass produced was used to formulate a mixed feed for the abalone. Nitrogen input to the abalone tanks, each stocked with 235 abalone with a total biomass of 2.2 kg, averaged 494 mg per month, of which 62 mg (14%) was assimilated in biomass, at an areal rate of 105 mg m^{-2} d^{-1}, 284 mg (59%) was discharged in the effluent, the remainder was unaccounted for. Production of *U. lactuca* (initial standing crop 1.5 kg m^{-2}) averaged 230 g fresh weight d^{-1}, with maximum and minimum values of 412 and 52 g d^{-1} occurring in summer and winter, respectively. Seaweed harvested from the system removed 1,621 mg N m^{-2} d^{-1}, accounting for 34% of that entering the system. Mean annual specific growth rates for abalone ranged from 0.25-0.26% d^{-1}, compared with 1.16% d^{-1} reported by Mai, Mercer and Donlon (1996) for *H. tuberculata* fed with *U. lactuca*. Poor growth periods were attributed to competition and high feed conversion ratios associated with high summer temperatures, perhaps suggesting that the abalone species cultured was not suitable for the environment. Cumulative annual mortality rates for abalone of 32-39% were attributed partly to smothering and cannibalism, while handling stress following stocking and rapidly declining water temperatures during autumn were considered responsible for periods of high mortality. Growth, and therefore treatment capacity, of *G. conferta* was also affected by variable water temperatures, again indicating a lack of suitability for integration. Although practical management constraints were identified, the authors note that where integration is not practiced, double the water volume would be required to supply the separate culture units, nitrogen from abalone culture units would be discharged to the sea, whilst seaweed cultivation would require nitrogen fertiliser.

Despite the apparent benefits of culturing several species in horizontally integrated systems, prior to commercialisation the conclusions drawn from the pilot studies presented above will require additional validation and testing under a wider range of environmental and operating conditions. Horizontally integrated units producing inputs for primary aquaculture activities e.g. feed or seed, have considerable appeal as they potentially increase resilience to external factors. However, management will be subject to similar constraints to those identified for integrating single secondary aquaculture activities, and a higher degree of dependence on 'feedback loop' inputs will increase the need for control. Furthermore, the risk of disease transmission between culture units, both to the primary culture system from feed cultivated downstream and to seed from the on-growing unit should be carefully assessed, and where necessary, suitable precautions such as monitoring, depuration or processing should be introduced to avoid such problems.

Other candidate species for integration

Species other than macroalgae and molluscs, such as microalgae, zooplankton, marine diatoms, periphyton, aquatic macrophytes and fish may also be considered for integration.

Microalgae

Dumas, Laliberté, Lessard and de la Noüe (1998) investigated using a non-toxic cyanobacterium (*Phormidium bohneri*) cultured in completely mixed 70 l photobioreactors with retention times of 8, 12 and 24 h, to remove dissolved inorganic nutrients from wastewater discharged from two 260 l tanks stocked with 19 rainbow trout (mean weight 568 g). During a one month period, an average of 82% of TAN and 85% of soluble orthophosphate at mean influent concentrations of 1 and 0.12 mg l^{-1}, respectively, was removed. However, instantaneous growth rates for *P. bohneri* of 0.03-0.06 d^{-1} were only one tenth of those recorded in cultures receiving cheese factory effluent; this was attributed to higher concentrations of TAN (22 mg l^{-1}) and phosphorus (15 mg l^{-1}) in the cheese factory waste.

Lefebvre, Hussenot and Brossard (1996) batch cultured diatoms in outdoor 16 l tanks filled with wastewater from land-based marine fish farms. Over 3-5 days ~90% of inorganic matter was removed from the culture water. Diatom growth was limited by wastewater silicate concentrations, and supplementary silicate (Na_2SiO_3) inputs significantly enhanced both production and nutrient removal rates. Depending upon the nutrients present in the wastewater either *Skeletonema costatum* or *Chaetoceros* spp. dominated the batch cultures.

In the study by Dumas et al. (1998) treatment efficiency of *P. bohneri* was significantly reduced below 15°C, therefore the cost of maintaining a suitable temperature in the photobioreactor could represent a considerable limitation to this treatment technique in temperate climates. These authors also found that competition between nitrifying bacteria and *P. bohneri* could limit effectiveness. However, it is difficult to assess the potential effects of grazing and contamination with other algae species on the dominance and effectiveness of *P. bohneri* as the investigation only lasted for one month. The potential affect of chemotheraputants used in the aquaculture facility on microalgae cultures represents another constraint. Furthermore, retention times employed by Dumas et al. (1998) and Lefebvre et al. (1996) indicate that development of such approaches for commercial aquaculture operations will only be feasible where pond-based culture strategies are developed, similar to those used to culture microalgae commercially in Australia, America and Israel (Borowitzka, 1993).

Zooplankton

Zooplankton play an important role in the nutrition of various fish and prawn species (Kibria, Nugegoda, Fairclough, Lam and Bradly, 1997 and 1999). The integrated production of cladocerans, copepods, rotifers or euphausids in aquaculture wastewater could represent an important source of zooplankton for use as either a live feed for crustaceans and fish larvae or as a component in a formulated diet. Kibria et al. (1997 and 1999) described the harvest of zooplankton from wastewater treatment lagoons at Werribee sewage treatment plant, the largest in Australia. Zooplankton (*Daphnia carinata* and *Moina australiensis*) proliferate in the final treatment lagoons and are harvested at a rate of 40-84 kg h^{-1} by filtering the water through screens mounted on a floating platform. The harvest of 100 kg of fresh zooplankton removes approximately 0.1 kg of phosphorus and 1 kg of nitrogen. Furthermore, these zooplankton were found to be rich in protein, essential amino acids, lipids and phosphorus, and the specific growth rate (2.97), feed conversion ratio (1.18) and protein efficiency ratio (1.57) for silver perch (*Bidyanus bidyanus*) fed frozen *D. carinata* were not significantly different to those of fish receiving a commercial diet (Kibria et al., 1999).

Combined algae-zooplankton

Gnudi et al. (1991) cultivated zooplankton (*Daphnia* spp.) in sixteen 1 m deep ponds (1,200 m^2) that received microalgae from cultures produced in four 0.5 m deep basins (2,000 m^2). The microalgae culture was fertilised with 1.5-6 l m^{-2} d^{-1} of pig manure, depending on climatic conditions. The dry weight microalgae standing crop decreased during the 4 month study from 62 mg l^{-1} in October to 41 mg l^{-1} in January; this was attributed to lower solar radiation, 940 cal cm^{-2} d^{-1} in October as compared with 615 cal cm^{-2} d^{-1} in January, and reduced mean daily temperatures, 12°C in October and 3°C in January. The dry weight of microalgae transferred to the zooplankton cultures also declined during the study from 4.4 to 1.6 g m^{-3} d^{-1}, and in combination with declining temperatures resulted in mean weekly wet weight zooplankton harvests (40% of standing biomass) falling from 80 to 28 g m^{-3}. Groeneweg and Schlüter (1981) fed rotifer

(*Brachionus rubens*) cultures in 50 l indoor tanks on algae diets produced in shallow 9.7 m^2 raceways fertilised with diluted pig manure. Each day 12.5 l of pond water was introduced to the rotifer culture, giving a retention time of 4 days. Cultures were produced containing 200-580 rotifers per ml, depending on the concentration of algal-bacterial biomass in the pond water, thus:

$$Y = 1.54X - 226.86$$

where, Y = rotifers (ml^{-1})

X = total SS ($mg\ l^{-1}$)

Schlüter and Groeneweg (1981) described the environmental requirements e.g. nitrite, salinity, DO and pH for culturing *B. rubens*; DO levels were considered especially important, with minimum concentrations of 1.15 mg l^{-1} being recommended; below this reproduction and survival decreased. Although other food sources have been successfully used for culturing rotifers, those produced with algae diets had a higher nutritive value for juvenile fish.

Combined algae-bivalves

Hussenot, Lefebvre and Brossard (1998) investigated culturing bivalves (*C. gigas*) on microalgal cultures grown in a 1.2 m deep reactor receiving wastewater from an intensive seabass (*Dicentrarchus labrax*) farm. A mean dilution rate of 70% was maintained in the 48 m^3 reactor. Concentrations of TAN (1.85 mg l^{-1}) and phosphorus (1.1 mg l^{-1}) in wastewater passing through the microalgal reactor were reduced by 67% and 47%, respectively, however, total pigment concentrations (0.02 μg l^{-1}) increased by 1,139%. Adding sodium silicate to achieve an optimal Si:P ratio of 4:1, ensured that a native diatom

(*Skeletonema costatum*) that had shown good growth in a previous study (Lefebvre et al., 1996) proliferated, dominating the algal species leaving the reactor to the bivalve culture unit. Bivalves cultured in 1 m^3 tanks, receiving 1.2 m^3 d^{-1} of wastewater and stocked at a biomass of 424 mg l^{-1}, removed 50% of total pigments; however, during gametogenesis the treatment efficiency was lower as food availability was reduced to avoid mortalities. Although promising, the cost of sodium silicate and a system to regulate and monitor its levels in the microalgae reactor may prohibit commercial development, and reduced treatment capacity during gametogensis may represent a further practical constraint.

Combined periphyton-fish

The cultivation of bivalves using microalgae suspensions grown on wastewater appears feasible, however, few freshwater bivalves are commercially important. Edwards and Sinchumpasak (1981) predicted that culturing tilapia (*O. niloticus*) in ponds receiving microalgae entrained in high rate stabilization pond effluent could achieve yields of ~20 t ha^{-1} y^{-1}; however, fish production in the optimised system proposed by the authors was subsidised by an area of stabilisation ponds ~1.9 times larger than the fishponds. Furthermore, no trials were undertaken to validate an extrapolated yield of 20 t ha^{-1} y^{-1}, based on a three month trial, whilst collapses in the stabilisation pond algal population and early morning oxygen depletion by algal populations in the fishpond were also identified as possible constraints. Dempster, Baird and Beveridge (1995) reviewed ingestion rates for tilapia feeding on algae suspensions in laboratory studies and, despite the good growth reported by Edwards and Sinchumpasak (1981), found the efficiency of this feeding strategy unable to account for algae volumes in tilapia guts reported in the literature. Furthermore, a bio-energetic model developed by the authors suggested that tilapia feeding on only algae suspensions were unable to fulfill basic nutritional demands. It was concluded that high algae ingestion rates must be achieved by additional feeding on algae aggregates in the water column, grazing on algae-based detritus, particles covered in cyanobacteria and periphyton. Trials undertaken by Shrestha and Knud-Hansen (1994) confirmed that 20 g tilapia (*O. niloticus*) stocked at a rate of 3 m^{-2} in concrete tanks (2.5 x 2 x 1.1 m) with five vertical plastic baffles (1.26 x 0.6 m) grazed extensively on attached microorganisms and detrital biomass. However, comparing average fish production under these conditions (0.88 g m^{-2} d^{-1}) with that in tanks stocked at similar rates without baffles (1.05 g m^{-2} d^{-1}) showed no statistical difference. Consequently, these authors concluded that there was a trade-off between phytoplankton versus attached algae growth, which accounted for the similarities in net fish yields.

Drenner, Day, Basham, Smith and Jensen (1997) investigated the use of stoneroller minnows (*Campostoma anomalum*) and tilapia (*Tilapia mossambica*) to graze periphyton growing on porous screens in order to remove nitrogen and phosphorus from wastewater. It was expected that fish feeding on periphyton would either assimilate nutrients or expel them in faeces, resulting in the nutrients being permanently removed from the system by harvesting fish and removing faeces collected in sediment traps. However, TP removal in tanks stocked with fish (~48 mg m^{-2} d^{-1}) was considerably lower than that reported in studies where periphyton was harvested by vacuuming (104-139 mg m^{-2} d^{-1}). Three factors were suggested to account for the observed differences:

- fish may be less efficient as they excrete excess phosphorus back into the water;
- the low nitrogen to phosphorus ratio in the water used in the investigation, although equivalent to that in domestic wastewater, may have caused periphyton production to be limited by the availability of nitrogen and not phosphorus, hence reducing the rate of phosphorus removal in periphyton biomass;

- the design of the tanks used in the investigation and the configuration of the screens within the tanks had not been optimally designed for periphyon growth and nutrient uptake.

High mortality rates for fish during the trial, which Drenner et al. (1997) attributed to low temperatures, indicated restricted potential in temperate climates by the limited choice of cold tolerant fish that consume periphyton. Although the system could be enclosed, or heat recovery applied to regulate the temperature, this may not be cost effective.

The bamboo-*acadja* operated in Benin represents an alternative approach for enhancing primary production in nutrient rich water through the introduction of a substrate that can be colonised by periphyton (Hem, Avit and Cisse, 1995). In the traditional *acadja*, fish are attracted to bundles of twigs placed in coastal lagoons, as they offer protection and shelter and provide a substrate suitable for the growth of microfauna and periphyton. This concept has also been developed as a sustainable aquaculture system capable of enhancing fish production from freshwater ponds in West Africa (Hem et al., 1995). The traditional *acadja* resulted in local forests being overexploited, and therefore bamboo has been introduced as an alternative substrate. When tested in the traditional setting of coastal lagoons the bamboo-acadja resulted in fish production of between 4-8 t ha^{-1} y^{-1} (Hem et al., 1995). When transferred to freshwater ponds, an annual production of 3.6 t ha^{-1} was obtained, a significant improvement over traditionally managed rural ponds.

A similar pattern of development has been observed in Bangladesh where the principles of the traditional *katha* fisheries, like the *acadja* of West Africa, have been applied to pond-based culture systems (Wahab, Azim, Ali, Beveridge and Khan, 1999a; Wahab, Mannan, Huda, Azim, Tollervey and Beveridge, 1999b). In ponds containing bamboo poles at a rate ~93,000 ha^{-1} and stocked with calbaush (*Labeo calbasu*) at densities of 10,000 fingerlings ha^{-1}, production over 120 days was 713 kg ha^{-1}, significantly higher than in ponds without substrate (399 kg ha^{-1}) (Wahab et al., 1999a). However, Wahab et al. (1999b) found that similar densities of bamboo poles (90,000 ha^{-1}) in ponds stocked with rohu at densities of 10,000 fingerlings ha^{-1}, resulted in production during a four month period of 1,899 kg ha^{-1}, compared with 1,089 kg ha^{-1} in control ponds. Despite significant increases in production in both studies the authors note the need to conduct a financial analysis to test the appropriateness of the system for rural pond operators in Bangladesh. Furthermore, Shrestha and Knud-Hansen (1994) suggested that in ponds receiving low fertiliser inputs and where primary production in the water column was limited, the addition of substrate may facilitate more efficient feeding by tilapia. However, with greater fertilisation and primary production, fish yields may be as high without using substrates. Consequently, to fully assess the potential of periphyton-based aquaculture, further investigation is required concerning the relative trade-off between attached and free-floating algae biomass in fertilised ponds.

The potential value of periphyton in other production systems, such as pond-based prawn (*Macrobrachium rosenbergii*) grow-out, cage-based tilapia (*Oreochromis mortimeri*; *O. niloticus*; *Tilapia rendalli*) culture and in rearing catfish (*Clarias gariepinus*) fry, has also been reported (Norberg, 1999; Nwachukwu, 1999; Tidwell, Coyle, Van Arnum and Weibel, 2000). The acadja described by Hem et al. (1995) received organic inputs as chicken droppings inserted into the top sections of the bamboo poles pushed into the bottom of the pond. However, nutrient enrichment with wastewater from commercial aquaculture could be used to stimulate production within a similar system.

Aquatic macrophytes

A number of aquatic macrophytes may have roles in horizontally integrated units for aquaculture wastewater. Studies have described using aquatic macrophytes to treat discharges of industrial, agricultural and domestic wastewater (Nelson, Smith and Best, 1981; Ghobrial and Siam, 1998; Sun, Gray and Biddlestone, 1998; Griffin and Upton, 1999). Floating aquatic macrophytes such as duckweed (*Lemna* spp., *Spirodela* spp., *Wolffia* spp., *Wolffiella* spp.), water hyacinth (*Eichhornia cassipes*), water fern (*Azolla* spp.) and water lettuce (*Pistia* spp.) have received considerable attention due to their high productivity and ability to grow on turbid waters that would exclude most submerged and emergent aquatic macrophyes.

Alaerts, Mahbubar and Kelderman (1996) described the treatment performance of a duckweed-covered lagoon receiving wastewater from Kumidini Hospital, Mirzapur, Bangladesh. The 0.6 ha duckweed lagoon was 500 m long, ~13 m wide and increased in depth from 0.4 m at the inflow to 0.9 m at the outflow. On a weekly basis, the lagoon received ~2,000 m^3 of pre-settled wastewater, surface BOD loads ranged from 48 to 60 kg ha^{-1} d^{-1}, and the retention time was estimated at 20.4 d. Monitoring work over four weeks in the dry season showed that the lagoon removed 95-99% of BOD and 74-77% of both TN and TP; during this period duckweed production range from 58-105 kg (dw) ha^{-1} d^{-1}. Based on these production data, nitrogen and phosphorus removal in harvested duckweed biomass was estimated at 0.26 and 0.05 g m^{-2} d^{-1}, respectively. Annual duckweed yields of 21.2-38 t (dw) ha^{-1} were extrapolated, equivalent to 261-438 t ha^{-1} y^{-1} of fresh duckweed biomass. Further studies concerning duckweed cultivation employing wastewater resources described by Edwards, Hassan, Chao and Pacharaprakiti (1992), Oron (1994) and van der Steen, Brenner and Oron (1998) have confirmed the effectiveness of this approach and suggested that biomass production levels similar to those reported by Alaerts et al. (1996) may be expected.

Constructed wetlands

The term 'constructed wetland' refers to wetlands that are established specifically to treat wastewater; the majority are planted with either the common reed (*Phragmites australis*), bulrush (*Scirpus* spp.) or cattail (*Typha* spp.), and they have been used to treat domestic and industrial wastewater (Butler, Loveridge, Ford, Bone and Ashworth, 1990; Kadlec and Knight, 1996). Two hydrological regimes prevail in constructed wetlands, subsurface and surface-flow. In the former, wastewater percolates either horizontally or vertically through the substrate-root matrix, whilst in the latter, overland flow dominates. Based on the assessment of empirical data from a wide range of systems, design parameters for subsurface and surface-flow wetlands have been proposed (see Kadlec and Knight, 1996). Constructed wetlands facilitate a range of physical, chemical and biological processes that remove pollutants from wastewater; these are summarised in Table 2.1. The possibility of using constructed wetlands to treat aquaculture wastewater has been reviewed (Baldwin, 2000) and studies have been conducted in various settings. Conventional constructed wetlands have been developed primarily to receive freshwater. However, the saline water used to culture shrimp has meant that species from mangrove ecosystems have been incorporated into constructed wetlands (Samocha and Lawrence, 1997; Wilson, 1999).

A subsurface-flow constructed wetland planted with mangrove fern (*Acrostchum aureum*) was used at the pilot-scale to treat wastewater from commercial shrimp farms in Thailand (Sansanayuth, Phadungchep, Ngammontha, Ngdngam, Sukasem, Hoshino and Ttabucanon, 1996). Results indicated that the wetland reduced concentrations of SS by 84%, BOD by 91%, total organic carbon by 46% and TN and TP by 48% and 31%, respectively. Robertson and Phillips (1995) studied the potential of mangroves as filters for wastewater from shrimp ponds in Thailand. Employing a mass balance analysis they estimated that the area of *Rhizophora*-dominated mangrove required to assimilate excess nitrogen and phosphorus from 1 ha of semi-intensive (4 PL m^{-2}) and intensive (52 PL m^{-2}) culture ponds yielding 1 and 13.8 t ha^{-1} y^{-1}, would be 3 and 22 ha, respectively. Furthermore, the authors report a production rate for mangrove wood of 20 t ha^{-1} y^{-1}, therefore, it may be estimated that shrimp production in 1 ha of semi-intensive and intensive ponds could sustain the production of 60 and 440 t y^{-1}, respectively, of marketable wood.

Rivera-Monroy, Torres, Bahamon, Newmark and Twilley (1999) estimated that the potential of a mangrove forest to reduce dissolved inorganic nitrogen in shrimp pond wastewater was ~190 mg m^{-2} d^{-1}. Considering typical concentrations of TAN (67 μg l^{-1}) and nitrate and nitrite combined (34 μg l^{-1}) for wastewater discharged from semi-intensive (13-24 PL m^{-2}) shrimp ponds on the Caribbean coast of Colombia, the authors predicted that 0.04-0.12 ha of mangrove would be required to remove the dissolved inorganic nitrogen discharged from 1 ha of shrimp ponds.

Robertson and Phillips (1995) noted several practical constraints regarding the management of mangroves as filters for wastewater from shrimp farms. Mangroves are not homogenous; their hydrology depends on a number of topographical and other factors e.g. sediment characteristics, root architecture, canopy thickness and the activity of burrowing animals. Consequently, it is unlikely that wastewater will be evenly dispersed throughout the mangrove, potentially causing short-circuiting, and resulting in the assimilative capacity of certain areas being overwhelmed. Excessive loads of TAN, particulate organic matter and dissolved organic matter could lead to anaerobiosis within the mangrove sediments, resulting in tree mortality and negative impacts on benthic fauna, including the loss of burrowing animals. These authors also discuss the long term assimilative capacity of mangrove sediments for phosphorus, the influence of different sediment types on the retention and transformation of nitrogen, and the silviculture practices resulting in the optimal combination of forest growth and nutrient uptake. They note the need for such issues to be fully considered.

Brown, Glenn, Fitzsimmons and Smith (1999) investigated using halophytic species with potential as forage crops (*Suaeda esteroa*) and oil seed production (*Salicornia bigelovii* and *Atriplex barclayana*) to remove nutrients from saline aquaculture wastewater.

Table 2.1. Wastewater treatment functions of constructed wetlands

Function	Removal			Description
	Primary	Secondary	Incidental	
Physical				
Sedimentation	- settleable particles	- colloidal particles	- BOD, heavy metals, refactory organics, N, P, bacteria, viruses	- gravity settles solids and constituent contaminants
Filtration		- settleable and colloidal particles		- particles filtered mechanically as water passes through substrate, root mass, or fish
Adsorption		- colloidal particles		- inter-particle attractive forces (van der Waals force)
Chemical				
Precipitation	- P, heavy metals			- formation of or coprecipitation with insoluble compounds
Adsorption	- P, heavy metals			- adsorption on substrate and plant surface
Decomposition	- refactory organics	- refactory organics		- decomposition or alteration of less stable compounds by processes such as UV irradiation, oxidation and reduction
Biological				
Microbial metabolism	- colloidal particles, BOD, N, refactory organics, heavy metals			- removal of colloidal solids and soluble organics by suspended, benthic and plant-supported bacteria; bacterial nitrification/denitrification; microbially mediated oxidation of metals
Plant metabolism		- refactory organics, bacteria, viruses		- uptake and metabolism of organics by plants; root excretions may be toxic to enteric organisms
Plant absorption		- N, P, heavy metals, refactory organics		- plant uptake under normal operating conditions
Natural die-off	- bacteria, viruses			- decay of organisms in unfavourable environment

Adapted from Watson, Reed, Kadlec, Knight and Whitehouse (1989)

Wastewater with salinities of 0.5‰, 10‰ and 35‰ was used to irrigate lysimeters planted with monocultures of these species. Hydraulic loading rates were modified in each of the treatments to meet the demands of evapotranspiration and to ensure that 30% of the applied wastewater leached from the container. Irrespective of plant species and salinity, the plant-soil systems consistently sequestered TN and phosphorus at rates of 98% and 99%, respectively, during a 4 month trial period. Rates of inorganic nitrogen and phosphorus removal were slightly lower at 94% and 97%, respectively.

Examining the fate of nutrients applied in 0.5‰ and 10‰ salinity treatments, only 21% of nitrogen and 15% of phosphorus, were sequestered in plant shoots; the majority of nitrogen (76%) and phosphorus (84%) was retained within the root-soil matrix or released as gaseous products of bacterial degradation. Longer-term trials could determine the ultimate fate of nutrients applied in wastewater as the root-soil matrix may eventually become saturated, leading to nutrient leaching.

Brown and Glenn (1999) investigated reusing saline (31‰) tilapia culture wastewater to cultivate *S. esteroa*, a halophyte shrub with potential as a forage crop. Over four months, shrubs planted in 0.78 m^3 lysimeters (0.76 m deep, 1.02 m^2 surface area) were irrigated with wastewater three times per week at rates ranging from 50 to 250% of the pan evaporation rate (E_{pan}). Plant biomass increased significantly with higher application rates: with irrigation volumes of 212 l (50% E_{pan}) and 1,060 l (250% E_{pan}) mean net plant dry weights after four months were 385 g and 694 g per lysimeter, respectively. However, water use efficiency was lower at 1.7 g l^{-1} with an application rate of 250% E_{pan}, as compared with 2 g l^{-1} at 50% E_{pan}. Nitrogen leaching from the lysimeters decreased with time and increasing irrigation rates, in contrast leachate phosphorus concentrations increased with time and higher irrigation rates. Mean nitrate and soluble reactive phosphorus (SRP) concentration in composite irrigation water samples were 8.8 and 14.6 mg l^{-1}, respectively. During week 14, with an application rate of 100% E_{pan} the leachate nitrate concentration was 27 mg l^{-1}, whilst at 250% E_{pan} it was 0.8 mg l^{-1}; at 100% E_{pan} the leachate SRP concentration was 1.2 mg l^{-1}, whilst at 250% E_{pan} it was 3.7 mg l^{-1}. The authors concluded that where phosphorus is not a limiting factor in the receiving environment, using saline aquaculture wastewater to irrigate halophyte crops is an appropriate treatment strategy. It was estimated that wastewater from a 1 ha shrimp pond (1 m deep) with exchange rates of 20% and 70% per week could be used to grow ~2.5 and 13 ha of *S. esteroa*, respectively, at irrigation rates of 250% E_{pan}. Furthermore, these authors reported that production of *S. esteroa* (6.9 g m^{-2} d^{-1}) irrigated with 1.04 m of water over 98 days (250% E_{pan}) was greater than that of Sudan grass (6.1 g m^{-2} d^{-1}) irrigated with 1.2 m of water.

Aquaculture wastewater treatment and reuse through irrigation could be regarded as challenging to the definition of horizontally integrated aquaculture; based on the definition, horizontal integration would only occur if the plants being irrigated were aquatic in nature. Drawing a distinction between aquatic plants that tolerate drying and terrestrial plants that tolerate flooding may be difficult; the concept of aquaponics, cultivating terrestrial plants using aquaculture wastewater in the absence of soil further tests the definition. However, despite possible problems concerning the delineation of production strategies that constitute valid approaches to horizontally integrated aquaculture it was considered important to propose a tentative definition to assist in focusing the present study on a diverse range of emerging production strategies which have been developed to address common problems and meet specific objectives. Furthermore, although the definition appears to exclude apparently closely allied practices, this demonstrates that the definition may be applied rigorously, contributing to the focus of the study and consequently the validity of resulting conclusions and recommendations.

Schwartz and Boyd (1995) used reedbeds to treat water from a channel catfish pond. Two cells, measuring 84 m long and 14 m wide, were constructed in series; the first was planted with California bulrush (*Scripus californicus*) and giant cutgrass (*Zizaniopsis miliacea*); the second with Halifax maidencane (*Panicum hemitomon*). Hydraulic residence times (HRTs) of 1-4 days and hydraulic loading rates from 77-91 l m^{-2} d^{-1} were tested. Reductions in waste concentrations ranged from 1-81% for TAN, 43-98% for nitrite, 51-75% for nitrate, 45-61% for TN, 59-84% for TP, 37-67% for BOD and 75-87% for SS. The best overall performance figures were obtained in vegetated cells with a HRT of 4 days; SS, TN and TP mass loadings of 34.6, 4.8 and 0.16 kg ha^{-1} were reduced by 31.4, 3.1 and 0.14 kg ha^{-1}, respectively. However, good pollutant removal rates were also recorded with shorter retention times and when the vegetation was dormant. The authors recognised that the large area of land required to treat wastewater discharged from aquaculture can be a limitation. However, using large wetland areas could be justified through the semi-intensive production of additional crops; for example, fish, plants or crustacea.

Some forms of valliculture in the northern Adriatic, described by Melotti, Colombo, Roncarati and Garella (1991), utilised wastewater from the intensive culture of seabass (*D. labrex*) and seabream (*S. aurata*) to fertilise areas for the extensive culture of seabass, seabream, mullet (*Mugil* spp.) and eel (*A. anguilla*). The authors noted that the mean harvest weight for mullet increased significantly following wastewater reuse from 302 g in 1987, to 375 g in 1990; however, at the same farm, mean harvest weights for seabass declined significantly, from 525 g to 422 g. It was also proposed that wastewater reuse in extensive culture systems could reduce coastal zone pollution, although no change was recorded in nutrient levels in neighbouring coastal waters.

Where constructed wetlands for primary aquaculture wastewater treatment are prohibited by physical or hydrological constraints, development of smaller constructed wetlands could assist in managing sludge produced following treatment using conventional approaches. Nielson (1990) incorporated common reed (*P. australis*) in a conventional unplanted drying bed for concentrated domestic sludge and concluded that this increased the rate of de-watering and mineralisation. The potential of using vertical and horizontal flow wetlands planted with vetiver grass (*Vetiveria zizanioides*) to treat backwash water from filters in an aquaculture system stocked with trout and employing water reuse has been described (Summerfelt, Adler, Glenn and Kretschmann, 1999). Sludge was applied to the wetlands at a rate of ~13.5 l m^{-2} d^{-1} or 30 kg dry solids m^{-2} y^{-1}. Vertical and horizontal flow wetlands removed 98% and 96% of SS and 91% and 72% of COD, respectively. Concentrations of TN, TP and dissolved phosphate were reduced by 82-93% in both wetland types. Mineralisation in the wetlands was also significant, with total volatile solids in the sludge reduced by 50%.

Aquaculture sludge reuse in agriculture

The use of sludge, produced during the treatment of aquaculture wastewater, to fertilise agricultural crops has been proposed (Bergheim, Cripps and Liltved, 1998; Chen, 1998). However, this strategy fails to meet the proposed definition of horizontally integrated aquaculture. Reuse of sludge and wastewater from aquaculture facilities as production enhancing inputs to other aquatic systems would meet the definition, and the use of enriched pond water to irrigate rice fields that may be used to raise fish has been described (Little and Muir, 1987).

2.6.5. Open horizontally integrated aquaculture

The potential for self-pollution in aquaculture is greatest in open production units e.g. cage and pen systems, situated within the ecosystem to which they discharge waste. Costa-Pierce (1996) describes how waste collected under lake-based trout cages could be pumped on to the land to facilitate environmental enhancement through the creation of wetlands and the rehabilitation of eroded hillsides. This may be suitable for cages in lakes or sheltered coastal locations close to land, where topographic and nutrient load conditions are suitable, but is unlikely to work for cages in exposed or offshore settings or where the additional nutrients may overload terrestrial systems. In addition to regular maintenance, the pump-ashore facility would require a localised energy source, a possible constraint at isolated sites. Furthermore, capital costs of infrastructure for collecting and transferring waste matter and constructing the wetland may deter commercial development. As a complementary approach to this system Costa-Pierce (1996) suggested that a secondary enclosure stocked with unfed baitfish could be positioned around the grow-out cages, and that this *ecosystem enclosure* would capture fine particles and dissolved phosphorus released from the grow-out cage. Fish produced could then be either sold or released to enhance capture fisheries.

Troell, Halling, Nilsson, Buschmann, Kautsky and Kautsky (1997) integrated cultivation of *G. chilensis* on ropes close to cages producing salmonids (*O. mykiss* and *O. kisutch*) in Chile. Extrapolated yields indicated that 1 ha of *G. chilensis* cultivated in this way had the potential to remove 5% of the dissolved nitrogen and 27% of the phosphorus released from the cages. However, an economic analysis of the proposed system was not undertaken, leaving its commercial viability uncertain. Petrell, Mazhari Tabrizi, Harrison and Druehl (1993) developed a bioeconomic model for the integrated culture of kelp (*Laminaria saccharina*) in coastal waters close to a 12 cage salmon farm producing 250 t y^{-1}. It was predicted that a kelp farm consisting of ten 60 m long ropes would produce 1.6 t y^{-1} of dried seaweed, furthermore, based on an initial investment of ~61,000 Canadian dollars[2] a pay-back period of 6 years was estimated and the net present value of the enterprise was positive at rates of return up to 25%. The authors also estimated that with typical current velocities on salmon farms of 0.1 m s^{-1} and critical oxygen concentrations of 5 mg l^{-1}, oxygen consumption in the kelp farm (0.026 mg g^{-1} h^{-1}) would account for less than 1% of the available oxygen.

[2] 1 Canadian dollar = US$1.19; May 1992 (Petrell et al., 1993)

Stirling and Okumus (1995) studied the growth of mussels (*Mytilus edulis*) suspended from sea cages containing Atlantic salmon in a loch on the west coast of Scotland. The growth of mussels was related to temperature and food availability and consequently showed a clear seasonal trend, being relatively rapid from May to October but significantly slower during the rest of the year. Mussel growth near to salmon cages was better than that at a neighbouring shellfish farm, probably due to the higher concentration of suspended organic matter at the cage site. However, in temperate climates, the seasonal growth of shellfish may limit their effectiveness as bio-filters. Despite the apparent benefits of integrating the culture of both seaweed and shellfish with open aquaculture systems, little empirical evidence is available to confirm practical application and economic performance (Buschmann, Lopez and Medina, 1996a).

Ranching

Thorpe (1980) stated that salmon ranching may be defined as: "an aquaculture system in which juvenile fish are released to grow, unprotected, on natural foods in marine waters from which they are harvested at marketable size". In addition to feeding on natural biota, fish stocked as part of a ranching programme could feed directly on waste feed and faeces discharged from aquaculture facilities. Johansson, Hakanson, Borum and Persson (1998) studied the fate of indicator particles contained in commercial feeds given to rainbow trout cultured in a commercial farm in Lake Southern Bullaren, Sweden. Results indicated that wild fish consume a significant proportion of faeces expelled by cultured fish. Therefore, fish stocked close to a cage facility as part of a ranching programme may be expected to exploit this source of nutrition, assimilating nutrients and reducing the impact on underlying sediments. The foraging behaviour of grey mullet (*Mugil cephalus*), stocked into open-bottom cages on the seabed, significantly improved the quality of sediments below a commercial cage farm in the northern Gulf of Aqaba, Eilat (Porter, Krost, Gordin and Angel, 1996). After a period of seven weeks, the proportion of organic matter and redox potentials in sediments below the farm returned to levels comparable to an undisturbed reference site. Furthermore, despite the foraging of the mullet, the invertebrate assemblage in sediments below the cage shifted from a mono-specific population of small nematodes to a mixed community of nematodes and polychaetes; a diverse macrofauna assemblage consisting of nematodes and polychaetes was observed at the reference site.

Several factors could constrain the adoption of ranching; unrestricted movement may lead to the species stocked avoiding heavily impacted sites, and therefore potential benefits, such as those reported by Porter et al. (1996) may not be realised. Species being ranched may also range widely, however, optimal foraging theory suggests that this freedom-of-movement may assist in optimising the assimilation of nutrients from the receiving environment. Containment strategies, such as bubble curtains and behavioural conditioning, could be employed to prevent animals being ranched from leaving the immediate vicinity of the farm site. However, such approaches are likely to be expensive and difficult to manage. Problems with retaining stocked species close to aquaculture facilities indicates that organisms which are largely sessile in nature, e.g. scallops, have the greatest potential as candidate species for horizontal integration through ranching.

Deliberately stocking species in aquatic ecosystems to induce changes in ecological processes, ameliorating negative environmental impacts, is a key element to biomanipulation. Approaches to biomanipulation include: introducing predatory fish, or suppressing zooplanktivores to increase phytoplankton grazing by zooplankton; encouraging filter-feeders, reducing the seston concentration; introducing herbivores to increase the conversion efficiency of primary production and promoting populations of organisms which are readily consumed by fish (Klapper, 1991). Despite several examples where the biomanipulation of fish populations has produced significant improvements in water quality, several constraints to this approach have been identified. Suppressing populations of fish, particularly small species and juveniles, is labour intensive and difficult to maintain (Moss, 1992) and grazing pressure from the increased population of zooplankton may lead to the selection of undesirable phytoplankton species. Furthermore, unless nutrient levels in the water-body are permanently reduced, any improvement in water quality will be unsustainable and the ecosystem will tend to revert to its original trophic state. The removal of nutrients by harvesting species stocked in aquatic ecosystems has been considered to a limited extent. Fichtner (1983; cited in Klapper, 1991) described how harvesting grass carp stocked in a weed-infested lake removed 5-10% of the phosphorus from the water-body. Beveridge (1984) reported that operators of a Scottish cage farm producing 200 t y^{-1} of rainbow trout recovered 10 t of escaped fish using nets, whilst anglers caught a further 2.5 t; this generated income for the farm through increased sales and angling revenue and reduced the annual phosphorus load to the lake by 1.3%.

To be considered within the definition of aquaculture (FAO, 1995) operators engaging in ranching must retain ownership of organisms being cultured. Ownership may be difficult to establish unless the stocked species can be identified e.g. by using tagging to identify populations of scallops, lobsters and fish. Previous studies concerning ranching can be used to identify high potential management strategies and suitable species for ranching. However, concerns regarding the impact of introducing non-indigenous species may restrict ranching operations to using native species or organisms originating from local populations. Harvesting may also require operators to invest in suitable equipment and develop new skills. Furthermore, operators may face problems in obtaining exclusive rights to harvest the ranched species. Exclusion of other actors from the fishery may be difficult, especially where commercial fishing operations are established. Although ranching has the potential to enhance commercial fisheries, there may be little incentive to invest unless mechanisms are developed that allow operators to benefit from increased catches. A potential strategy might be to transfer revenue generated through the sale of rights to enhanced fisheries to aquaculture operators engaged in ranching. However, this would cause conflicts where access to the fishery changed. An alternative strategy would be for the operator to trade-off enhanced nutrient assimilation through ranching against a permit to discharge nutrients, with any surplus being traded with other operators.

Artificial reefs

Employing artificial reefs to remove nutrients from aquatic ecosystems is closely related to ranching as a strategy facilitating horizontal integration in association with open culture facilities. Laihonen, Hänninen, Chojnacki and Vuorinen (1997) outlined the theory of removing nutrients from aquatic ecosystems using artificial reefs. These represent an increased area capable of supporting colonial organisms and other assembelages e.g. grazers and predators. The growth of these assimilates nutrients in biomass that may be harvested, resulting in a net removal of nutrients and output of value from the ecosystem. Laihonen et al. (1997) reported the effectiveness of using a variety of artificial reefs to remove nutrients from the Baltic Sea, proposing that sessile filter-feeding organisms and aquatic plants that are capable of efficient and rapid assimilation of dissolved nutrients are likely to be most promising. These authors used concrete pipes (0.6 m diameter, 1 m long) to construct star, tube and pyramid shaped artificial reefs at depths of 9-12 m in both inshore and offshore locations in Pomerian Bay, Poland. Offshore, colonisation of the star shaped reef was highest at 216,030 individuals m^{-2}; inshore the highest settlement level (112,500 individuals m^{-2}) was observed on the pyramid reef. Mussels (*M. edulis*) and barnacles (*Balanus improvisus*) dominated both reefs. The filtration rate of the organisms colonising the reef was estimated at 5,500 $m^3\ m^{-2}\ y^{-1}$. The authors linked colonisation of the artificial reefs in Pomerian Bay with increased visibility, rising from 2-5 m in June 1991, to 5.5-7 m in October 1993, and decreased nitrate concentrations, falling from 0.5-2 mg l^{-1} to <0.5 mg l^{-1} during the same period. However, data from control sites and monitoring prior to establishing the reefs was not presented, making it impossible to assess the influence of the reefs with respect to regional water quality trends.

Several factors require to be addressed when considering artificial reefs (Laihonen et al., 1997). The physical, chemical and biological conditions at the site must be assessed, temperatures should be high enough to allow a reasonable level of biological activity, light penetration into the water column should be sufficient to allow photosynthesis and nutrients circulating around the reef should be readily bio-available. The reef should also be positioned as close as possible to the nutrient source as the dilution of nutrients will reduce the removal efficiency of the reef system.

The dispersal of waste discharged from open culture systems e.g. cages and pens, is a dynamic process and depends upon a range of factors. The physical properties of the waste, the topography and composition of the substrate underlying the site, tides and currents that affect the hydrology of the site, and the interaction of natural biota all influence the ultimate dispersion pattern. Detailed information is therefore required to produce a reliable model of expected dispersion, thereby allowing the horizontally integrated culture unit or artificial reef to be positioned correctly.

2.7. Evaluating the potential of horizontally integrated aquaculture

Table 2.2 presents a summary of information concerning the range of strategies investigated or demonstrating potential for horizontal integration with semi-intensive and intensive aquaculture. However, there are few examples of full-scale systems associated with commercial aquaculture. Therefore, in addition to the technical constraints, financial, economic, managerial, social, institutional and environmental factors also require consideration. These issues are discussed further in subsequent chapters.

Table 2.2. Appropriate horizontally integrated systems for the aquaculture units described together with constraints and opportunities

Primary aquaculture unit	Secondary integrated unit	Species groups with potential for integration	Constraints	Opportunities
Semi-intensive and intensive ponds, tanks and raceways	Single species flow-through	- bivalve molluscs - phytophagous fish - sea cucumbers - macroalgae - microalgae - floating macrophytes	- fouling with particulate matter - bio-fouling with epiphytes and colonial organisms - predation by fish and shrimps - supply of juveniles for stocking - aeration may be needed at night	- reduced waste load discharged to receiving environment - low-medium engineering demands - potential for shock waste loads during harvest and cleaning
	Multi-species flow-through	- macroalgae & bivalves - macroalgae & herbivorous fish - algae & phytophagous fish - periphyton & fish	- predicting nutrient dynamics - fouling and bio-fouling - finding suitable indigenous species combinations - medium-high engineering demands	- further reduction of waste compared with single species systems - increased opportunities for vertical integration
	Constructed wetlands	- reeds - willow - mangrove ferns - mangroves	- extensive area required - few accounts regarding wetlands used to treat aquaculture wastewater - capacity for nutrient retention potentially limited - limited opportunities to produce valuable crops - sediment accumulation	- appropriate for both freshwater and marine environments - varied treatment processes remove diverse pollutants and increase resilience to environmental perturbations - limited maintenance required
Semi-intensive and intensive cages and pens	Open culture	- macroalgae - bivalves	- difficulty in predicting dispersion patterns for nutrients in dynamic environments - herbivory and predation - loss due to storm damage - seasonal production in temperate climates	- compensates for lack of waste treatment systems developed for open aquaculture - produces oxygen to compensate for the respiratory demands of aquaculture
	Ranching & Artificial reefs	- macrophytes - bivalves - fish - lobsters - crabs	- physical interference with access to the aquaculture system - loss due to predation and poaching - securing sole access to the fishery	- nutrient removal from the system - foraging around reefs enhances assimilation by native communities - enhanced capture fisheries

Chapter Three

Modelling horizontally integrated aquaculture: a smolt farm, constructed wetland and trout fishery

3.1. Introduction

This chapter describes the development of the 'ADEPT' (Aquatic Downstream Ecological Production and Treatment) model that was employed to assess the treatment effect, management demands and financial implications of conventional aquaculture wastewater treatment strategies and horizontally integrated systems. The rationale behind the adoption of a modelling approach is explained and the methodologies used in constructing the model are presented. The majority of studies reviewed in Chapter 2 were conducted as pilot studies and assessed only the fundamental aspects of horizontally integrated aquaculture i.e. nutrient retention, treatment effect, species specific performance and production. Few studies have attempted to evaluate these physical parameters in association with managerial and financial indicators. Studies assessing strategies for horizontal integration employing models have, however, provided an insight to the functioning of different strategies under various operating conditions.

Bodvin, Indergaard, Norgaard, Jensen and Skaar (1996) present a linear model based on mass-balances of N and P in a hypothetical system that integrated the production of salmon, seaweed and mussels. Employing data from field trials, laboratory studies and the literature it was estimated that 112.5 t of mussels would be required to adequately filter 60 m^3 min^{-1} of water used to produce 300 t y^{-1} of salmon; it was estimated that the salmon culture wastewater contained 15 t of N and 2.4 t of P. Considering nutrient dynamics in the mussel culture unit it was predicted that 0.5 t of N would be assimilated and 0.6 t removed through sedimentation; the removal of P via these processes was estimated at 1.2 t. It was estimated that a standing stock of 45 t of seaweed would be required in the final integrated unit to sequester the remaining 13.1 t of N from the mussel culture wastewater, however, at this standing stock only 0.68 t or 57% of the remaining P would be assimilated. Although this approach provides guidance for potential operators of such systems, the sensitivity of the modelled system to different operating conditions was not tested and the model remains to be validated.

Ellner, Neori, Krom, Tsai and Easterling (1996) developed a model using FORTRAN to simulate TAN and total oxidised nitrogen dynamics in a seaweed (*U. lactuca*) biofilter treating water from a recirculating seabream (*S. aurata*) culture unit. These authors found the model able to predict the response of an experimental system (see description of Krom et al, 1995 and Neori et al. 1996, Chapter 2) to changing hydraulic loading rates and feed inputs. Troell and Norberg (1998) constructed a model employing STELLA to simulate the retention by mussels of particles discharged from 68 salmon cages producing 136 t y^{-1}. Based on production data from Chilean salmon farms, values from the literature for the output of particles from cages and an equation for particulate uptake by blue mussels (*M. edulis*), the model predicted the effect of changing fish size (after 92, 183, 274 and 365 days of cultivation), feeding regime (demand or pulsed feeding), and current speeds (0.03, 0.05, 0.1 and 0.15 m s^{-1}) on particulate retention by mussels with a mean individual weight of 25 g, stocked on ropes at a density of 0.5 l^{-1} in a 50 m long culture area. From the various simulations these authors concluded that particulate output from salmon cages would only stimulate mussel growth when ambient seston concentrations were low. However, no fieldwork was undertaken to validate the model.

Findings from a modelling exercise, conducted by Petrell et al. (1993) using a programme written in Turbo-C, to assess the potential of integrating kelp with salmon culture in coastal waters were described in Chapter 2. However, subsequent to this study Petrell and Alie (1996) produced a Quattro Pro spreadsheet-based bioeconomic model for the system, permitting the inclusion of biological, physical and financial variables, which facilitated a simultaneous assessment of productivity, nutrient assimilation and technical and financial viability. Following a comprehensive review of the role of modelling in managing and planning aquaculture systems, Leung and El-Gayar (1997) concluded, that of the approaches considered, "bioeconomic modeling ... offers the greatest potential for the development and planning of sustainable aquaculture."

3.2. Formulating the bioeconomic model

The ADEPT bioeconomic model was developed using [1]Excel, a widely available proprietary spreadsheet package. This application offers a range of facilities such as Boolean logic functions, financial accounting, macros and graphical presentation options that were considered useful in formulating the model and presenting outputs. The first stage was to develop a conceptual framework that outlined the relationship between inputs, stock dynamics, wastewater outputs, treatment, downstream production, costs, returns and financial indicators. This framework was employed as the front page to the model, providing an accessible user interface (Figure 3.1). A flowchart on the front page contains information concerning the function of the main pages in the model; macros associated with each enable navigation to the other pages, printing of selected pages or access to graphs showing model outputs.

Within the *Inputs* page, the user is required to enter basic managerial and financial information that should be readily accessible. Other arrays within the page permit the

[1]Microsoft Excel Office'97 version of the ADEPT model for this case study provided on the disk included.

selection of methodologies for simulating waste outputs, and allow the user to configure the treatment system using any combination of the strategies supported. Input variables concerning the stocking regime, feeding rate and expected FCR's are then imported to the *Stock model* page permitting the mean monthly smolt biomass to be estimated. The stock model, together with data on water temperatures, flow rates and information on the feeding regime and feed composition is used in the *Waste output* page to simulate mean monthly waste discharge concentrations of SS, TAN, BOD, DO and TP in water passing through the culture unit. These waste fractions were selected as they are commonly stipulated for aquaculture discharge consents. The mass balance of TN in the smolt unit and horizontally integrated units was simulated to assess the possibility that it may limit P sequestration.

Mean monthly waste concentrations from the *Waste output* page are imported to the *Treatment* page where, based on the expected flow rate and wastewater composition, the selected treatment strategies are dimensioned. Depending upon the type and order of treatment strategies selected in the *Inputs* page, the model uses a simple sub-routine to configure the required treatment system. The expected performance of these treatment strategies is used to estimate changes in the mean monthly waste concentrations. Where downstream production systems are selected, the *Horizontal integration* page uses either values inputted by the user or pre-defined rates to estimate the productivity or amenity level generated. Using values from the *Inputs* page and, where appropriate, quotes from the literature or key informants, the *Costs* page estimates the financial implications of developing and running the proposed treatment system. Cost estimates are imported to the *Cost return* page and together with estimates of the income generated by activities outlined in the *Horizontal integration* page, are used to calculate key financial indicators such as profit, rate of return on capital and operating costs and payback period. Cost and return data, together with user defined depreciation periods for equipment and infrastructure, are imported into the *Cash flow* page. Where the Net Present Value (NPV) of the investment and Internal Rate of Return (IRR) are calculated. Finally, key indicators from each of the pages are summarised in the *Outputs* page. Further details of the methods and approaches employed in constructing the ADEPT model are described in the following sections.

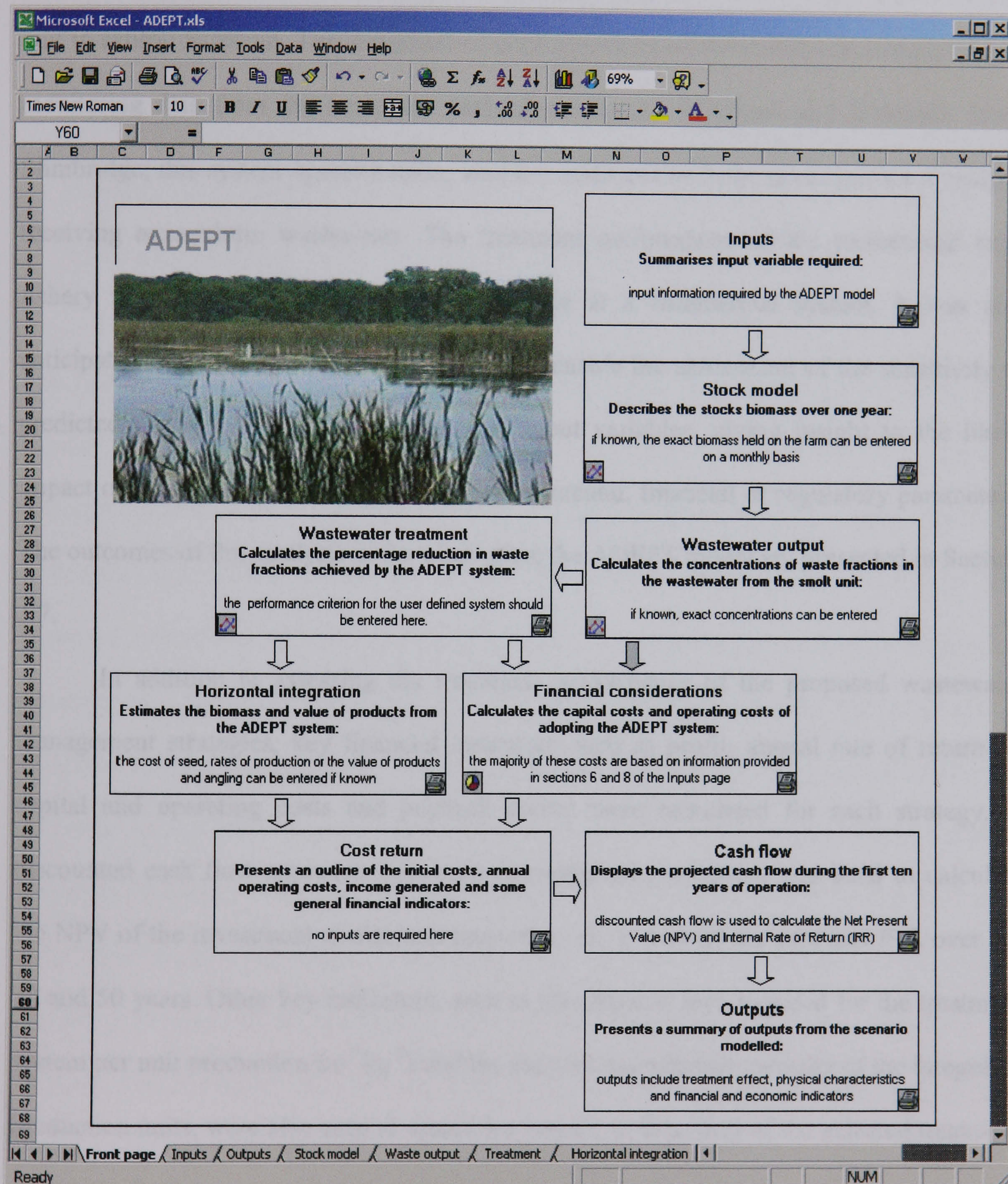

Figure 3.1. Conceptual framework and user-friendly front page for the ADEPT bioeconomic model.

With a suitable level of validation for discharge loads expected under normal operating conditions, it was proposed that the model would constitute a useful tool for simulating the performance of a variety of treatment systems required by the user. Therefore, to test the model, the simulated composition of the smolt unit wastewater and predicted treatment effect of a drumfilter, were compared with samples collected from a commercial culture unit in central Scotland. The simulated reedbed treatment performance was compared with monitoring data from the treatment wetland run by the Wildfowl and Wetlands Trust, Slimbridge; this system operates under similar conditions to those envisaged for a reedbed receiving aquaculture wastewater. The treatment performance of the recreational trout fishery was compared with observations made at a commercial system. It was also anticipated that bioeconomic modelling would enable the assessment of the sensitivity of predicted outcomes to changes in selected input variables, giving insight to the likely impact of revised design, management, environmental, financial or regulatory parameters. The outcomes of this analysis undertaken using the ADEPT model are presented in Section 3.7.

In addition to assessing the treatment performance of the proposed wastewater management strategies, key financial indicators such as profit, annual rate of return on capital and operating costs and payback period were calculated for each strategy. A discounted cash flow approach following conventional procedures was used to calculate the NPV of the investment at discount rates of 5, 10, 15 and 20 per cent and IRR over 10, 20 and 50 years. Other key indicators, such as the physical area required for the treatment system per unit production (m^2 kg^{-1}) and the nutrient assimilation capacity of the integrated production units, were also used to assess the overall performance of the selected treatment strategies. Social, economic and environmental benefits that may be associated with each of the treatment strategies are discussed and potential constraints to development identified.

3.3. Case study development

The ADEPT model was used in this case study to assess the treatment performance, productivity, income generating potential and financial implications of horizontal integration for a smolt farm in Scotland. Three scenarios in which the wastewater stream was derived from a commercial unit producing 100,000 smolts y^{-1} were used:

- a conventional approach to treatment employing a propriety drumfilter,
- treatment with a surface-flow reedbed planted with the common reed (*P. australis*),
- treatment including a recreational trout fishery followed by a reedbed.

The first scenario was selected to establish a baseline as drumfilters are employed on a number commercial smolt farms in Scotland[2]. However, as discussed in Chapter 1, although conventional treatment technologies can remove specific waste fractions, settlement and mechanical filtration are inefficient in removing small particles and dissolved substances, i.e. TAN and ortho-phosphate, from the large volumes of wastewater generated by commercial aquaculture. Therefore, the second scenario was proposed as reedbeds have proved effective for the tertiary treatment and "nutrient polishing" of domestic wastewater. Several studies have focused on exploiting natural processes occurring in aquatic ecosystems such as lagoons, ponds and constructed wetlands to treat these waste fractions. Constructed wetlands planted with common reed have been used for the tertiary treatment of wastewater from several rural sewage works in Scotland and England (Bayes, Bache and Dickson, 1989; Chalk and Wheale, 1989; Upton, Green and Findlay, 1995; Clelland, 1998). Furthermore, constructed wetlands employed by Sansanayuth et al. (1996) and Schwartz and Boyd (1995) to treat wastewater from aquaculture facilities have been described in Chapter 2. Although it was assumed that reed

[2]A hard copy of the ADEPT model outputs for this scenario is given in Appendix 1.

biomass produced in the wetland would have value as an energy crop, it was anticipated that financial returns from this activity would not be sufficient to offset development costs. Therefore, in the third scenario a recreational trout fishery was included to facilitate further cost recovery; diversification in such a way would capitalise on the knowledge and experience of smolt farm managers concerning fish husbandry and water quality management.

Although, lagoons and settlement ponds have been employed to treat wastewater and sludge from a number of commercial aquaculture units (Henderson, Bromage and Watret, 1989; Hennessy, 1991; Chen, Ning and Malone, 1996; Teichert-Coddington et al., 1999), validated design equations based on treatment performance for aquaculture wastewater have not been defined. Therefore, for both the reedbed and trout fishery expected treatment performance levels are estimated using a series of empirical models proposed by Kadlec and Knight (1996) for surface-flow constructed wetlands.

3.4. Simulated commercial smolt production

The first step in applying the model was to establish basic operating parameters for a representative culture facility, including current stocking and feeding regimes, expected FCR's, feed composition data, flow rates and temperatures for abstracted water and baseline financial data. This was achieved through a series of consultations with managers of a farm in central Scotland, as shown in Table 3.1. The model farm produces 100,000 salmon smolts (*S. salar*) annually; process water for the smolt unit is abstracted from a stream at an average rate of 3,400 m^3 d^{-1} and following treatment, wastewater is discharged downstream. This baseline was used in all three scenarios to simulate a realistic stock model and estimate waste concentrations in the used culture water.

3.4.1. Stock model

The stock model estimates the mean weight of fish held at the unit on a monthly basis, enabling the model to simulate the relationship between changing stock biomass and waste output. Information was obtained in preliminary interviews on anticipated biomass at harvest, feeding regimes, feed composition and expected FCR's (Table 3.1).

Table 3.1: Physical parameters and management details for a commercial smolt unit.

Parameter	Value	Units
Stocking date	April	
Culture period	12	Months
Number of fish at harvest (May)	100,000	
Mean weight of individual fish at harvest	70	g
FCR	1.2	
Maximum feed input	4	% body weight d^{-1}
P content of feed	12	g kg^{-1}
N content of feed	72	g kg^{-1}
Dry matter content of feed	950	g kg^{-1}
Average flow rate through the culture facility	3,400	m^3 d^{-1}

Based on the weight of fish at harvest, feeding rates and FCR, the model predicts the biomass on the farm during the culture cycle, thus:

$$b_{(s)} = [(FCR^{-1} \times f \times n)/ b_{(e)} + b_{(e)}^{-1}]^{-1}$$

where,

$b_{(s)}$ = biomass at start of month (kg)

$b_{(e)}$ = biomass at end of month (kg)

FCR = Feed Conversion Ratio

f = feed input (% body weight d^{-1})

n = number of days in month.

Timing of the maximum biomass on the farm was expected to coincide with harvest and was based on current commercial aquaculture practice. Furthermore, mortality at the farm was reportedly <1% and therefore was not considered significant. Based on previous monitoring data provided by the farm manager, the mean monthly temperature of abstracted water flowing into the culture unit was defined within the model (Figure 3.2), values ranged from 3.6°C in January to 14.6°C in August.

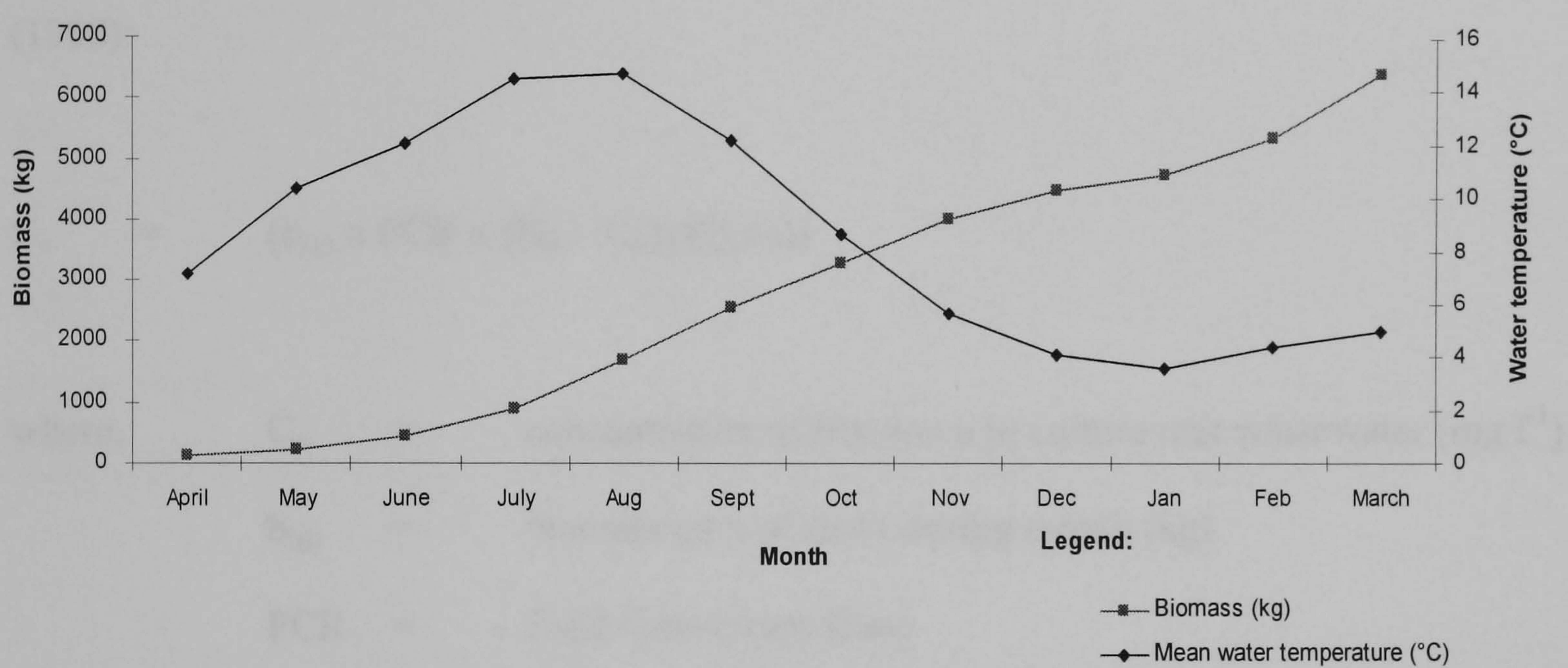

Figure 3.2. Stock model (kg) and mean temperature (°C) of abstracted water

Simulated stock model and temperature regime

Following the traditional production cycle for S_1 smolts, it was assumed that fish were stocked at the beginning of April and harvested the following March. It was anticipated that 100,000 fish would survive to harvest, with an average weight of 70 g, hence a total biomass of 7,000 kg. Adopting these parameters as a baseline, the stock model estimated the mean monthly biomass on the farm during the year. Figure 3.2 shows the change in biomass on the farm during the simulated culture cycle, from 123 kg in April the weight of stock increased gradually until harvest in March; the average biomass of fish held on the farm during the culture period was 2,860 kg.

3.4.2. Wastewater characteristics

Wastewater characteristics were simulated using a range of methodologies, including empirically derived relationships from previous studies, values from the literature and mass balance equations based on feeding rates, expected FCR's and the proportion of solids and nutrients in feed and fish (Table 3.2). In scenarios presented here, changes in SS, TN and TP concentrations in water passing through the smolt unit were estimated using a modified version of the mass balance approach presented by Einen, Holmefjord, Åsgård and Talbot (1995):

$$C_o = (b_{(g)} \times FCR \times (D_n - C_n))/(Q \times d)$$

where,

C_o	=	concentration of fraction n in culture unit wastewater (mg l^{-1})
$b_{(g)}$	=	biomass gain of stock during month (kg)
FCR	=	Feed Conversion Ratio
D_n	=	element n in diet (kg kg^{-1})
C_n	=	element n in carcass of cultured species (kg kg^{-1})
Q	=	average flow rate (m^3 d^{-1})
d	=	number of days in month.

Hennessy, Wilson, Struthers and Kelly (1996) present typical discharge rates for SS (191-606 kg t fish^{-1} d^{-1}), TAN (20-39.3 kg t fish^{-1} d^{-1}), TP (9.1-10 kg t fish^{-1} d^{-1}) and BOD (410 kg t fish^{-1} d^{-1}) from two commercial smolt units in Scotland. These authors also noted that previous studies had not been able to account for more recent changes in feed formulation and husbandry practices such as stocking and feeding regimes. Mass balances allow such changes to be incorporated. Input variables required to formulate the mass balances for SS and TP and simulate changes in TAN, BOD and DO should also be readily accessible to commercial managers. However, where such data are not available, the option exists in the ADEPT model to use stored data from studies published concerning comparable systems (Table 3.2) or for the user to insert data from previous monitoring work.

Table 3.2: Approaches supported by the ADEPT model for simulating discharges from smolt units.

Waste fraction	Literature ($mg\ l^{-1}$)	Stock biomass ($g\ kg^{-1}\ d^{-1}$)	Feed input ($g\ kg^{-1}\ d^{-1}$)	Other ($mg\ kg^{-1}\ min^{-1}$)
SS	*14	†10.8	+Mass balance	
TAN		†0.91	^(N_{feed}-N_{fish}) x 0.8 x 0.8	‡0.03 + 0.08 x G
TP	*0.125	†0.43	+Mass balance	
TN			+Mass balance	
BOD	*8	†17.85	^0.15A	
DO consumption				‡$0.66 \times 10^{(0.063 \times T)}$

‡Bergheim et al. (1991), *Cripps (1994), +Einen et al. (1995), †Hennessy et al. (1996), ^Petit (1989)

The approach proposed by Petit (1989) for determining the output of TAN from salmonid aquaculture was used in this case study, where, of N supplied in feed, but not assimilated by the fish, ~20% is bound in faeces and the remainder excreted in urine or across the gills. TAN constitutes ~80% of this soluble fraction. This author presents a further relationship for BOD output from salmonid culture:

$$\text{BOD} = 0.15A$$

where, A = weight of feed given.

The consumption of DO in meeting the respiratory demand of the stock was simulated using the relationship established by Bergheim, Seymour, Sanni, Tyvold and Fivelstad

(1991) which predicted oxygen consumption for first year smolts based on the ambient water temperature:

$$O_2 = 0.66 \times 10^{0.063T}$$

where, O_2 = oxygen consumption rate (mg kg^{-1} min^{-1})

T = water temperature (°C).

Based on data from the farm managers regarding the mean monthly water temperature in the smolt unit, the DO concentration in water abstracted for the culture system was estimated using the formula given by Elmore and Hayes (1960, cited in Kadlec and Knight, 1996):

$$DO_{sat} = 14.652 - 0.41022T + 0.007991T^2 - 0.00007777T^3$$

where, DO_{sat} = concentration of oxygen in saturated water at 1 atm

T = water temperature (°C).

3.4.3. Predicted wastewater composition

Based on the approaches presented above, the ADEPT model simulated the mean monthly concentrations (mg l^{-1}) of five waste fractions in the 3,400 m^3 d^{-1} of discharged water, the results of which are summarised in Table 3.3, profiles for SS, BOD and DO are presented in Figure 3.3, and those for TAN and TP are shown in Figure 3.4. Kadlec and Knight (1996) described how treatment wetlands have a limited power to remove waste fractions below a certain threshold, commonly referred to as the background concentration (C*); examples are give in Table 3.6. Therefore, to assess the treatment performance of a wetland receiving aquaculture wastewater it is important to consider waste concentrations present in water abstracted for culture purposes. Data from previous monitoring work at the site (Hennessy et al., 1996) was used to simulate background SS, TAN, TP and BOD concentrations, while the expected DO concentrations was estimated using the formula given by Elmore and Hayes (1960, cited in Kadlec and Knight, 1996).

Table 3.3: Mean waste concentrations (mg l^{-1}) observed in abstracted water and predicted for untreated wastewater and water treated using strategies shown; percentage change after treatment is given in parenthesis.

Waste fraction	Abstracted water	Untreated wastewater	Drumfilter	Reedbed	Reedbed and trout fishery
SS	1.75*	6.55	1.18 (-112)	3.08 (-72)	3.03 (-73)
TAN	0.07*	0.25	0.25 (0)	0.09 (-91)	0.09 (-91)
TP	0.00*	0.05	0.02 (-54)	0.02 (-58)	0.02 (-58)
BOD	1.25*	2.42	2.3 (-10)	3.63 (+104)	3.7 (+110)
DO	11.7†	9.46	9.46 (0)	3.7 (-246)	3 (-277)

*Values from previous monitoring work undertaken at the site (Hennessy et al., 1996)
†Based on 100% saturation at 1atm and ambient temperatures given by the farm manager

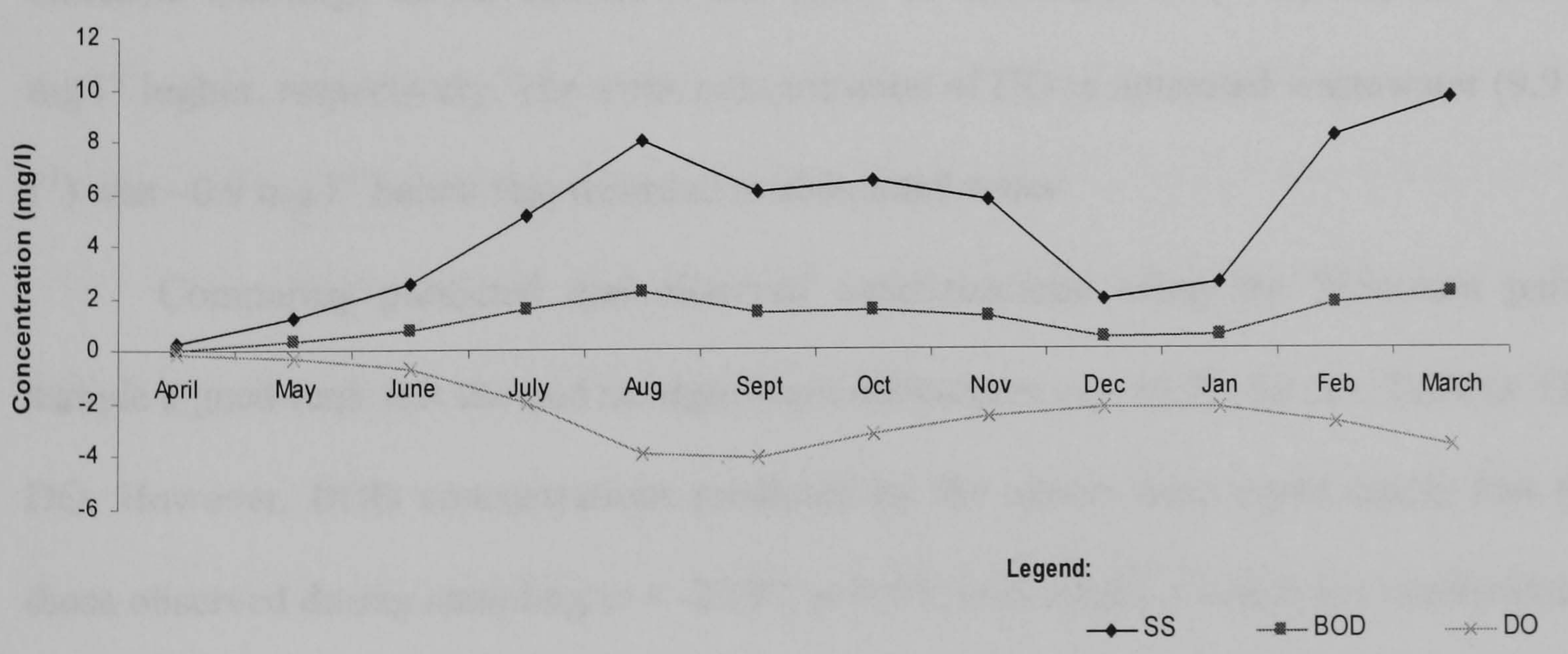

Figure 3.3. Mean change in concentration (mg/l) of SS, BOD and DO in smolt unit wastewater

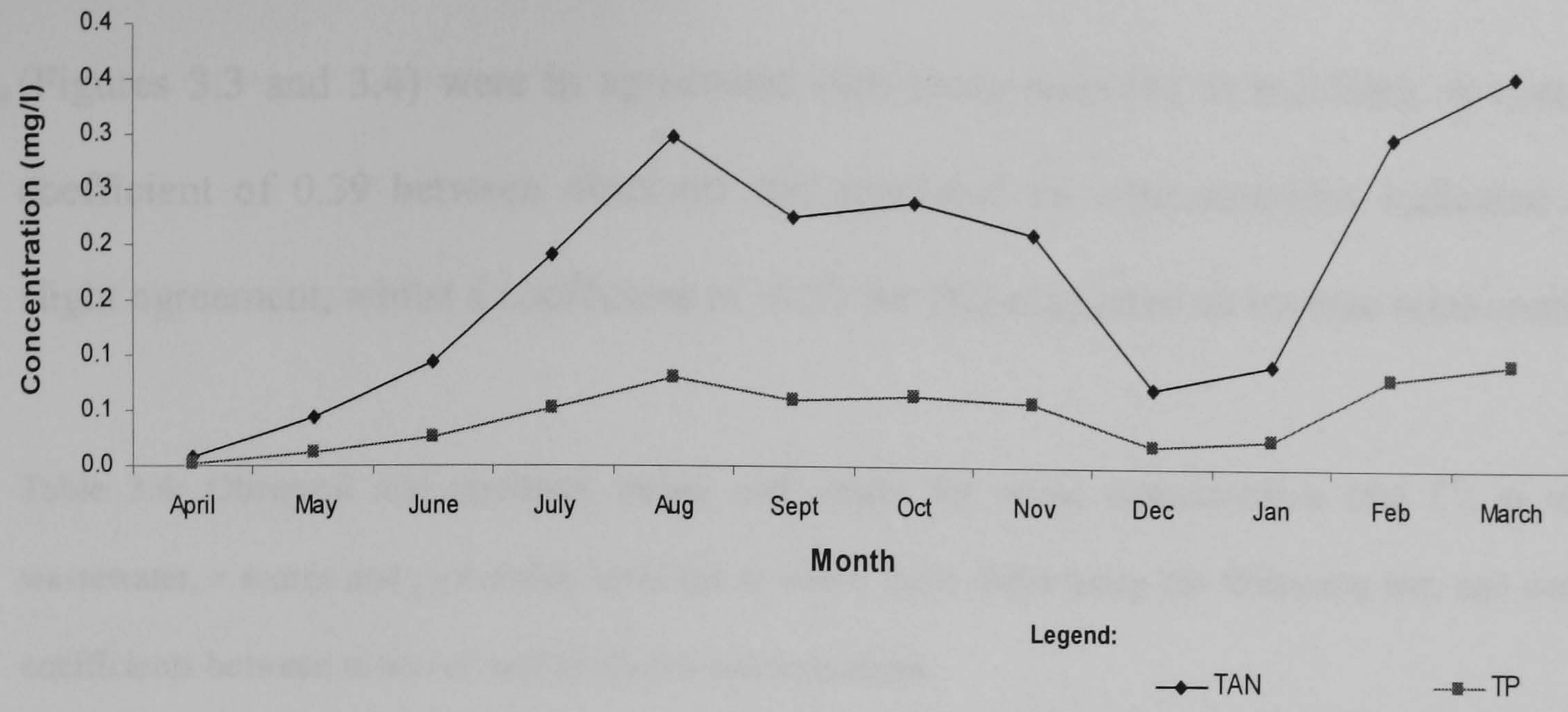

Figure 3.4. Mean change in concentration (mg/l) of TAN and TP in smolt unit wastewater

Model testing: wastewater composition

The composition of wastewater discharged from a commercial smolt unit was assessed with a monitoring programme undertaken between May and October 1998; the results are summarised in Table 3.4. On each occasion 1 litre samples were taken for analysis from the filtered site inflow, smolt tank discharge, drumfilter sump and filtered site outflow. DO and temperature readings were taken *in situ* with a YSI meter. Samples were collected between 10:00 and 12:00 hours, stored in darkness and returned immediately to the laboratory for analysis of SS, BOD, TAN and TP levels, following standard procedures.

Accounting for background concentrations present in abstracted water, the mean untreated discharge concentrations of SS, TAN, TP and BOD were 7.2, 0.3, 0.09 and 2.9 mg l^{-1} higher, respectively. The mean concentration of DO in untreated wastewater (9.9 mg l^{-1}) was ~0.9 mg l^{-1} below that recorded in abstracted water.

Comparing predicted and observed concentrations using the Wilcoxon paired-sample signed-rank test showed no significant differences at p=0.05, for SS, TAN or TP or DO. However, BOD concentrations predicted by the model were significantly less than those observed during sampling (z = -2.197, p<0.05, two-tailed). Correlation coefficients of 0.74-0.77 between observed and predicted concentrations for TAN, TP and BOD demonstrated that trends in the discharge concentrations simulated by the ADEPT model

(Figures 3.3 and 3.4) were in agreement with those recorded at the farm. A correlation coefficient of 0.39 between observed and predicted SS concentrations indicated only a slight agreement, whilst a coefficient of -0.53 for DO suggested an inverse relationship.

Table 3.4: Observed and predicted means and ranges for waste concentrations (mg l^{-1}) in untreated wastewater, *z* scores and probability level (*p*) at which these differ using the Wilcoxon test, and correlation coefficients between observed and predicted concentrations.

Waste fraction	n	Observed		Predicted		Wilcoxon test		Correlation
		Mean	Range	Mean	Range	*z*	*p*	R
SS	9	7.2	0~43.5	5.2	1.2~8	-0.652	.515	0.39
TAN	5	0.3	0.09~0.52	0.17	0.05~0.3	-1.753	.080	0.77
TP	9	0.09	0.02~0.22	0.05	0.01~0.08	-1.599	.110	0.75
BOD	7	2.9	1~7	1.6	0.8~2.3	-2.197	.028	0.74
DO deficit	9	0.86	0.2~1.6	2.55	0.26~4.13	-1.955	.051	-0.53

Poor correlation between observed and predicted SS and DO concentration changes could be explained by the sampling regime adopted. Daily variations in husbandry practices e.g. feeding and cleaning, would cause fluctuations in the discharge of solids and respiratory demands of the stock; this may have masked seasonal trends. Consistently higher observed BOD concentrations, as compared to those predicted, suggest that the approach to modelling BOD output taken from Petit (1989) may not be appropriate. However, this finding could also be a consequence of designing the ADEPT model to calculate average waste flows, whilst sampling was undertaken at discrete intervals and during the day, when waste loads are likely to be higher than average.

3.5. Modelling and testing outputs from the treatment scenarios

Approaches used to simulate the performance of the various treatment strategies are described below and the effect of employing each on the composition of smolt unit wastewater was assessed using the ADEPT model. The predicted treatment performance of each strategy was then tested against observations from systems operating under comparable conditions.

3.5.1. Scenario 1: wastewater treatment with a drumfilter

Predicted treatment performance

Ulgenes (1992b, cited in Cripps and Kelly, 1996) described the treatment performance of a drumfilter; SS removal ranged from 67-97% and TP removal ranged from 21-86%. The mid-point of these ranges was used to estimate treatment performance. Information on removal of TAN and BOD is absent from the literature; however, Bergheim and Åsgård (1996) reported that ~72% of BOD from Atlantic salmon cage farms was associated with particulate matter. Assuming a removal rate for SS of 82%, a BOD treatment efficiency of 59% may be estimated. Furthermore these authors reported that the majority of TAN in aquaculture wastewater is dissolved and therefore, the effect of mechanical filtration is likely to be negligible; it was also assumed that the drumfilter would not affect the DO concentration.

Wastewater treatment with a drumfilter

Average mean monthly concentrations predicted by the model for drumfilter treated wastewater are presented in Table 3.3; profiles for SS, BOD and DO are presented in Figure 3.5, and those for TAN and TP are shown in Figure 3.6. The assumption that the drumfilter removes a fixed proportion of waste fractions regardless of the concentration resulted in post-treatment waste concentrations having similar trends to those of untreated wastewater. Mean concentrations of SS, TP and BOD in filtered wastewater were expected to be 1.18, 0.02, and 2.3 mg l^{-1}, respectively. Concentrations of TAN and DO following treatment were not expected to change.

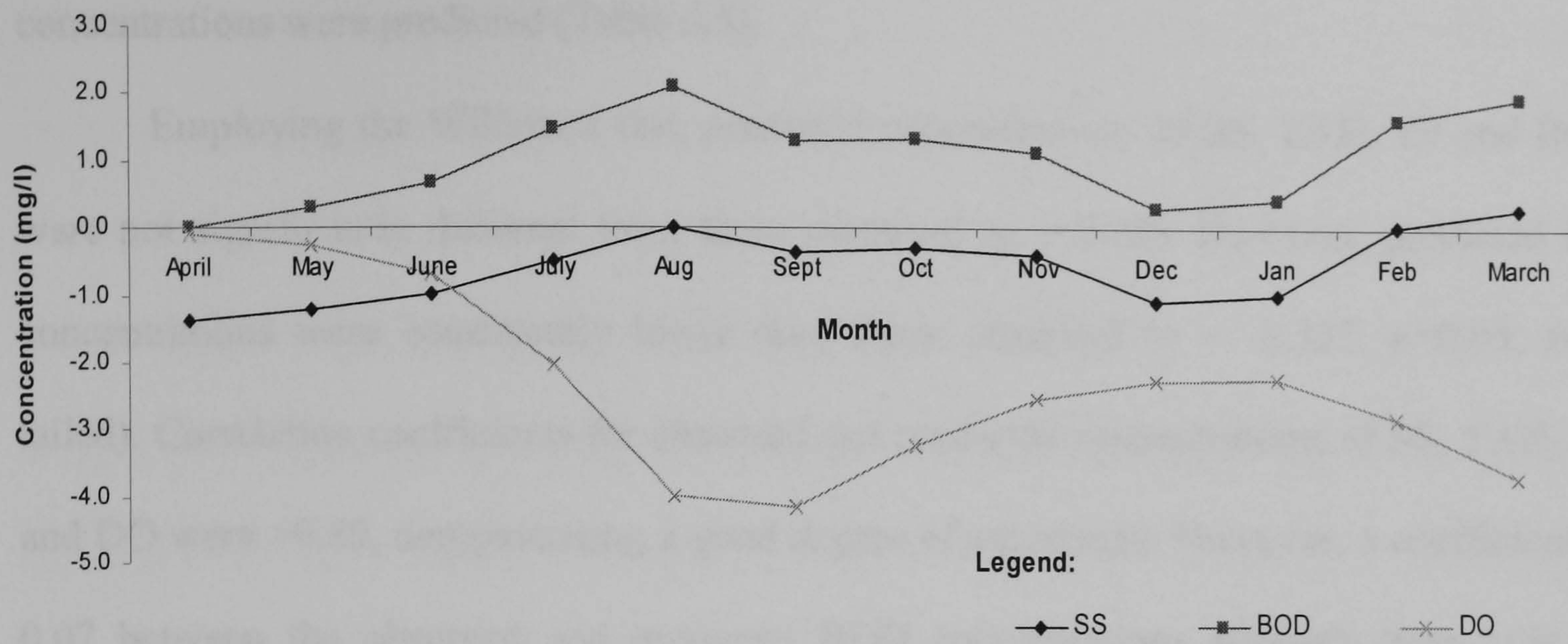

Figure 3.5. Mean SS, BOD and DO concentrations (mg/l) in water treated using a drumfilter (minus background levels)

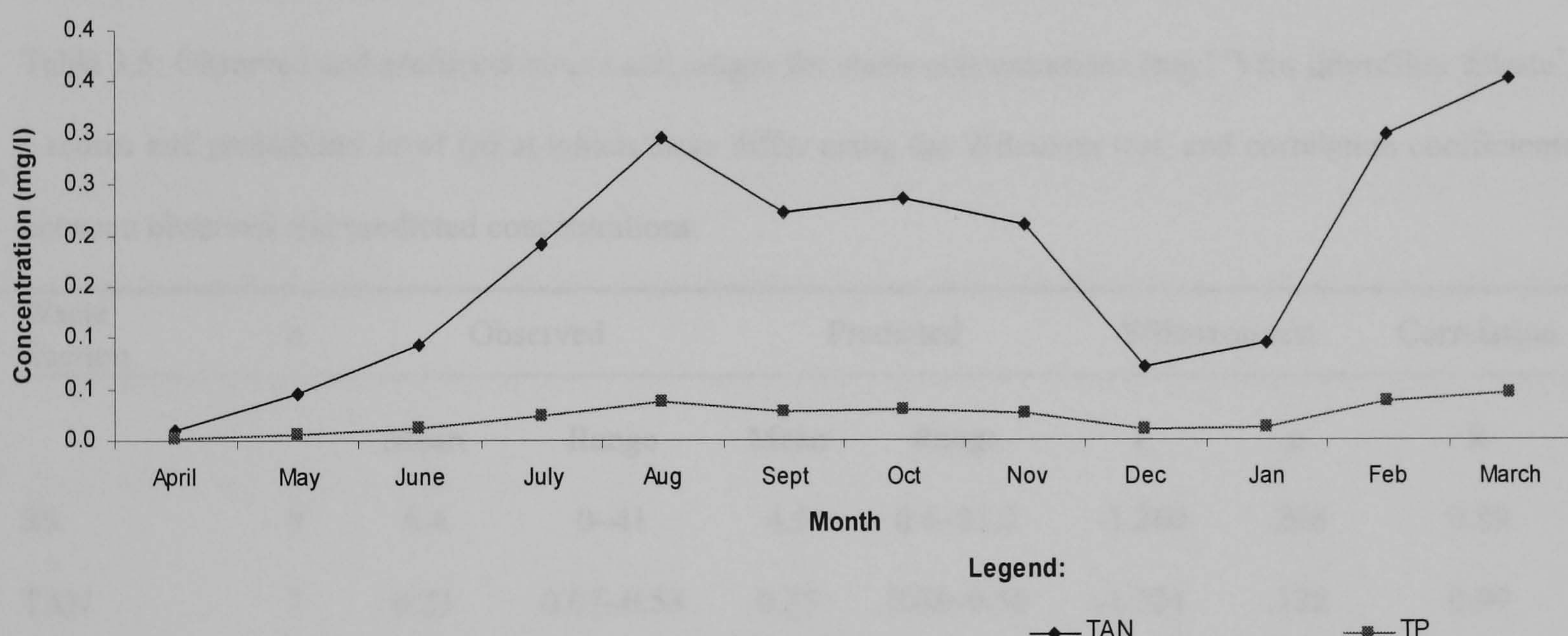

Figure 3.6. Mean TAN and TP concentrations (mg/l) in water treated using a drumfilter (minus background levels)

Model testing: treatment with a drumfilter

The mean and range of waste concentrations recorded in samples of drumfilter filtrate collected during monitoring at the commercial smolt farm are presented in Table 3.5. Mean concentrations of SS, TP and BOD in filtered wastewater were 6.4, 0.17 and 4.5 mg l^{-1}, respectively, 74.3, 44.1 and 25.9 per cent less as compared with untreated wastewater. Mean TAN concentrations were 8.3% lower at 0.23 mg l^{-1}, whilst mean DO levels were 1.7% higher at 10.1 mg l^{-1}. Based on concentrations observed in untreated wastewater, and using the treatment efficiencies used by the ADEPT model, expected post-treatment concentrations were predicted (Table 3.5).

Employing the Wilcoxon test, predicted concentrations of SS, TAN, TP and BOD were not significantly different from those observed (p = 0.05). However, predicted DO concentrations were consistently lower than those observed ($z = -2.327$, $p<0.05$, two-tailed). Correlation coefficients for observed and predicted concentrations of SS, TAN, TP and DO were >0.89, demonstrating a good degree of agreement. However, a coefficient of 0.07 between the observed and predicted BOD concentrations suggests there was no relationship.

Table 3.5: Observed and predicted means and ranges for waste concentrations (mg l^{-1}) for drumfilter filtrate[1], *z* scores and probability level (*p*) at which these differ using the Wilcoxon test, and correlation coefficients between observed and predicted concentrations.

Waste fraction	n	Observed		Predicted		Wilcoxon test		Correlation
		Mean	Range	Mean	Range	*z*	*p*	R
SS	9	6.4	0~41	4.5	0.4~21.2	-1.260	.208	0.89
TAN	7	0.23	0.07~0.58	0.25	0.08~0.56	-1.521	.128	0.99
TP	10	0.17	0.04~1.03	0.18	0.04~0.95	-0.051	.959	0.98
BOD	8	4.5	1.6~9.6	2.3	1.4~3.7	-1.680	.093	0.07
DO	9	10.1	9.2~11.2	9.9	8.9~11	-2.327	.020	0.98

[1]Corrected for background concentrations in abstracted water

According to the Wilcoxon statistic, observed DO concentrations were higher than those predicted; this could have been a result of reaeration due to turbulence caused by the action of the drumfilter. However, as the range of observed and predicted concentrations was similar, and the correlation coefficient shows a very high degree of agreement, the effect of reaeration appears limited. The absence of correlation between observed and predicted

BOD concentrations may be due to the manner in which the treatment effect was estimated; further work is required to evaluate the capacity of a drumfilter to remove BOD under a range of operating conditions.

3.5.2. Scenario 2: treatment with a reedbed

Wastewater treatment in reedbeds involves several processes, e.g. sedimentation, entrapment by plant material, uptake of nutrients by plants and bacteria, decomposition of organic matter, nitrification and the adsorption of ions on soil particles. Watson et al. (1989) reviewed the physical, chemical, biochemical and biological treatment processes acting in wetlands and described the performance of reedbeds operating in a variety of settings (Chapter 2; Table 2.1). Following a comprehensive review of the treatment performance of constructed wetlands, Kadlec and Knight (1996) proposed a general design equation for constructed wetlands relating the area required to reduce a waste fraction to a desired concentration to the flow rate, waste concentration in untreated water, background limit and areal rate constant:

$$A = [0.0365xQ]/k \text{ x } \ln[C_i-C^*]/C_e-C^*$$

where,	A	=	area required (ha)
	Q	=	flow rate ($m^3\ d^{-1}$)
	k	=	areal rate constant ($m\ y^{-1}$)
	C_i	=	concentration of fraction in water entering wetland ($mg\ l^{-1}$)
	C_e	=	concentration of fraction in water exiting wetland ($mg\ l^{-1}$)
	C^*	=	background limit ($mg\ l^{-1}$).

Expected background limits (C*), the lowest concentration to which the constructed wetland is expected to reduce waste fractions, and areal rate constants are presented in Table 3.6.

Concentrations for SS, TAN and BOD stated in the discharge consent standards of the site at 5, 0.1 and 4 mg l^{-1} respectively, are substituted for C_e to dimension the wetland. This estimated the wetland area required to reduce the highest mean monthly discharge concentrations of these waste fractions to discharge consent levels.

Simulated treatment performance

The predicted change in concentration of SS, TAN and BOD is based on k-C* models developed by Kadlec and Knight (1996) for surface flow wetlands. The models have a similar derivation, though the areal rate constant (k) and wetland background limit (C*) vary with the waste fraction concerned (Table 3.6). The general model is:

$$C_o = C^* + (C_i - C^*)\exp\text{-}[kA]/0.0365Q$$

where,

where,	C_o	=	concentration of fraction in water exiting wetland (mg l^{-1})
	C_i	=	concentration of fraction in water entering wetland (mg l^{-1})
	C^*	=	background limit (mg l^{-1})
	k	=	areal rate constant (m y^{-1})
	A	=	wetland area (ha)
	Q	=	flow rate (m^3 d^{-1}).

Table 3.6: Design parameters for constructed wetlands.

Waste fraction	C^*_{20}	C^*	k_{20}	k_T	Ø
SS	$5.1 + 0.16C_i$	$C^*_{20}Ø^{(T-20)}$	1000		1.065
TAN		0	18	$k_T = k_{20}Ø^{(T-20)}$	1.04
TP		0.02	12		
BOD		$3.5 + 0.053C_i$	34	$k_T = k_{20}Ø^{(T-20)}$	1.06
†TN		1.5	22	$k_T = k_{20}Ø^{(T-20)}$	1.05

(compiled from Kadlec and Knight, 1996) †parameters for TN are used in Chapters 4

The expected change in phosphorus concentration is also taken from Kadlec and Knight (1996) and has the form:

$$C_o = C_i \exp(-k/q)$$

where, C_o = phosphorus concentration in water exiting wetland (mg l^{-1})

C_i = phosphorus concentration in water entering wetland (mg l^{-1})

k = areal rate constant (m y^{-1})

q = hydraulic loading rate (cm d^{-1}).

The wetland background limit is expected to be negligible.

DO concentration changes in surface-flow wetlands have been found to depend on levels in water entering the wetland and the areal loading of TAN; Kadlec and Knight (1996) presented the following relationship:

$$C_{DO,out} = 2.52 + 0.126C_{DO,in} - 2.41qC_{NH4\text{-}N,in}$$

where, $C_{DO,out}$ = DO concentration in water exiting wetland (mg l^{-1})

$C_{DO,in}$ = DO concentration in water entering wetland (mg l^{-1})

$qC_{NH4\text{-}N,in}$ = areal loading rate for TAN (g m^{-2} d^{-1}).

In addition to estimating treatment performance using the k-C* models proposed by Kadlec and Knight (1996), the ADEPT model uses a mass-balance equation to assess the capacity of the wetland to assimilate P discharged from the culture unit in aboveground macrophyte biomass. Based on observations from comparable systems in temperate environments and planted with the same reed species, the production rate for aboveground biomass was set at 29 t ha^{-1} y^{-1} (Li, Ding and Wang, 1995) and assumed to contain 11.65 and 2.5 g kg^{-1} of N and P, respectively (Vymazal, 1995).

Wastewater treatment with a reedbed

Based on wastewater discharge concentrations of SS, TAN and BOD from the smolt unit, and existing discharge consent standards, the model estimated that a 5.8 ha constructed wetland would be required, equivalent to 8.3 m^2 kg^{-1} of smolts produced. The reedbed was estimated to have a hydraulic retention time of 4.4 days.

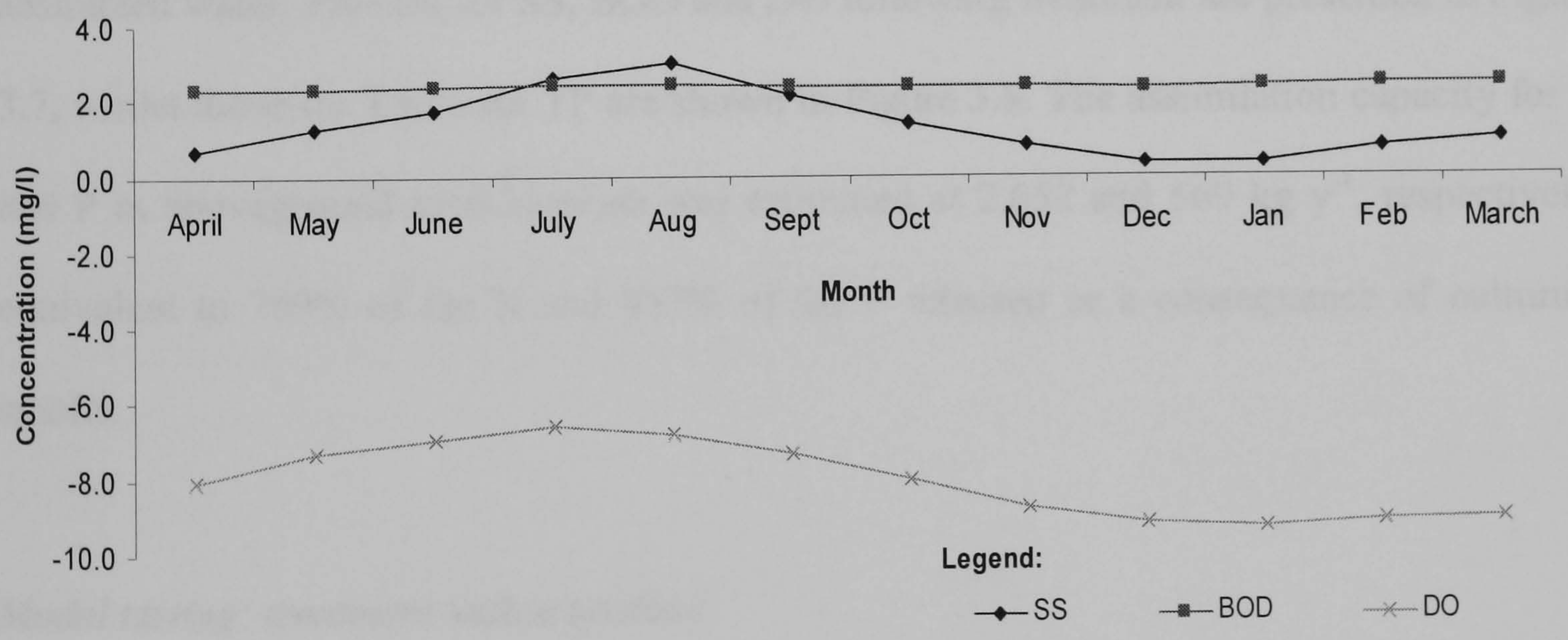

Figure 3.7. Mean SS, BOD and DO concentrations (mg/l) in water treated using a reedbed (minus background levels)

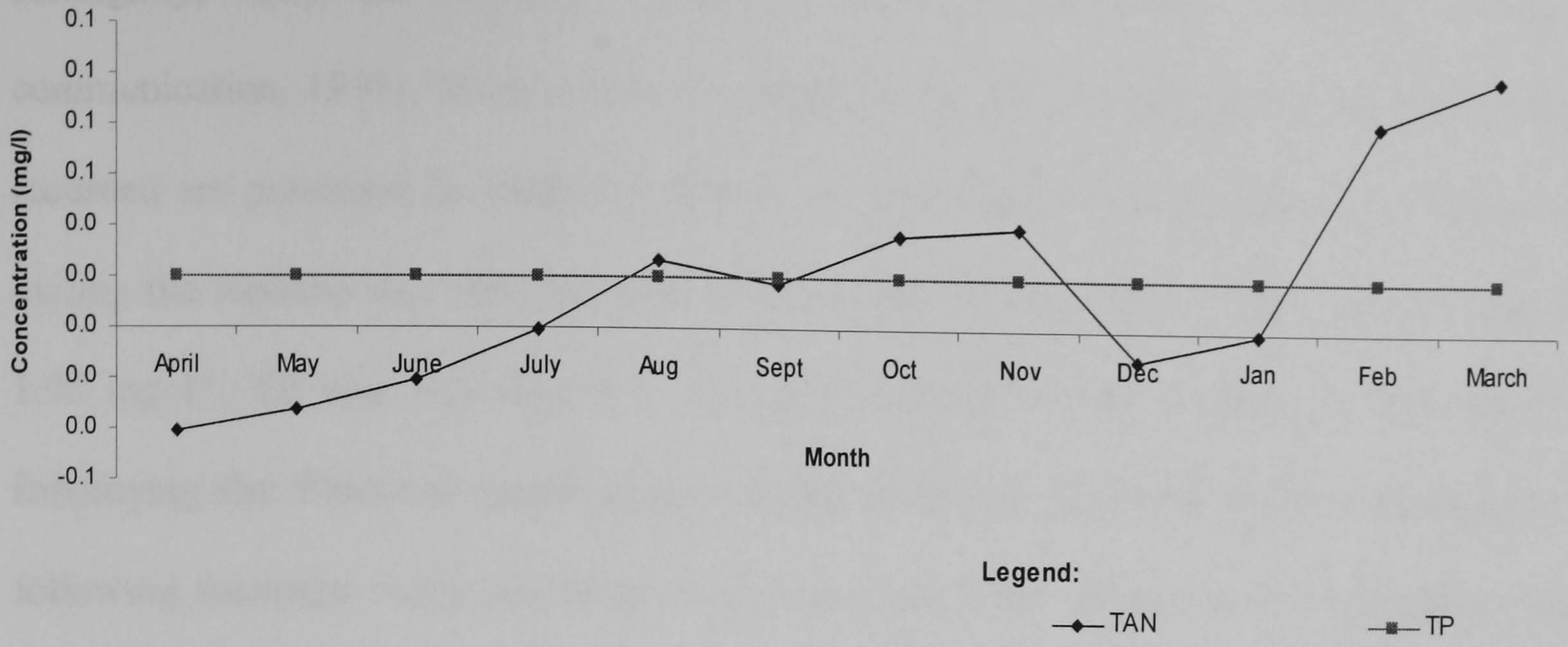

Figure 3.8. Mean TAN and TP concentrations (mg/l) in water treated using a reedbed (minus background levels)

The model predicted that the reedbed would reduce mean discharge concentrations of SS, TAN and TP in smolt unit wastewater by 72, 91 and 58 per cent, respectively (Table 3.3), whilst BOD was expected to increase by 104%. The mean DO concentration was predicted to decrease from 9.3 to 3.7 mg l^{-1}; combined with oxygen consumption predicted in the smolt unit (2.4 mg l^{-1}) this represents a 70% reduction as compared with levels in abstracted water. Profiles for SS, BOD and DO following treatment are presented in Figure 3.7, whilst those for TAN and TP are shown in Figure 3.8. The assimilation capacity for N and P in aboveground reed biomass was estimated at 2,652 and 569 kg y^{-1}, respectively, equivalent to 769% of the N and 957% of the P released as a consequence of culturing smolts.

Model testing: treatment with a reedbed

The predicted treatment effect of the constructed wetland was validated through comparison with the performance of a reedbed planted with common reed operated by the Wildfowl & Wetlands Trust, Slimbridge. The reedbed occupies 0.13 ha and receives 1,000 $m^3\ d^{-1}$ of wastewater (Millett, 1997), a hydraulic loading rate of ~0.8 $m^3\ m^{-2}\ d^{-1}$. Over the period from February 1995 to March 1997 water quality parameters were measured fortnightly, with all analyses following standard procedures (Millett, personal communication, 1999). Mean waste concentrations in pre and post-treatment wastewater recorded are presented in Table 3.7. The mean SS concentration of 12.8 mg l^{-1} in water exiting the wetland was 19% less than that observed in the inflow; TAN was 7% less at 1.95 mg l^{-1}, TP was 10% less at 1.17 mg l^{-1} and BOD was 3% less at 3.63 mg l^{-1}. Employing the Wilcoxon paired-sample signed-rank test, observed waste concentrations following treatment were compared with those predicted using the k-C* models; this showed that predicted concentrations of TAN, TP and BOD did not differ significantly from those observed at a level of $p<0.05$. However, predicted SS concentrations following treatment were significantly lower than that observed (z = -3.779, $p<0.05$, two-tailed). It was not possible to validate the predicted change in DO concentrations, as data from a reedbed operating under comparable conditions were not available. Correlation coefficients between 0.95-0.99 for observed and predicted concentrations of TAN, TP and BOD demonstrated a good degree of agreement. However, a coefficient of 0.12 between the observed and predicted SS concentrations suggested no relationship.

Table 3.7: Observed and predicted means and ranges for waste concentrations (mg l^{-1}) for wastewater treated in a reedbed, z scores and probability level (p) at which these differ using the Wilcoxon test, and correlation coefficients between observed and predicted concentrations.

Waste fraction	n	Observed		Predicted		Wilcoxon test		Correlation
		Mean	Range	Mean	Range	z	p	R
SS	43	12.8	(3.1~42.9)	6.2	(2.4~14.3)	-3.779	< .001	0.12
TAN	33	1.95	(0.04~3.9)	1.97	(0.04~3.7)	-0.853	.394	0.99
TP	22	1.17	(0.02~3.5)	1.23	(0.02~2.8)	-0.800	.424	0.95
BOD	40	3.63	(1.3~14.8)	3.72	(1.9~13.9)	-1.734	.083	0.96

Note: Limited access meant it was not possible to monitor DO concentrations within the wetland.

There was a generally good agreement between observed and predicted concentrations following treatment. However, observed SS concentrations were higher than those predicted and no trend between the two was apparent. This suggests that SS dynamics in the reedbed studied do not conform to the k-C* model proposed by Kadlec and Knight (1996). Potential explanations could include short-circuiting or non-ideal flow regimes causing turbulent resuspension or internal loading due to phytoplankton growth. Also, considering the location of the reedbed, the activity of waterfowl in the wetland could provide another explanation for the higher than expected SS concentrations.

3.5.3. Scenario 3: treatment with a trout fishery and reedbed

Recreational trout fisheries commonly have depths ranging from 1.5-2 m, hydraulic loading rates above ~0.05 $m^3\ m^{-2}\ d^{-1}$ and extend to several hectares (Anderson, personal communication, 1999). With data for the removal of waste fractions in such extensive fisheries absent from the literature, treatment performance rates for surface-flow constructed wetlands described above were employed. Such constructed wetlands commonly have depths <0.6 m, but irrespective of depth it is expected that a similar range of treatment processes would occur in a trout fishery. However, the treatment efficiency and power of the fishery may vary due to rate limiting factors which are a function of depth such as light penetration, diffusion and mixing.

Reviewing design parameters used to engineer constructed wetlands, Watson et al. (1989) recommend hydraulic loading rates of 0.02-0.1 $m^3\ m^{-2}\ d^{-1}$ for surface-flow wetlands used to polish municipal wastewater, comparable to those reported for recreational fisheries. Furthermore, the semi-natural conditions that prevail in many fisheries mean that emergent macrophytes are common around the periphery and on shoals and bars within the water body, and this vegetation may facilitate a number of the treatment processes associated with constructed wetlands (Chapter 2; Table 2.1). Therefore, a secondary objective of this study was to test the validity of the k-C* models proposed by Kadlec and Knight (1996) for surface-flow constructed wetlands in predicting waste removal rates in extensive trout fisheries, and highlight opportunities to develop these models to describe the treatment potential of such systems.

Wastewater treatment with a trout fishery and reedbed

Based on the expected wastewater flow of 3,400 $m^3\ d^{-1}$, the model calculated that a 6.8 ha trout fishery was required to achieve the desired hydraulic loading rate (0.05 $m^3\ m^{-2}\ d^{-1}$). Furthermore, assuming the treatment effect of the trout fishery conforms to the k-C* models for surface-flow wetlands, a 9.2 ha reedbed would be needed to reduce SS, TAN and BOD concentrations to discharge consent levels. The total surface area of the trout fishery and reedbed would be 16 ha, equivalent to 22.9 $m^2\ kg^{-1}$ of smolts produced. The hydraulic retention time of the system was estimated at 47 days.

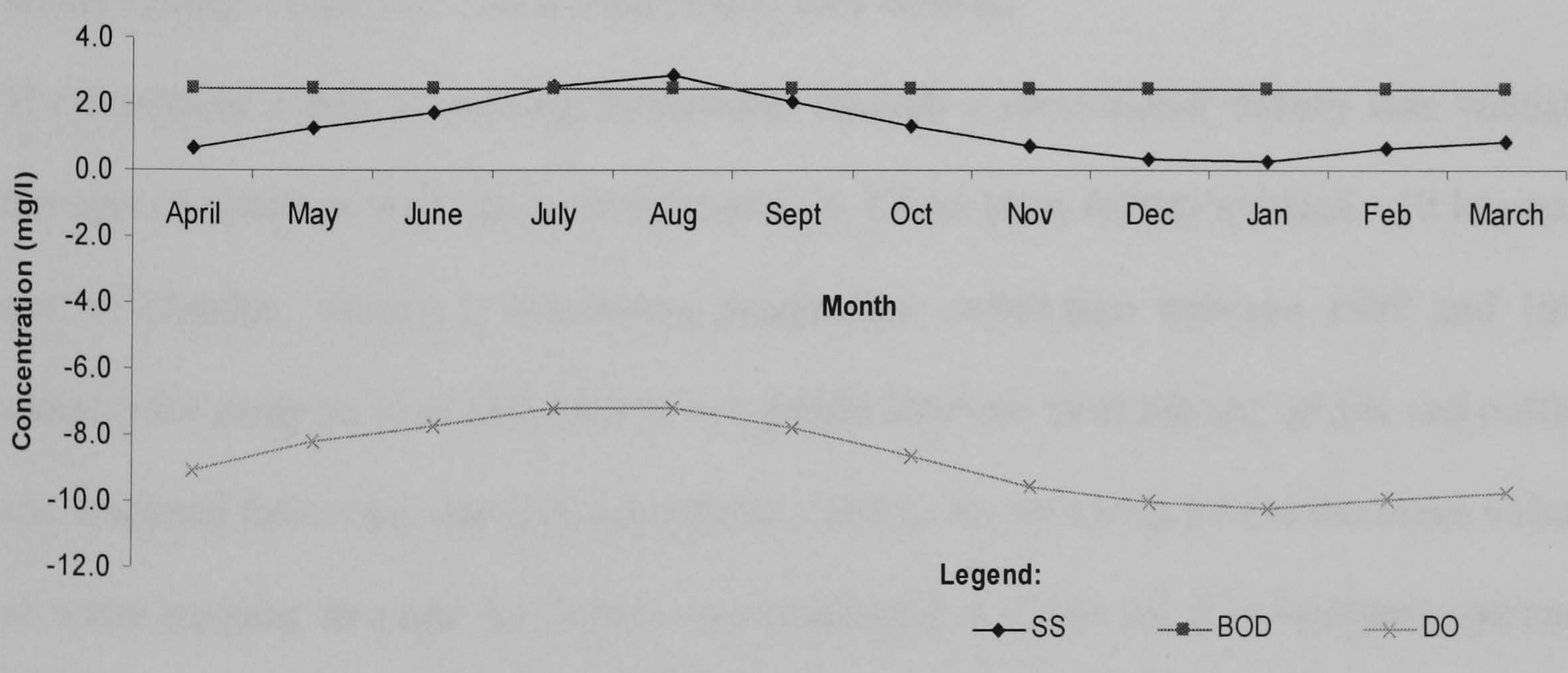

Figure 3.9. Mean SS, BOD and DO concentrations (mg/l) in water treated using a trout fishery and reedbed (minus background levels)

The model predicted this treatment strategy would reduce mean SS, TAN and TP concentrations by 73, 91 and 58 per cent, respectively (Table 3.3), with BOD increasing by 110%. The mean DO concentration would be reduced from 9.3 to 3 mg l^{-1}, only 26% of the predicted level in abstracted water. Profiles for SS, BOD and DO following treatment are presented in Figure 3.9, whilst those for TAN and TP are shown in Figure 3.10. The capacity for N and P retention in aboveground reed biomass was estimated at 2,147 and 461 kg y^{-1}, equivalent to 623% and 775%, respectively, of the output from the smolt unit.

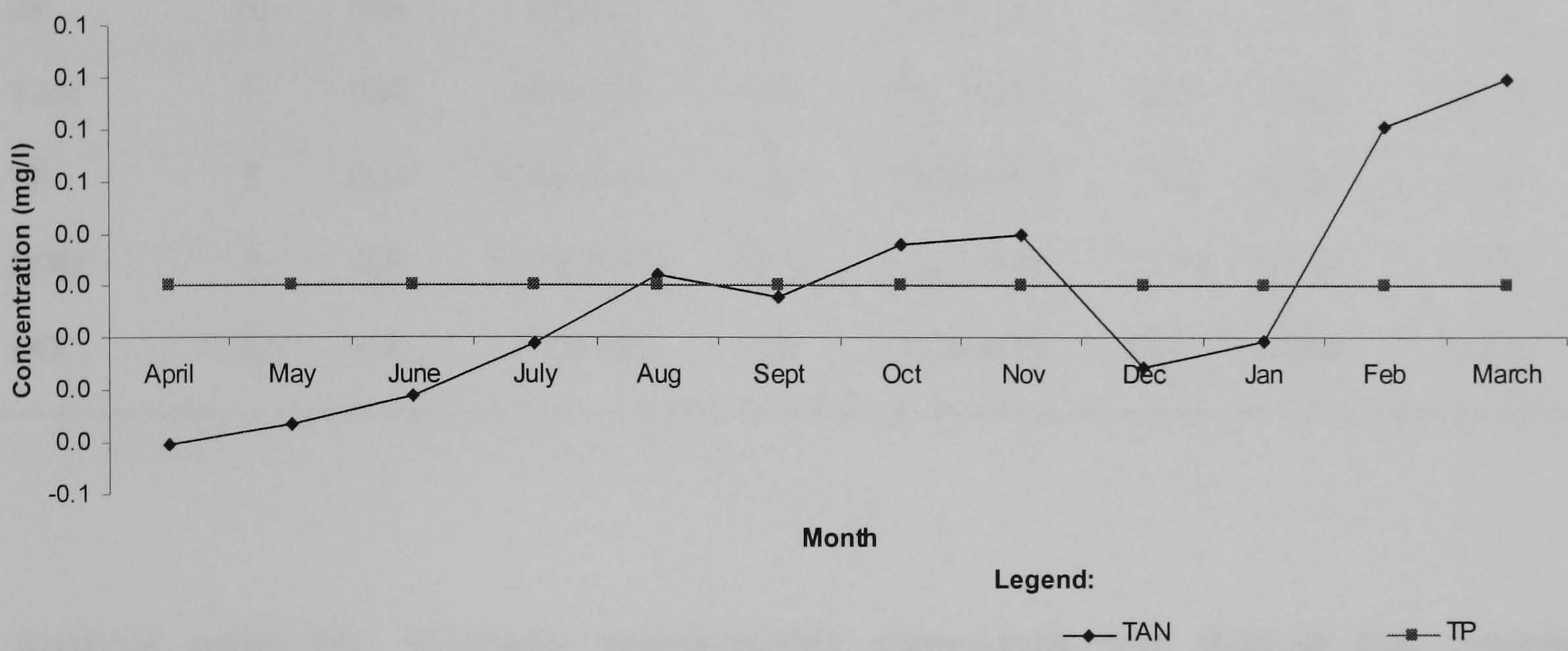

Figure 3.10. Mean TAN and TP concentrations (mg/l) in water treated using a trout fishery and reedbed (minus background levels)

Model testing: treatment with a trout fishery and reedbed

The treatment effect of passing wastewater through a recreational fishery was validated through comparison with the performance of a 2.4 ha trout fishery situated ~10 km north east of Dundee. During a monitoring programme undertaken between 1997 and 1999, samples for analysis were collected at two month intervals from the site inflow and outflow and analysed following standard procedures. During the sampling period the mean volume of water passing through the fishery was estimated at 2,500 m^3 d^{-1} (Anderson, personal communication, 1999), equating to a hydraulic loading rate of ~0.1 m^3 m^{-2} d^{-1}.

Mean waste concentrations observed in the fishery discharge are presented in Table 3.8; TAN was 52% lower at 0.04 mg l^{-1} and TP was 40% less at 0.34 mg l^{-1}. Mean SS and BOD concentrations increased by 170% and 44% to 6.75 and 2.86 mg l^{-1}, respectively, whilst DO concentrations were on average 12% lower at 9.63 mg l^{-1}.

Table 3.8: Observed and predicted means and ranges for waste concentrations (mg l^{-1}) for water having passed through a trout fishery, z scores and probability level (p) at which these differ using the Wilcoxon test, and correlation coefficients between observed and predicted concentrations.

Waste fraction	n	Observed		Predicted		Wilcoxon test		Correlation
		Mean	Range	Mean	Range	z	p	R
SS	10	6.8	(0~32)	3.2	(2.1~4.6)	-.764	0.445	0.06
TAN	7	0.04	(0~0.17)	0.06	(0.01~0.27)	.000	1.000	-0.14
TP	5	0.34	(0.19~0.45)	0.18	(0.08~0.3)	-2.023	0.043	0.95
BOD	9	2.9	(1.35~5.95)	2.7	(1.3~3.9)	-0.296	0.767	0.73
DO	10	9.6	(7.9~12)	3.9	(3.6~4.1)	-2.803	0.005	0.60

Analysis using the Wilcoxon paired-sample signed-rank test showed that recorded concentrations of SS, TAN and BOD were not significantly different from those predicted using the k-C* models for surface-flow wetlands. However, predicted TP concentrations were significantly lower than those observed (z = -2.023, $p<0.05$, two-tailed) as were DO concentrations (z = -2.803, $p<0.05$, two-tailed). A correlation coefficient of 0.95 between observed and predicted TP concentrations demonstrated a good degree of agreement, whilst coefficients of 0.73 and 0.6 for BOD and DO indicated a reasonable level of agreement. Coefficients of 0.06 and -0.14 for observed and predicted SS and TAN concentrations suggested no definable relationship.

Generally, agreement between observed and predicted concentrations was poor, suggesting that with the exception of BOD, the k-C* models for surface-flow wetlands were not appropriate in this context. Poor correlation between observed and predicted SS concentrations may be due to internal loading by phytoplankton, and although the k-C* model accounts for this phenomena to an extent, it is likely to be more significant in an open-water system. TAN concentrations also showed poor correlation, suggesting the need to develop a refined k-C* model for open water wetlands, or to investigate alternative approaches. Relationships established for sewage oxidation ponds and wastewater treatment lagoons may be useful (for example see Benfield and Randall, 1980; Mara, 1997). However, the more concentrated nature of effluents treated in such systems may mean that design equations and expected treatment performance levels may not be valid when considering the relatively dilute discharges that characterise aquaculture wastewater. Although trends were evident in the observed and predicted TP and DO concentrations, the magnitude of change differed, again suggesting the need to refine the k-C* models employed. Refining these models will require data from further comparable systems to ensure robust outputs and facilitate validation.

3.6. Financial implications

Cash flows associated with the various treatment strategies were assessed using conventional financial appraisal methods. Financial requirements and income levels associated with horizontally integrated aquaculture are difficult to predict and published accounts are limited, therefore, where examples from the literature were unavailable, estimates were derived from consultations with the managers of comparable systems (Table 3.9). Expected variable construction costs included the value of land required, site development and infrastructure, variable operating costs were associated with electricity and contracted labour requirements, whilst wages paid to full-time staff and managers constituted the fixed operating costs. The depreciation period for infrastructure and equipment was assumed at 15 and 10 years, respectively; the salvage value of equipment and infrastructure was assumed to be zero. In the baseline scenario land was also assigned a salvage value of zero as the cost of wetland reclamation could prohibit future uses; the influence of this decision was tested in the sensitivity analysis. The annual cost of maintaining the site and infrastructure was estimated at 1% of initial capital costs.

Income from the reedbed was estimated from the value of aboveground biomass produced as a biofuel, and a representative production level of 29 t ha^{-1} y^{-1} (Li et al., 1995). The value of reed biomass was estimated from preliminary data presented by Björk and Granéli (1978), with 2 t of reed biomass energetically equivalent to 1 m^3 of commercial heating oil. Assuming heating oil costs £100 m^{-3} (Butler Fuels, personal communication, 2000) and disregarding handling and processing costs, the value of reed biomass is equivalent to £50 t^{-1}. The cost of harvesting the reed biomass, £41 ha^{-1} y^{-1} (£1 equivalent to €1.5) was based on engaging a contractor with a single-phase self-propelled forage maize harvester (Walsh, 1999). This strategy has been recommended for harvesting *Miscanthus* spp. grown as an energy crop.

Table 3.9: Financial parameters and baseline assumptions employed.

Parameter	Value	Units	Source
Land	7,400	£ ha^{-1}	Millet (pers. comm. 1999)
Development cost for wetlands	49,500	£ ha^{-1}	Millet (pers. comm. 1999)
Drumfilter (167 m^3 h^{-1}) running costs	500	£ $filter^{-1}$ y^{-1}	Murray (pers. comm. 1999)
Reed biomass harvesting	41	£ ha^{-1} y^{-1}	Walsh (1999)
Workers labour	10	£ h^{-1}	Millet (pers. comm. 1999)
Managers labour	20	£ h^{-1}	Millet (pers. comm. 1999)
Reed production	29	t ha^{-1} y^{-1}	Li et al. (1995)
Energy content of reed biomass	5	MWh t^{-1}	Björk and Granéli (1978)
Heating oil	100	£ m^{-3}	Butler Fuels (pers. comm. 1999)

Financial aspects of developing and operating the recreational trout fishery were obtained by consulting with the managers of a commercial operation; details are presented in Table 3.9. The fishery is initially stocked with trout costing £2 each, at a rate of 0.1 m^{-2}; it was anticipated that 1 ha of trout fishery would attract 2 anglers a day, and that each angler would remove two fish that must be replaced. The value of angling provided was estimated at £25 per day. As described above, conventional financial indicators were employed to evaluate the management strategies selected here for a commercial smolt farm; the results of this analysis are presented in Table 3.10.

3.6.1. Scenario 1: drumfilter

Based on fixed and variable construction and operating costs assumed for the scenario (Table 3.9) a capital cost of £16,000 is incurred for a drumfilter and associated infrastructure, details are presented in Table 3.10. The largest cost incurred is £10,000 for the drumfilter, the instillation of which requires an area of 50 m^2. Other costs incurred include site development at £260, electricity supply at £500, a settling tank and sludge storage tank at £1,500 each, a sludge pump at £200, hydrological control structure at £2,000 and £40 for the land.

Operation of a drumfilter was estimated to require 91 h y^{-1} for maintenance and generate 13 h y^{-1} of administrative work. Variable operating costs included £250 y^{-1} for electricity, £910 y^{-1} for labour and £160 y^{-1} for site and infrastructure maintenance; the only fixed operating cost was £260 y^{-1} for administration. Total annual operating costs were £3,070 y^{-1} or £1,580 y^{-1} excluding depreciation. As no income would be generated, the net benefit associated with running the drumfilter was negative at -£2,080 y^{-1}. Employing a discounted cash flow approach and following conventional procedures, 10 year NPV's at discount rates of 5, 10, 15 and 20 per cent, are -£26,900, -£23,100, -£20,400 and -£18,200, respectively. The IRR is negative over ten, twenty and fifty years of operation.

3.6.2. Scenario 2: constructed wetland

Capital costs associated with developing the reedbed were established through consultation with the manager of a recently commissioned system (Millet, personal communication, 1999). Typical costs included land acquisition, basin construction, purchasing and planting reeds and hydrological control structures. Total capital costs for the 11.4 ha reedbed were estimated at £681,940; individual costs are presented in Table 3.10. Variable operating costs were £6,820 for site maintenance and £470 for a contractor to harvest the reeds. Fixed operating costs were restricted to £260 for administration time required by a manager.

3.10: Financial implications and key indicators of performance for the treatment strategies indicated.

Cost		Drumfilter	Constructed wetland	Trout fishery and constructed wetland
Capital costs				
Land		£40	£88,400	£124,400
Site development		£260	£591,510	£832,400
Infrastructure and equipment		£15,700	£2,000	£4,500
Stocking				£13,600
Total		£16,000	£681,940	£974,940
Operating costs				
Electricity		£250		
Labour		£910		
Contractor			£470	£380
Stock				£38,950
Maintenance		£160	£6,820	£9,750
Fixed labour costs		£260	£260	£19,260
Total		£1,580	£7,550	68,340
Income				
	Reed biomass		£10,240	£8,290
	Angling			£243,440
Profit excluding depreciation		-£1,600	£2,700	£183,400
Rate of return on capital costs (%)		-9.9	0.4	18.8
Rate of return on operating costs (%)		-100	36	268
Payback period (y)		na.	253	5.3
NPV at:	5%	-£25,900	-£631,200	£313,000
	10%	-£22,800	-£605,800	£73,900
	15%	-£20,500	-£581,800	-£86,800
	20%	-£18,600	-£559,200	-£196,400
IRR (%) over:	10 years	na.	na.	12.1
	20 years	na.	na.	18
	50 years	na.	na.	18.8

The value of 228 t of reed biomass produced annually was estimated at £10,240 y^{-1}, resulting in a net benefit excluding deprecation of £2,700 y^{-1} or £227 ha^{-1} y^{-1}, and an annual rate of return of 0.4% on initial capital costs and of 36% on operating costs. The payback period was calculated at 253 years. At discount rates of 5, 10, 15 and 20 per cent, ten year NPV's were -£631,200, -£605,800, -£581,800 and -£559,200, respectively, whilst the IRR over ten, twenty and fifty years was negative (Table 3.10).

3.6.3. Scenario 3: a trout fishery and constructed wetland

Development of the combined reedbed and trout fishery was estimated to incur a capital cost of £974,940, including £124,400 for land, £832,400 for site preparation and construction, £3,000 for water control structures, £13,600 for initial stocking and £1,500 for fish handling equipment. With an area >3 ha, no variable labour costs are incurred as all necessary work was included in the duties of the full-time fisheries manager, who would also undertake all administrative work. Operating costs were therefore estimated at £68,340 y^{-1}; variable operating costs included £9,750 y^{-1} for site maintenance, £380 y^{-1} for a contract reed harvester and £38,950 y^{-1} to purchase fish. The only fixed operating cost was £19,260 y^{-1} for a fishery manager.

An income of £251,730 y^{-1} was estimated, with £8,290 y^{-1} and £243,440 y^{-1} realised through the sale of 184 t of reed biomass and provision of 9,737 angling days, respectively. The estimated net profit, excluding depreciation, was £183,400 y^{-1} or £10,900 ha^{-1} y^{-1} from the land occupied by the system, representing a rate of return of 18.8% on capital costs, 268% on operating costs and a payback period of 5.3 years. The ten-year NPV remained negative at discount rates of 15 and 20 per cent, but was £73,900 at 10% and £313,000 at 5%. The IRR over ten, twenty and fifty years was 12.1, 18 and 18.8 per cent, respectively.

3.7. Sensitivity analysis

The sensitivity analysis was achieved by modifying key input parameters in turn and assessing the impact on the ten year IRR of the system in question; results are summarised in Table 3.11. The drumfilter was excluded from this analysis, as it was clear that this strategy was unable to generate a positive IRR. Considering the reedbed scenario, 10% changes in the range of selected input variables failed to generate a positive rate of return; only when it was assumed that the land and earthworks would retain their value was an IRR of 0.4% calculated.

Table 3.11. Sensitivity of ten year IRR (%) to changing input parameters.

Parameter	Reedbed	Trout fishery and reedbed
Baseline	-	12.1
Land cost (-10%)	-	12.4
Land cost (+10%)	-	11.7
Land retains its value over ten years	-	13.2
Land and earthworks retain value over ten years	0.4	18.7
Land cost assumed at £0 ha^{-1}	-	16
Site development cost (-10%)	-	14.6
Site development cost (+10%)	-	9.8
Value of reed biomass (-10%)	-	11.9
Value of reed biomass (+10%)	-	12.2
Value of angling (-10%)	-	8.5
Value of angling (+10%)	-	15.5
Smolt numbers increased by 50%	-	6.4
Smolt number decreased by 50%	-	28.3
Water used decreased by 50%	-	0.9
Water used increased by 50%	-	21.3

A 10% change in land or reed biomass values had a minimal effect on the IRR generated by the trout fishery and reedbed combined. A 10% fall in site development costs or 10%

rise in the value of angling had a positive effect, raising the IRR to 14.6% and 15.5%, respectively. Assuming land retains its value generated a slight increase in the IRR to 13.2%, whilst assuming the initial land cost was zero increased this to 16%. Assuming both the land and earthworks retained their value resulted in a ten year IRR of 18.7%. However, the most notable change occurred when significant modifications to the operation of the smolt unit were assumed. A 50% increase in water use resulted in an IRR of 21.3% whilst a comparable fall resulted in an IRR of 0.9%. Increasing smolt numbers by 50% resulted in an IRR of 6.4%, whilst a fall of 50% resulted in an IRR of 28.3%.

3.8. Conclusions

The managers of a commercial farm (Murray, personal communication, 1999) considered the stock data predicted by the ADEPT model to be realistic. Average mean discharge concentrations (mg l^{-1}) of the various waste fractions estimated by the model for the smolt unit are comparable to those in the literature (Cripps, 1994). Furthermore, excluding BOD, ADEPT model estimates of mean monthly concentrations showed no significant differences from levels observed in wastewater samples taken from a commercial smolt farm. This demonstrated that the model was suitable for simulating waste concentrations discharge from smolt farms. Similar approaches to modelling wastewater discharged from commercial aquaculture operations have been described (Beveridge et al., 1991; Foy and Rosell, 1991; Einen et al., 1995; Eikebrokk, Piedrahita and Ulgenes, 1995). Foy and Rosell (1991) modelled the expected discharges from a trout farm in Ireland and found that employing a mass-balance approach it was possible to account for 89% and 103% of the observed discharges of N and P, respectively.

Concentrations of waste discharged from commercial aquaculture sites have been found to follow diurnal trends (Bergheim et al., 1991; Kelly and Karpinski, 1994; Hennessy et al., 1996) and exhibit peaks associated with feeding and cleaning, representing a possible limitation for the ADEPT model. However, when modelling large engineered systems such as constructed wetlands and trout fisheries, diurnal trends are likely to become less distinct due to mixing and the long residence times introduced. Problems may arise in modelling treatment systems that retain wastewater for limited periods e.g. drumfilters and settlement ponds. Higher waste concentration pulses following feeding, cleaning tanks or handling fish may adversely affect the performance of drumfilters as screens may become blocked, causing untreated wastewater to bypass the filter. Although the model does not simulate variations in treatment efficiency due to changes in waste loading, predicted concentrations of SS, TAN, TP and BOD treated using a drumfilter did not differ significantly from those observed. It was assumed that the drumfilter would have no effect on DO concentrations in filtered wastewater, but comparative results showed there was a consistent increase, thus suggesting the need to refine the ADEPT model.

The design strategy adopted for surface flow reedbeds in the integrated systems was intended to ensure that discharge achieved 100% compliance with discharge consent standards. However, compared with wastewater treated in the majority of constructed wetlands, aquaculture wastewater is relatively dilute. This may be an important factor as where waste concentrations are below the background concentration (C*), the use of reedbeds for treatment may be constrained. Furthermore, the k-C* models predicted that constructed wetlands may contribute to the waste load discharged to the receiving environment during periods when waste concentrations from the smolt unit are low.

Despite these possible limitations, the model predicts that concentrations of SS, TAN and BOD in discharges in the two scenarios incorporating wetlands achieve a 100% level of consent compliance, avoiding fines or legal action, or reducing tax liabilities were a pollution tax to be levied on waste discharges. Operators of the farm could also promote an environmentally friendly image, possibly leading to a premium for fish produced. The mean DO concentration (3.7 mg l^{-1}) predicted for the reedbed discharge in the baseline modelling exercise may represent a cause for concern, however, it was not possible to validate this output. Further work is required to address this potentially serious constraint to using constructed wetlands for aquaculture wastewater treatment. One potential solution could be to position the trout fishery after the reedbed, as DO levels observed at the fishery were significantly above those expected based on the k-C* model for surface-flow wetlands. However, problems with elevated SS concentrations have constrained using open water systems in the final stages of ecologically based treatment systems (Kadlec and Knight, 1996). Furthermore, due to the poor association between the observed treatment efficiency and that predicted using the k-C* models for surface flow wetlands, additional work is required to assess the potential of trout fisheries as a viable treatment option. Such a study should consider the treatment performance of other fisheries and comparable systems, functioning under a range of operating conditions, to develop appropriate and robust models.

Constructed wetlands have proved effective at treating wastewater from a wide range of sources (Kadlec and Knight, 1996). However, the long-term storage of nutrients and other pollutants has not been studied widely. Phosphorus entering a constructed wetland is removed initially through adsorption on substrate particles and assimilation in plant and microbial biomass (Watson et al., 1989). However, Richardson (1985) noted, "wetlands tested as wastewater filtration systems became phosphorus-saturated in a few years, with the export of excessive quantities of phosphate". Furthermore, plant communities that are not harvested, naturally senesce, releasing significant amounts of both nitrogen and phosphorus (Landers, 1982). The accumulation of litter may provide additional binding sites, prolonging the effective life of a wetland. Eventually, however, accumulated sediments will require removal to maintain the hydraulic characteristics of the wetland, and there is a risk that the inappropriate disposal of these sediments may create a diffuse pollution source. In contrast, horizontal integration leads to pollutants assimilated in biomass produced in the system being removed during harvest, avoiding problems associated with the development of HEAP traps (Gunther, 1997) and the release of nutrients when the macrophytes senesce. Harvesting reeds for use as a 'green fuel' represents an important innovation in the management of constructed wetlands, in which nutrients sequestered from the unidirectional flow of matter in the hydrological cycle are permanently removed from the ecosystem and a renewable energy source is created.

Despite such potential there may be practical constraints. During harvest the wetland will not be available for wastewater treatment, although a possible solution would be to harvest the reedbed between culture cycles, when the smolt unit is empty. However, at this time, commonly between March and April, the reeds will have started to grow, a less optimal harvest time than the dormant winter period. An alternative strategy would be to construct the wetland in two separate basins, permitting one basin to remain functional whilst the other was harvested. Although engineering a wetland with multiple basins may increase the area of land required and add to capital costs, increased opportunities to adopt cut and fill could reduce earthmoving costs. This also presents opportunities for diversifying the macrophyte species cultivated. The proposition to plant only with the common reed was used to simplify management, but, harvesting and processing, risks from disease, infestations and grazing by waterfowl must be considered. Planting species that are dormant during different periods could spread the workload associated with harvesting extensive wetland areas, an important consideration if for instance employees from the smolt unit were to meet this labour demand.

In the second scenario, it is estimated that the reed biomass produced in the constructed wetland could assimilate considerably more N (769%) and P (957%) than is discharged from the smolt unit. This indicates that the capacity of the reedbed to retain nutrients is not fully exploited. The extra capacity of constructed wetlands for nutrient retention could potentially be exploited (and costs shared) by assimilating nutrients discharged from septic tanks, small sewage treatment works or runoff from agricultural land. However, such an integrated approach would only be possible where the hydraulic loading rate for the reedbed could be maintained, e.g. by reducing the volume of water used in the culture facility during periods when the oxygen requirement and production of waste is low. This contrasts to the situation at many commercial culture facilities where a constant volume of water is abstracted throughout the year. The additional capacity of the wetland to retain nutrients may also reduce the nutrient concentrations in discharges from the smolt unit to below those in its supply.

Using constructed wetlands to control nutrient flows at the local and regional scale may make a significant contribution to catchment-scale management initiatives. Furthermore, where the cost of nutrient discharge and value of nutrient retention is realised in the form of tradable permits, the operators of the constructed wetland may be in a position to benefit financially from the sale of this extra capacity. Although no such arrangements currently exist in Scotland, the ADEPT model could be further developed to facilitate an assessment of these likely financial impacts. Other possible areas of investigation would be the potential impact of introducing charges for water use or a pollution tax on nutrient discharges.

Consistent performance throughout the year is a basic requirement for treatment in meeting discharge standards. The capacity of constructed wetlands to maintain removal rates during winter when the plants are dormant is primarily due to non-biological processes such as sedimentation, filtration and soil adsorption. Describing the treatment effect of reedbeds receiving municipal and industrial wastewater in Beijing, China, Li et al. (1995) noted that performance did not deteriorate significantly during the winter months when the air temperature regularly dropped below zero. Furthermore, Schwartz and Boyd (1995) reported good pollutant removal rates during periods when vegetation in the reedbeds was dormant. Non-biological processes e.g. sedimentation and adsorption, are particularly important in temperate environments during winter when temperatures retard microbial activity and assimilation of nutrients by vegetation may cease.

Retaining water abstracted from a lotic environment in a trout fishery for several days may cause a change in the species assemblage present. Phytoplankton blooms and macrophyte communities proliferate in lentic environments receiving nutrient inputs and sustain complex food webs that may include species of zooplankton, herbivorous invertebrates and fish. The discharge of this water with its entrained lentic species assemblage could potentially disrupt the biotic community in a receiving lotic water body. In the present scenario water discharged from the trout fishery passes through a reedbed before being discharged from the site, preventing the majority of biota from the fishery passing to the receiving environment.

Detaining lotic waters in lakes and lochs is a natural process that can confer benefits on downstream environments. Water bodies retain significant amounts of sediment (White, Labadz and Butcher, 1996) and nutrients (Berge, Fjeld, Hindar and Kaste, 1997) that might otherwise degrade downstream ecosystems. Wetlands also facilitate the assimilation of nutrients into complex organic material, which if discharged, may cause receiving environments to shift back to being driven by externally derived detritus, as opposed to *in situ* primary production i.e. eutrophication. Consequently, fisheries and constructed wetlands may be viewed as ecosystems suitable for internalising the potentially negative environmental impact associated with discharging wastewater from smolt farms.

In addition to retaining nutrients, constructed wetlands, particularly reedbeds, have been cited as capable of reducing the level of pathogenic bacteria and viruses e.g. *Salmonella* sp. (Gersberg, Gearheart and Ives, 1989), chemicals e.g. refactory organics (Watson et al., 1989) and trace elements, including heavy metals (Goodrich-Mahoney, 1996) in wastewater. This is significant as the consent to discharge granted to aquaculture facilities "does not authorise the presence in the effluent discharged to the watercourse of any substance or other matter not mentioned in the consent or the conditions" (SEPA, 1996).

Horizontally integrated aquaculture has been proposed as a strategy capable of making a positive contribution to environmental protection, maximising production with respect to inputs and potentially reducing financial risks (Enander and Hasselstrom, 1994). Furthermore, Schwartz and Boyd (1995) considered constructed wetlands as inexpensive to build and operate compared with conventional treatment strategies. However, in these examples developing a constructed wetland for wastewater treatment results in a significant capital cost (£681,940) and the small profit generated (£2,700 y^{-1}) means that the IRR over ten, twenty and fifty years is negative.

In the baseline scenarios, depreciation on land and earthwork structures for extensive aquaculture were assigned a salvage value of zero, as alternative activities at the site may be limited, and redevelopment may represent a substantial cost. However, facilities that receive adequate maintenance and continue to be used for similar purposes, may be expected to retain a proportion of their initial value. Therefore, at the end of the financial assessment period used, the wetland could potentially be sold for an amount equivalent to the initial capital cost of the land, and possibly earthworks. However, this would depend on the proposed use of the site by the prospective buyer. The sensitivity analysis demonstrated that if the land and earthworks do retain their value, that the combined fishery and reedbed becomes a more attractive investment. However, the effect on the returns generated by the reedbed demonstrated that even in a favourable setting this strategy, with its high initial capital cost, remains financially unattractive.

In the baseline scenario, employing a trout fishery and reedbed for wastewater treatment generated a modest IRR (12.1%) over ten years. In most analyses, an IRR from 10-20% is considered desirable, although where a social benefit can be identified, Bojö

(1991) suggested that a lower rate of ~5% may be acceptable. However, managers of commercial aquaculture operations are unlikely to factor in such social benefits to any financial assessment. Furthermore, a relatively small IRR requires a greater weight to be attached to long-term effects, increasing the significance of financial risks. It could be argued that where there is an appropriate level of investment, structural integrity and functionality of ecologically based wastewater treatment systems may be maintained indefinitely, and a medium to long-term time horizon of 30-50 years may be considered. However, from the scenarios presented here it is apparent that returns generated over twenty and fifty years are not markedly different, whilst the risk attached to the longer time horizon increase significantly.

For a twenty-year period an integrated trout fishery and reedbed returns a reasonable IRR (18%), however, as the time horizon increases it becomes difficult to predict the future setting in which the system will function. For example, demand for fishing may decline or the effectiveness of the treatment strategy may decrease, particularly if stricter discharge regulations are introduced. The viability of the primary aquaculture activity may change. However, it should also be considered that the conditions for such a treatment approach might improve. As the fishery becomes more established greater numbers of anglers may be attracted, especially as productive fisheries receiving nutrient rich water from smolt units may be expected to enhance the growth and condition of stocked trout, or policies may develop that recognise the wider values of ecological based wastewater treatment, exempting the operator from pollution taxes or more stringent discharge standards. This suggests a need to develop a more wide ranging sensitivity or cost benefit analysis to consider the likely impact of such developments; the ADEPT model represents an important starting point. Furthermore, considering potential improvements in wastewater treatment, the adoption of constructed wetlands could potentially open up new sites demanding more stringent discharge standards or permit the expansion of existing farms. These are both attractive propositions where site availability constrains expansion in the aquaculture sector.

A criticism of commercial aquaculture is that increased efficiency and mechanisation leads to a decrease in employment; potentially causing resentment in local communities, and in some cases, actually reducing their viability. Employment opportunities associated with developing recreational fisheries in rural locations where smolt farms frequently occur could potentially improve the perceptions held by stakeholders concerning commercial aquaculture.

It is significant that the integrated trout fishery and reedbed creates a position for a manager, who in addition to duties connected directly with the trout fishery, would be responsible for general site maintenance, harvesting the reedbed and dealing with administration relating to wastewater management. In contrast, it is assumed that work associated with using a drumfilter or constructed wetland to treat wastewater will be undertaken by contract labours or staff employed to operate the primary culture facility. Employees at the smolt farm considered in this case study are frequently engaged in activities related to wastewater management. The operation of the drumfilter is monitored on a daily basis and maintenance e.g. greasing of gears and pressure washing screens, is carried out on a monthly basis; furthermore, accumulated sludge requires disposal once or twice a year and screens must be replaced at 2-3 year intervals.

An important constraint to using constructed wetlands could be the extensive land area required. Irrespective of local land values, suitable land may not be available adjacent to an existing aquaculture facility, whether due to inappropriate physical characteristics or prohibited by legislation. Where suitable land is identified but unavailable for sale, the proposed wetland may represent an interesting diversification for the landowner concerned, based on a revenue contract.

In the third scenario, the model predicted that land occupied by the integrated wetland generates a considerable annual income (£10,900 ha^{-1}) as compared with the return from agriculture. Should it prove possible to subsidise this income with grants for habitat creation, waste reduction or business development, then the economic viability of these systems would be improved. However, the risk associated with the initial capital costs must be considered. A landowner engaged in activities other than aquaculture e.g. farming, may also be in a position to engage their own labour and machinery to develop the site, reducing site development costs.

3.9. Summary

The ADEPT model was developed to facilitate an evaluation of the treatment effect, management demands and economic implications for conventional aquaculture wastewater treatment strategies and horizontally integrated systems. The model provides a framework in which to assess several of the parameters that managers of commercial aquaculture operations must evaluate when considering alternative waste management strategies. These parameters include treatment capacity and performance, financial requirements and economic implications. This study has demonstrated that the modelling approach adopted is appropriate to simulate the composition of wastewater discharged from a commercial smolt farm and the treatment effect of a drumfilter and horizontally integrated wetlands and recreational trout fishery. It is concluded that horizontally integrated systems may have a role to play in facilitating the reuse of nutrients in aquaculture wastewater, achieving higher treatment standards, generating additional income, creating employment opportunities and possibly assisting commercial aquaculture in becoming more sustainable.

Although horizontally integrated aquaculture may offer several advantages, two major constraints identified in the current scenarios are the land required and the initial capital expenditure. However, where land and capital are available for development for the proposed integrated trout fishery and reedbed system, the positive but relatively small economic returns could be subsidised by other potential benefits.

The conceptual model (Figure 3.1) elucidated a range of factors that must be evaluated when considering alternative wastewater management strategies. However, despite the apparent detail of the ADEPT model, it is important to reflect on a sentiment expressed by Thompson (1993a) that "the major follies of our time are rarely due to miscalculation of the detail, but more often the lack of appreciation of the totality of the effect". In addition to the treatment efficiency and financial demands, the decision whether to adopt horizontal integration depends on a number of additional factors, as demonstrated through the use of a stakeholder Delphi, the results of which are presented in Chapter 6. The next two chapters describe the use of the ADEPT model to explore alternative approaches to horizontally integrated aquaculture; resource recovery from wastewater, trends observed and possible constraints and opportunities are described. Chapter 4 assesses the potential of using constructed mangrove wetlands to treat wastewater from commercial shrimp farms.

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Chapter Four

Constructed mangrove wetlands for intensively managed shrimp pond wastewater

4.1. Introduction

Domestic and industrial water pollution, including that from aquaculture, has made it increasingly difficult for shrimp farms to appropriate supplies of good quality water (Beveridge et al., 1997). Wastewater treatment processes facilitated by mangroves have the potential to improve the quality of run-off flowing into coastal lagoons, estuaries and near-shore areas. Mangroves have been used to treat sewage and wastewater discharged from societal systems, with the majority demonstrating a high capacity for retaining nutrients and degrading organic matter (Clough, Boto and Attiwill, 1983; Dwivedi and Padmakumar, 1983; Tam and Wong, 1995; Wong, Lan, Chen, Li, Chen, Liu and Tam, 1995; Machiwa, 1998). Improving water quality in coastal areas would increase the availability of good quality water for shrimp culture and contribute to the maintenance of the trophic status of receiving habitats, ensuring that ecosystem support functions provided by the coastal environment persist and the utility of the resource is preserved.

The potential to exploit mangroves to condition water for use in shrimp aquaculture and to treat wastewater from shrimp farms has been considered by a number of authors (e.g. Robertson and Phillips, 1995; Rajendran and Kathiresan, 1996; Frederiksen, Sorensen, Finster and Macintosh, 1998; Wolanski, Spagnol, Thomas, Moore, Alongi, Trott and Davidson, 2000). Robertson and Phillips (1995) employed a mass-balance approach to estimate the mangrove area needed to assimilate nutrients present in wastewater and sediments from a semi-intensively managed shrimp pond in Tra Vinh Province, Vietnam and an intensively managed pond in Chantaburi Province, Thailand. They estimated that excess nitrogen and phosphorus from 1 ha of semi-intensive shrimp ponds could be assimilated by 2.5 ha and 3.4 ha of mangrove, respectively; assimilation of excess nitrogen and phosphorus from 1 ha of intensive shrimp ponds could occur with 7.2 ha and 21.7 ha of mangrove, respectively. However, the treatment effect of mangroves with respect to the concentration of waste fractions present in water discharged from shrimp farms, and the financial and economic implications of developing and managing constructed mangrove wetlands have not been investigated.

Scenarios developed in the second case study using the ADEPT model examine the potential of using constructed mangrove wetlands to treat wastewater from intensively managed shrimp ponds stocked at low and high densities. Employing the ADEPT model, the composition of wastewater discharged from intensively managed shrimp ponds stocked at low and high densities was simulated. The mangrove area required for nutrient assimilation was used to dimension the wetland and the expected treatment effect was simulated employing the k-C* models described in Chapter 3.

The rationale for employing a mangrove wetland for wastewater treatment was that such systems, in addition to facilitating a range of treatment processes, have the potential to generate a variety of both marketed and non-marketed goods. Marketed goods could include timber, thatch, charcoal, fish and shellfish; the provision of habitat and nursery areas, shoreline protection and the regulation of the local hydrology may be considered as non-marketed goods. In this case study it was assumed that wood harvested from the mangrove would represent a marketable good for sale by the shrimp farmer. Costs associated with developing the constructed wetland were assessed, income generated from selling wood produced in the horizontally integrated mangrove plantation was estimated and potential benefits associated with integrating further aquaculture activities are discussed.

4.2. Method

Scenarios developed here employ background information on environmental conditions and management practices for intensive shrimp culture, recorded at commercial farms in Thailand; this data is summarised in Table 4.1. Mean temperatures reported by Cowan, Lorenzen and Funge-Smith (1999) of 28°C during the wet season and 30°C during the dry season, were assumed. These authors also reported that intensively managed shrimp ponds stocked at low and high densities have similar mean depths of 1.6 m and that 0.4, 4, 6 and 8% of the water in these ponds was exchanged daily during the first, second, third and fourth months of culture, respectively. Information relating to the management of shrimp ponds stocked at low and high densities, methods employed to determine the composition of wastewater and predict the treatment effect of a constructed mangrove wetland and approaches used to assess financial and economic implications associated with development are presented in the following sections.

4.2.1. Pond management

Operating parameters reported by Briggs and Funge-Smith (1994) and Cowan et al. (1999) for intensively managed shrimp ponds in Thailand, stocked during the first year of production at low and high densities were employed to develop two scenarios (Table 4.1). Briggs and Funge-Smith (1994) reported that grow-out periods in intensively managed ponds average 124 days, culture periods are assumed to last 4 months, with fallow periods between each set at two months. Post larval ($PL_{15\text{-}30}$) shrimp (*Penaeus monodon*) are introduced at mean densities of 52.5 and 94.8 m^{-2} to intensively managed ponds to achieve low and high stocking densities, respectively.

Table 4.1. Operating parameters for intensively managed ponds in Thailand stocked at low and high densities

Operating parameter	Low	High
Pond depth (m)	1.6	1.6
Stocking density (no. m^{-2})	52.5	94.8
Survival (%)	40.6	44.8
Mean shrimp harvest weight (g)	21.7	21.5
FCR	2.13	1.93
Feed input (% body weight d^{-1})	5.2	10.6
Feed input (kg ha^{-1} $cycle^{-1}$)	8,800	17,280

4.2.2. Dimensioning the mangrove wetland

A simple stock model was developed to account for the stocking density (m^{-2}), anticipated FCR and feed input (% body weight d^{-1}); this, together with assumptions concerning the proximate composition of feed and shrimp, was used to estimate the proportion of nutrients and solids introduced as feed that are not assimilated in shrimp biomass. Feed pellets introduced to culture ponds stocked at high and low densities equate to 34.6 and 17.6 t ha^{-1} y^{-1}, respectively. Robertson and Phillips (1995) report that shrimp feed contains 7.1% of N and 1.6% of P, and that shrimp contain 2.94% of N and 0.19% of P. The solids content of commercial shrimp feed is assumed at 90.3% and that of shrimp at 24.6% (Funge-Smith and Briggs, 1998).

The estimated mass of N, P and solids introduced as feed on a monthly basis, but not assimilated in shrimp biomass, was divided by the volume of water discharged from the culture pond to simulate the concentration of these waste fractions in wastewater entering the constructed wetland. Representative concentrations for TAN, BOD and DO in wastewater from tropical shrimp ponds reported by Briggs and Funge-Smith (1994) were used in both scenarios.

A mass balance approach was employed to dimension the mangrove area required to assimilate excess N and P; the difference in N and P introduced as feed and assimilated in shrimp biomass was used to estimate the mass of nutrients that the mangrove would be required to assimilate. Robertson and Phillips (1995) reported that the production of litter, wood and roots in a tropical *Rhizophora*-dominated mangrove forest averages 8, 20 and 28 t ha^{-1} y^{-1}, respectively; these values are used in the scenarios presented here. However, the species assemblage in a mangrove forest is likely to depend on local environmental and hydrological conditions and prior to using the ADEPT model to assess the potential of horizontal integration in specific settings, the likely productivity of the prevailing mangrove type should be evaluated. The value for wood accumulation in a *Rhizophora*-dominated mangrove employed in this case study, represents a mean value derived from a comprehensive review of biomass accumulation rates in such mangrove stands by Clough (1992; cited in Robertson and Phillips, 1995). In this review estimated accumulation rates for wood in natural and managed mangrove stands ranged from 6 to 45 t ha^{-1} y^{-1}, however, Robertson and Phillips (1995) considered 20 t ha^{-1} y^{-1} a conservative estimate for wood accumulation in managed mangroves fertilised with aquaculture wastewater. Furthermore, fertilisation of *Rhizophora* trees has been shown to increase tissue nutrient concentrations (Clough et al., 1983). Ideally, therefore, further to considering the productivity of the prevailing mangrove type, the response of such a mangrove stand to varying cropping patterns and fertilisation rates should be assessed when considering the potential of constructed mangrove wetlands for aquaculture wastewater treatment.

However, in the absence of data for mangrove stands managed for aquaculture wastewater treatment, Robertson and Phillips (1995) noted that the N content of litter, wood and roots was 1, 0.12 and 0.41 per cent, respectively, and it was estimated that 219 kg ha^{-1} y^{-1} of N was required to sustain mangrove production. Employing a similar approach, and assuming the P content of litter, wood and roots is 0.1, 0.03 and 0.02 per cent, respectively, it was estimated that 20 kg ha^{-1} y^{-1} of P is required to sustain mangrove production. As the mangrove areas required to assimilate N and P differed, the larger area was used to dimension the constructed wetland.

4.2.3. Treatment performance

The treatment effect of constructed mangrove wetlands was based on k-C* models for surface-flow constructed wetlands given by Kadlec and Knight (1996) and described in Chapter 3. Concentrations expected in water abstracted for culture purposes were taken from Briggs and Funge-Smith (1994). These authors reported that in water abstracted for use in shrimp ponds in Thailand mean concentrations of SS were 108 mg l^{-1}, TAN 0.08 mg l^{-1}, TN 0.69 mg l^{-1}, TP 0.12 mg l^{-1} and BOD 1.9 mg l^{-1}. As in Chapter 3, background DO levels were estimated based on 100% saturation levels. Permanent nutrient removal by the constructed wetland was based on the harvest of mangrove wood; assuming wood accumulates 20 t ha^{-1} y^{-1}, N and P removal rates equate to 24 and 6 kg ha^{-1} y^{-1}, respectively.

4.2.4. Financial and economic implications

Costs associated with developing the site were based on values presented by Mara et al. (1993) and Mukherjee (personal communication, 1998) for constructing large-scale ecologically based wastewater treatment facilities in Asia. The cost of land for construction was assumed at £2,060 ha^{-1} and the cost of developing this area based on the cost of digging 1 m deep ponds and transferring earth manually to trucks, estimated at £3,960 ha^{-1}. However, by employing machinery on large-scale developments and optimising arrangements for cut-and-fill it may be possible to reduce site development costs. An additional 5% land area was included for access and infrastructure. It was assumed that one control structure was required to regulate the flow of water entering the constructed wetland, and a second to regulate the discharge of treated water to the receiving environment. The cost of the control structures was estimated at £290 per installation and it was anticipated that miscellaneous equipment costing £5 per employee would be required.

Operating costs included wages to pay managers and workers and money to purchase materials and equipment to maintain the wetland and associated infrastructure. Based on prevailing rates in India, it was assumed that a manager is required when the constructed wetland exceeds 10 ha, and that the salary would be £825 y^{-1}; workers are employed at a rate of one every 5 ha, each receiving £410 y^{-1} (Mukherjee, personal communication, 1998). The cost of materials and equipment to maintain infrastructure was set at 1% of initial capital cost. Income would be generated through the sale of mangrove wood, and based on information given by [1]Primavera (1995), a market value of £0.05 kg^{-1} was assumed. Accounting for the costs and potential returns described above, key financial indicators were calculated as in Chapter 3. Furthermore, the same discounted cash flow approach was used to calculate the NPV over ten years at discount rates of 5, 10, 15 and 20 per cent and the IRR over ten, twenty and fifty years. Salvage values and replacement costs were also treated as in Chapter 3.

[1]Primavera (1995) reported values for goods from mangroves of \$30-2,000; assuming an exchange rate of £1 to \$1.5 and that 20 t of marketable wood is produced annually, the value of this wood is estimated to range between £0.001-0.067 kg^{-1}. Although the upper quartile of the range, £0.05 kg^{-1}, is used here, this author noted that the sustained harvest of wood products in Malaysia could realise a return of \$11,561 ha^{-1} y^{-1}.

4.3. Results

Physical characteristics of the constructed mangrove wetlands required to treat wastewater from intensively managed shrimp ponds stocked at low and high densities are summarised in Table 4.2. Other key indicators regarding the potential of employing mangrove wetlands to treat shrimp culture wastewater are presented below.

4.3.1. Biomass production and waste outputs

The stock model predicted that intensively managed ponds stocked at high densities produce 18.2 t ha^{-1} y^{-1} of shrimp, with 9.1 t produced from each culture cycle. For ponds stocked at low densities, 9.3 t ha^{-1} y^{-1} of shrimp biomass is produced, each culture cycle producing ~4.6 t. The model estimated that the weight of solids contained in feed entering ponds stocked at high densities was 32.9 t ha^{-1} y^{-1}. Furthermore, based on the assumed FCR and proximate analysis values for shrimp and feed, it was estimated that 22.7 t ha^{-1} y^{-1} of solids would not be assimilated and would be released to the receiving environment. Based on the assumptions in Section 4.2.2, it was estimated that on an annual basis, 1,427 kg of N and 501 kg of P would be released from 1 ha of culture ponds stocked at a high density. It was estimated that of 15.9 t ha^{-1} y^{-1} of solids delivered in feed to ponds stocked at low densities, 11.6 t ha^{-1} y^{-1} would not be assimilated; the mass of N and P introduced as feed but not assimilated was estimated at 727 kg and 255 kg, respectively.

4.3.2. Dimensioning the mangrove wetland

Assuming mangroves assimilate 20 kg P ha^{-1} y^{-1}, it was estimated that ~12.8 and 25 ha of mangrove would be required to assimilate the 255 and 501 kg of excess P released from 1 ha of ponds stocked at low and high densities, respectively. An additional land area of 5%

required for embankments, access roads and supporting infrastructure increases the total area needed to 13.4 and 26.3 ha for ponds stocked at low and high densities, respectively.

Based on mean growth rates, the production of litter, wood and roots in 12.8 ha of mangrove equates to 102, 256 and 358 t, respectively. The potential for N assimilation in this area was estimated at 2,803 kg y^{-1}, however, the estimated release of N from ponds stocked at low densities was 727 kg y^{-1}, indicating that only 26% of the N assimilation capacity of the mangrove would be exploited. The mangrove area (25 ha) required to assimilate excess P from 1 ha of ponds stocked at high densities was estimated to produce 200 t of litter, 501 t of wood and 700 t of roots annually. Excess N released from these ponds was estimated at 1,427 kg y^{-1}, however, the mass of N required to sustain production in this mangrove area equates to 5,475 kg y^{-1}, again indicating that only 26% of the N assimilation capacity of the mangrove stand would be utilised.

Table 4.2. Physical characteristics of mangrove wetlands treating wastewater from 1 ha of shrimp ponds stocked at low and high densities.

Characteristic			Low	High
Shrimp production (kg ha^{-1} y^{-1})			9.3	18.2
N not assimilated			727	1,427
P not assimilated			255	501
Mangrove area required for P assimilation (ha)			12.8	25
Total area required (ha)			13.4	26.3
Production (t y^{-1}):	litter		102	200
	wood		256	500
	roots		358	700
Permanent removal through harvesting wood (kg y^{-1}):		N	306	601
		P	77	150

Permanent nutrient removal from the horizontally integrated mangrove system is predicted to occur primarily through the harvest of accumulated wood. Harvesting 256 t of wood produced annually in the mangrove receiving wastewater from low density ponds would remove 306 kg of N and 77 kg of P, equivalent to 42 and 30%, respectively, of the excess nutrients entering the culture system. Wood harvested annually in the 27 ha wetland treating wastewater from high density ponds would contain 601 kg of N and 150 kg of P, equivalent to 42 and 30%, respectively of excess nutrients discharge from the ponds.

4.3.3. Wastewater composition and volume

Means and ranges predicted by the ADEPT model for waste concentration changes in water used in shrimp ponds stocked at low and high densities are presented in Table 4.3. For the low stocking density scenario, profiles for the predicted discharges of SS, BOD and DO are presented in Figure 4.1, those for TAN, TN and TP are given in Figure 4.2. Profiles for SS, BOD and DO for the high stocking density scenario are presented in Figure 4.3, those for TAN, TN and TP are given in Figure 4.4.

Table 4.3. Predicted mean changes in waste concentrations (mg l^{-1}) in treated and abstracted water for low and high density shrimp culture (where appropriate, ranges given in parenthesis).

Waste fraction	Low density		High density	
	Pond discharge	Treated water	Pond discharge	Treated water
SS	109 (46~265)	-38 (-59~14)	130 (52~195)	-32 (-7~-57)
TAN	0.31	-0.08	0.84	-0.08
TN	6.8 (2.9~16.7)	0.8	8.2 (3.2~12.3)	0.8
TP	2.4 (1~5.9)	-0.1	2.9 (1.1~4.3)	-0.1
BOD	5.3	2	6.5	2.1
DO	0.9	-4.2 (-4.1~-4.3)	1.7	-4.3 (-4.2~-4.4)

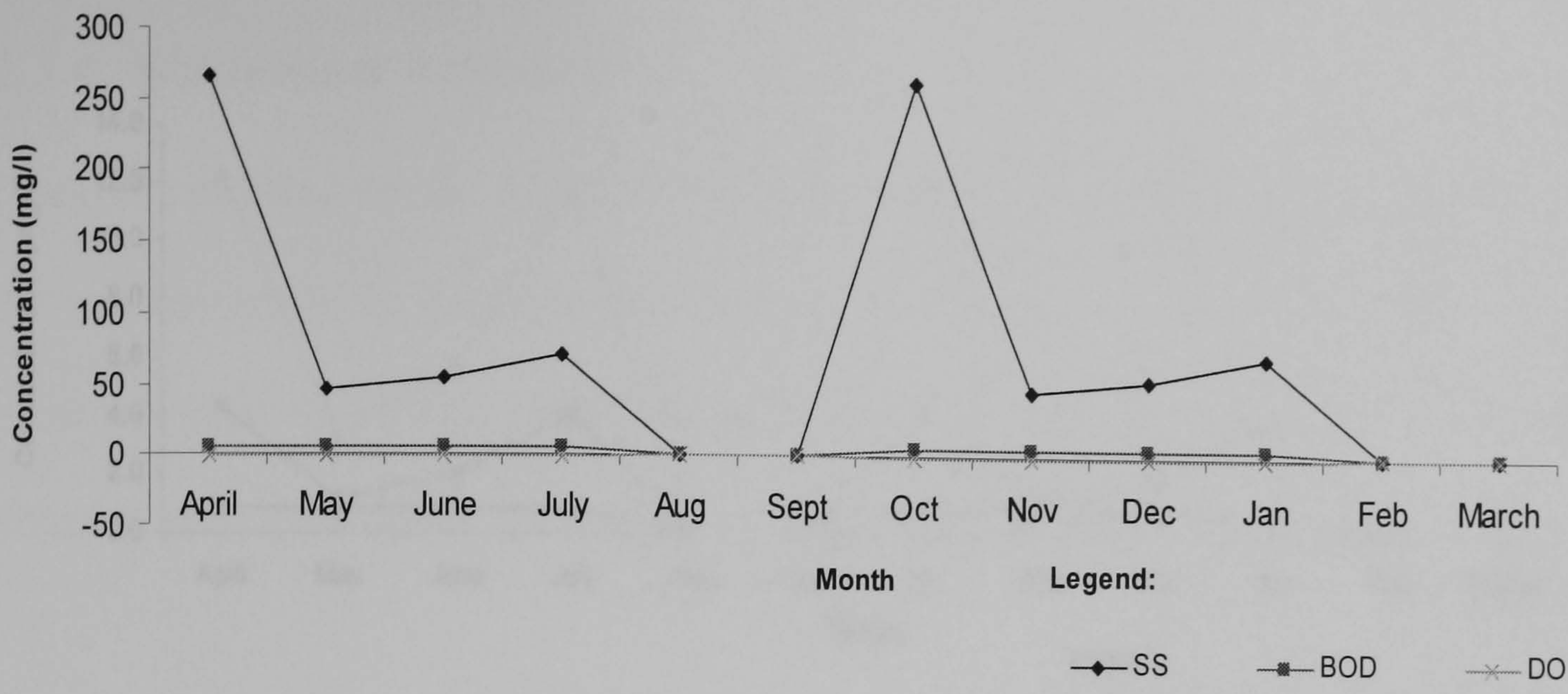

Figure 4.1. Mean predicted concentration (mg/l) change for SS, BOD and DO in wastewater from shrimp ponds stocked at low densities

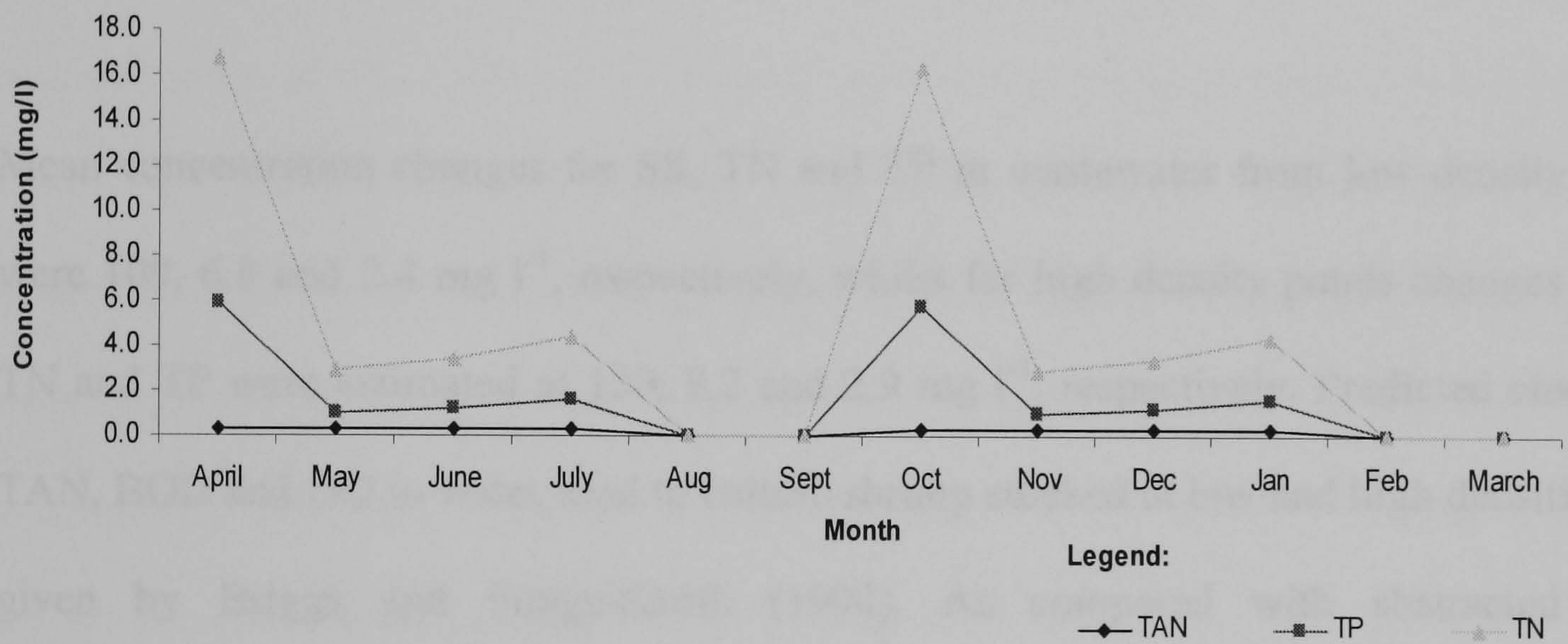

Figure 4.2. Mean predicted concentration (mg/l) change for TAN, TN and TP in wastewater from shrimp ponds stocked at low densities

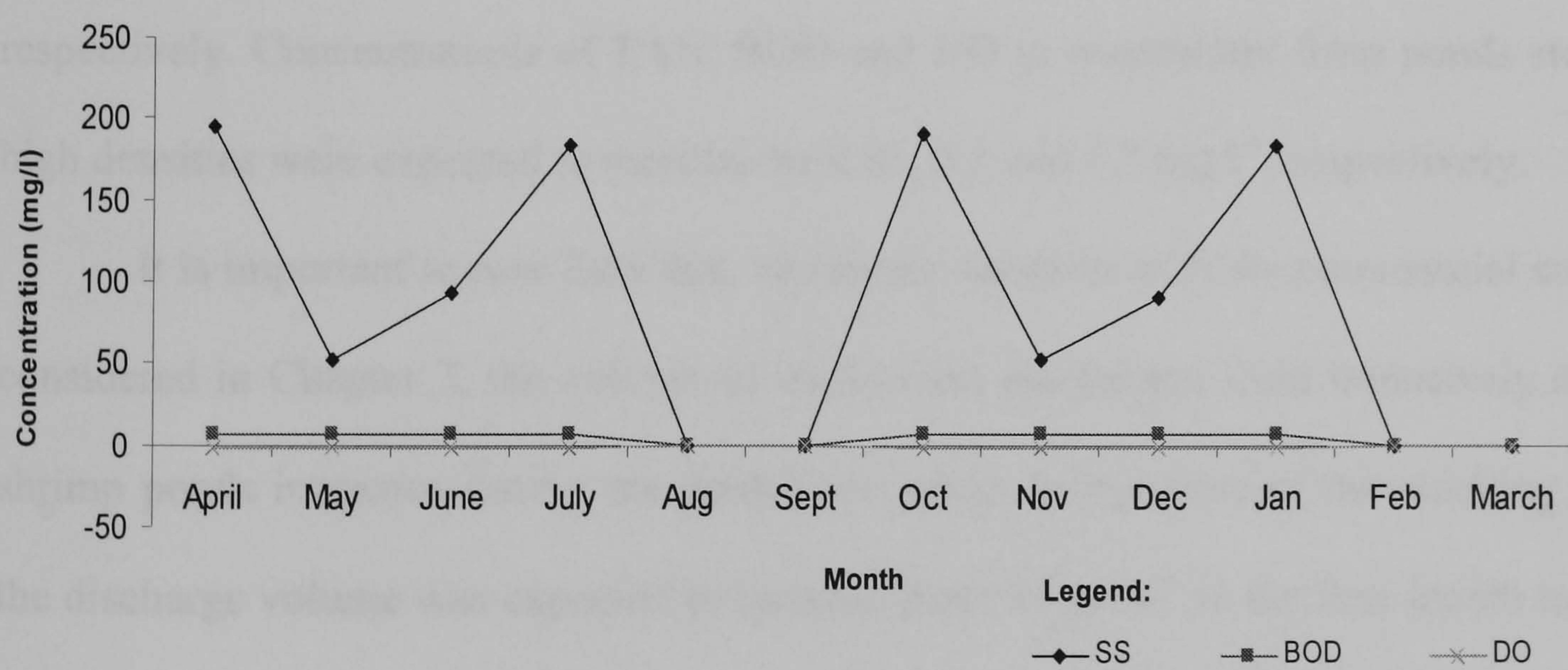

Figure 4.3. Mean predicted concentration (mg/l) change for SS, BOD and DO in wastewater from shrimp ponds stocked at high densities.

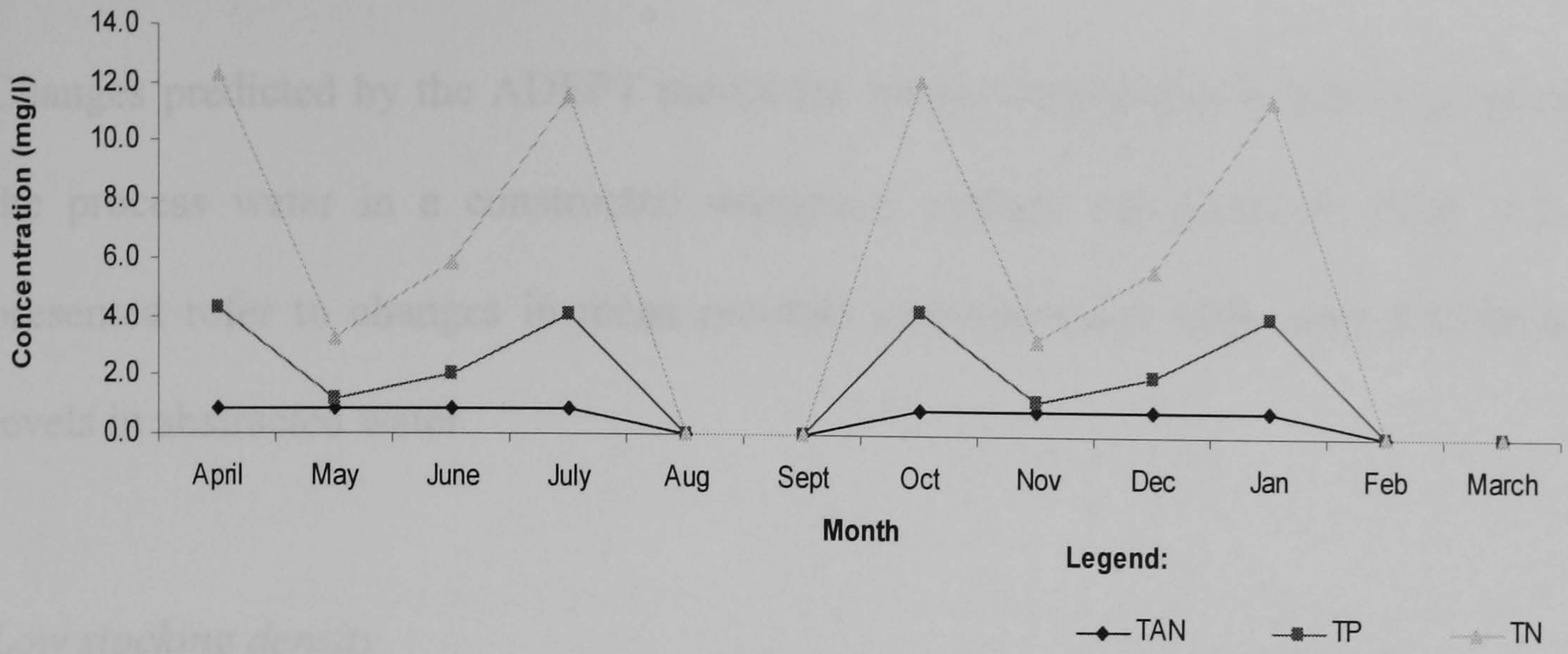

Figure 4.4. Mean predicted concentration (mg/l) change for TAN, TN and TP in wastewater from shrimp ponds stocked at high densities.

Mean concentration changes for SS, TN and TP in wastewater from low density culture were 109, 6.8 and 2.4 mg l^{-1}, respectively, whilst for high density ponds changes for SS, TN and TP were estimated at 130, 8.2 and 2.9 mg l^{-1}, respectively. Predicted changes in TAN, BOD and DO in water used to culture shrimp stocked at low and high densities were given by Briggs and Funge-Smith (1994). As compared with abstracted water, concentrations of TAN, BOD and DO in wastewater from intensively managed shrimp ponds stocked at low densities were expected to increase by 0.31, 5.3 and 0.9 mg l^{-1}, respectively. Concentrations of TAN, BOD and DO in wastewater from ponds stocked at high densities were expected to increase by 0.81, 6.5 and 1.7 mg l^{-1}, respectively.

It is important to note here that, unlike the situation with the commercial smolt unit considered in Chapter 3, the volume of wastewater discharged from intensively managed shrimp ponds increases during the production cycle. Irrespective of the stocking density, the discharge volume was expected to increase from 64 m^3 d^{-1} in the first month to 640 m^3 d^{-1} in the second, 960 m^3 d^{-1} in the third, reaching a maximum of 1,280 m^3 d^{-1} in the fourth. It was assumed that during the fallow periods the ponds are left to dry.

4.3.4. Wastewater treatment

Changes predicted by the ADEPT model for waste concentrations following treatment of the process water in a constructed mangrove wetland are given in Table 4.2; values presented refer to changes in mean monthly concentrations with respect to background levels in abstracted water.

Low stocking density

Profiles for the predicted discharges of SS, BOD and DO following treatment are presented in Figure 4.5, those for TAN, TN and TP are given in Figure 4.6. Predicted mean concentrations of SS, TAN and TP following treatment are on average below background levels expected in abstracted water by 38, 0.08 and 0.1 mg l^{-1}, respectively representing reductions of 135, 126 and 104 per cent. However, mean TN and BOD concentrations were predicted to remain above concentrations in abstracted water by 0.8 and 2 mg l^{-1}, respectively, indicating that only 88% and 63% of the TN and BOD released from the culture ponds would be removed.

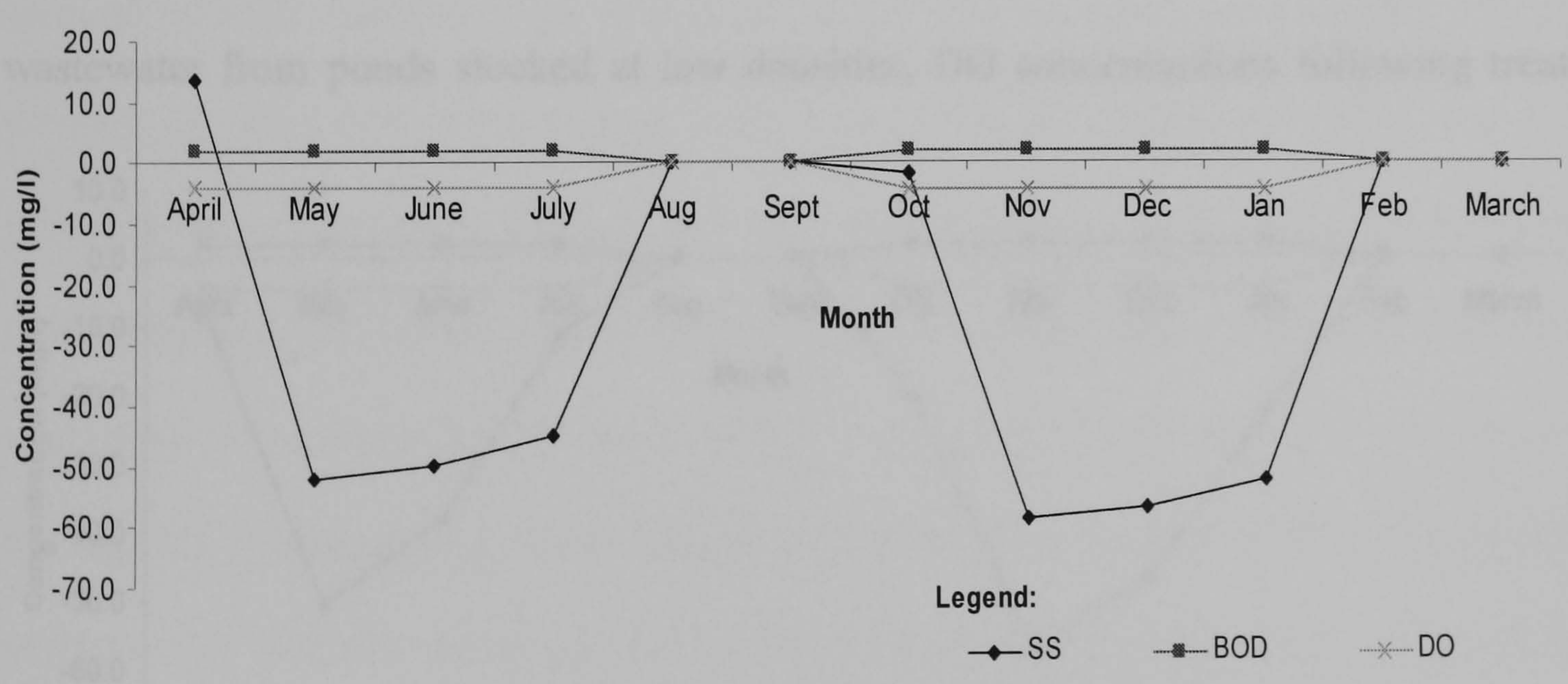

Figure 4.5. Mean predicted SS, BOD and DO concentrations (mg/l) in wastewater from low density ponds treated using a mangrove wetland (minus background levels)

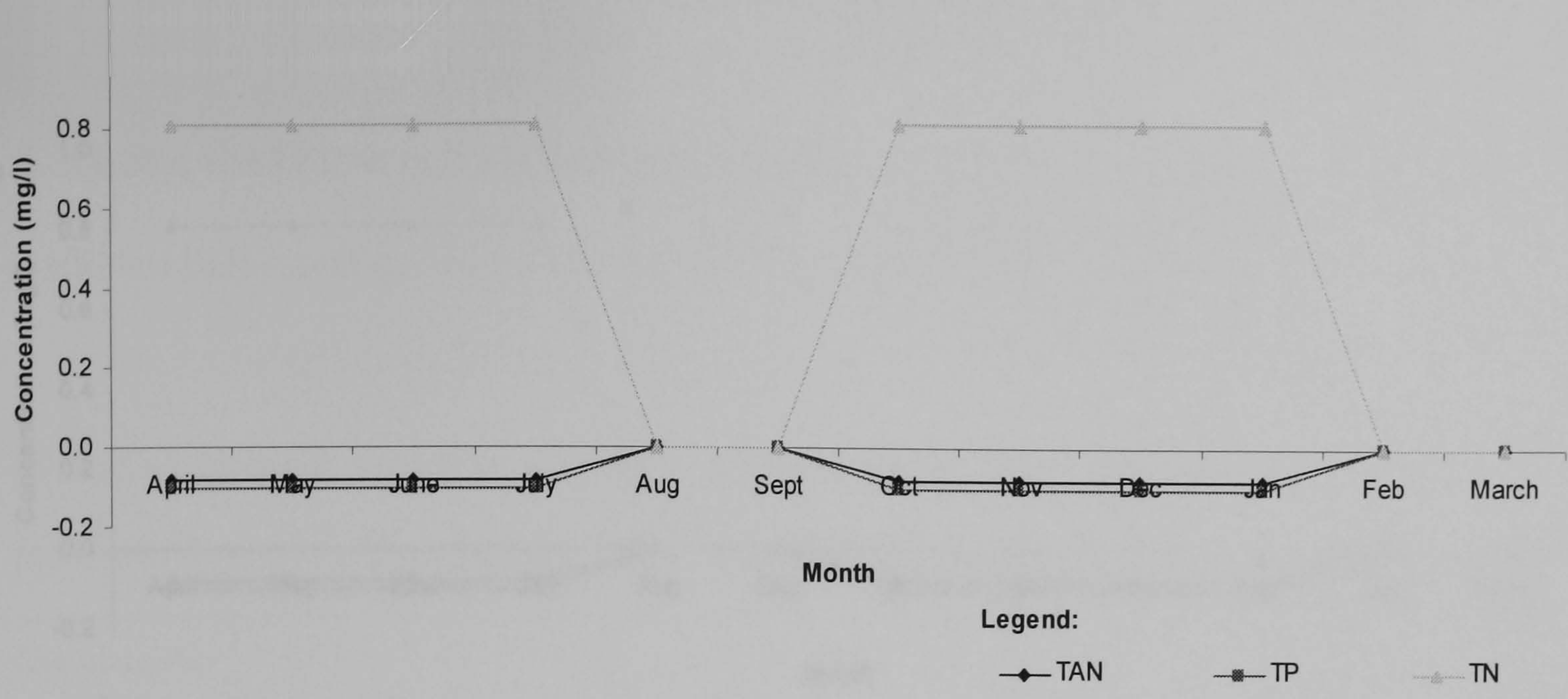

Figure 4.6. Mean predicted TAN, TN and TP concentrations (mg/l) in wastewater from low density ponds treated using a mangrove wetland (minus background levels)

High stocking density

Profiles for the predicted discharges of SS, BOD and DO following treatment are presented in Figure 4.7, whilst those for TAN, TN and TP are given in Figure 4.8. Treatment in the constructed wetland was expected to reduce mean SS, TAN and TP concentrations to below background levels in abstracted water by 32, 0.08 and 0.1 mg l^{-1}, respectively, representing reductions of 124, 109 and 104 per cent, as compared with outputs from the culture unit. Mean concentrations of TN and BOD were predicted to remain above those expected in abstracted water by 0.8 and 2.1 mg l^{-1}, respectively, indicating that treatment removes on average 90% of TN and 69% of BOD. As with wastewater from ponds stocked at low densities, DO concentrations following treatment

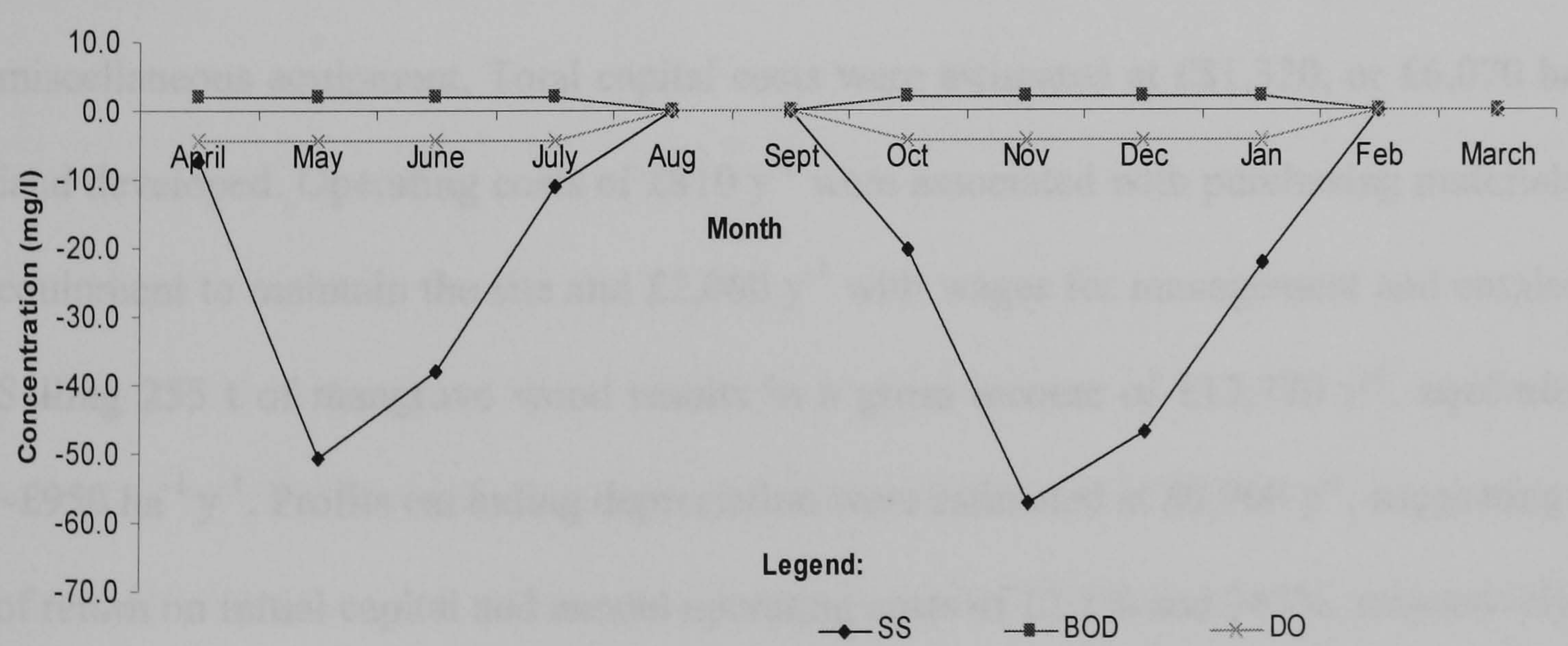

Figure 4.7. Mean predicted SS, BOD and DO concentrations (mg/l) in water from high density ponds treated using a mangrove wetland (minus background levels)

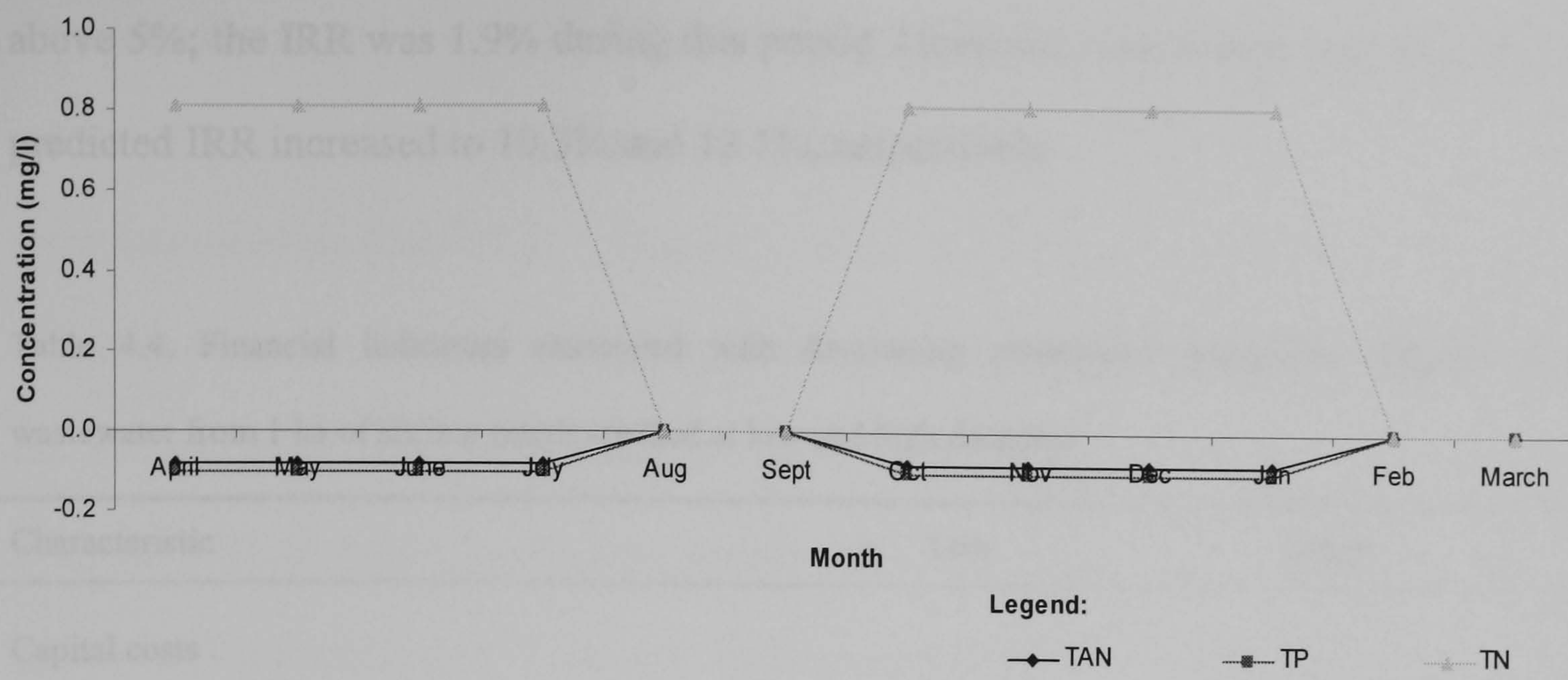

Figure 4.8. Mean predicted TAN, TN and TP concentrations (mg/l) in water from high density ponds treated using a mangrove wetland (minus background levels)

4.3.5. Financial implications

The financial demands, including expected capital and operating costs, for constructed mangrove wetlands treating wastewater from ponds stocked at high and low densities are summarised in Table 4.4.

Low stocking density

Major capital costs associated with developing an area of 13.4 ha for the mangrove wetland in this scenario were estimated at £27,600 to purchase land and £53,100 to develop the site. Additional capital costs were £580 for control structures and £15 for miscellaneous equipment. Total capital costs were estimated at £81,320, or £6,070 ha^{-1} of land developed. Operating costs of £810 y^{-1} were associated with purchasing materials and equipment to maintain the site and £2,060 y^{-1} with wages for management and employees. Selling 255 t of mangrove wood results in a gross income of £12,770 y^{-1}, equivalent to ~£950 ha^{-1} y^{-1}. Profits excluding depreciation were estimated at £9,900 y^{-1}, suggesting rates of return on initial capital and annual operating costs of 12.2% and 345%, respectively, and a payback period of 8.2 years. During the first ten years, cash flows generated by the horizontally integrated mangrove plantation result in negative NPV's at discount rates above 5%; the IRR was 1.9% during this period. However, over twenty and fifty years the predicted IRR increased to 10.3% and 12.1%, respectively.

Table 4.4. Financial indicators associated with developing constructed mangrove wetlands to treat wastewater from 1 ha of shrimp ponds stocked at low and high densities

Characteristic		Low	High
Capital costs			
Land		£27,600	£54,200
Site development		£53,100	£104,180
Miscellaneous		£595	£605
Total		£81,320	£158,980
Operating costs			
Maintenance		£810	£1,590
Labour		£2,060	£2,880
Total		£2,870	£4,470
Profit excluding depreciation		£9,900	£20,600
Pay-back period (y)		8.2	7.7
Return on capital costs		12.2%	13%
Return on operating costs		345%	461%
Ten year NPV at:	5%	-£10,400	-£12,000
	10%	-£22,100	-£36,700
	15%	-£29,600	-£52,800
	20%	-£34,500	-£63,300
IRR (%) over:	10 y	1.9	3.2
	20 y	10.3	11.2
	50 y	12.1	12.9

High stocking density

Major capital costs associated with developing 26.3 ha for the mangrove wetland were estimated at £54,200 to purchase land and £104,180 to develop the site. Other capital costs were £580 for control structures and £25 for miscellaneous equipment. Total capital costs were estimated at £158,980, or £6,045 ha^{-1} of area developed, slightly less than in the low stocking density scenario. Operating costs of £1,590 y^{-1} and £2,880 y^{-1} are associated with site maintenance and wages, respectively. Selling 501 t of mangrove wood resulted in an estimated income of £25,050 y^{-1}, equivalent to £952 ha^{-1} y^{-1}. Profits excluding depreciation were estimated at £20,600 y^{-1}; suggesting rates of return on initial capital and annual operating costs of 13% and 461%, respectively and a payback period of 7.7 years. During the first ten years, cash flows generated result in negative NPV's at discount rates above 5%; the IRR is 3.2% during this period. However, over twenty and fifty years the estimated IRR increased to 11.2% and 12.9%, respectively.

4.3.6. Sensitivity analysis

As in Chapter 3, the sensitivity analysis was achieved by modifying key input parameters in turn and assessing the impact on the ten year IRR. Outputs are summarised in Table 4.5, and it is apparent that changing costs produce a similar response in both scenarios. A 20% decrease in land or site development costs had a minimal effect on the IRR, which remains below ~7% in both scenarios.

Table 4.5. Sensitivity of ten year IRR (%) to changing input parameters at low and high stocking densities.

Parameter	Low	High
Baseline	1.9	3.2
Land cost (-20%)	3.5	4.8
Land cost (+20%)	0.4	1.7
Site development cost (-20%)	5.1	6.6
Site development cost (+20%)	-	0.4
Value of mangrove biomass (-20%)	-	-
Value of mangrove biomass (+20%)	7	8.1
Land retains its value over ten years	6.6	7.6
Land and earthworks retain value over ten years	12.1	12.9
Land cost assumed at £0 ha^{-1}	12.3	13.9
Stocking density increased by 50%	3.2	3.4
Stocking density decreased by 50%	0.8	1.8

Assuming the value of mangrove wood produced was 20% higher generates IRRs of 7% and 8.1% at low and high stocking densities, however, accounting for a 20% decrease results in negative IRRs in both cases. Assuming land retains its value over ten years results in higher IRRs of ~7% in both cases, however, assuming both land and earthworks retain their value results in an IRR of 12.1% in the low stocking density case and 12.9% at a high density. The greatest increase in IRR to 12.3% and 13.9% at low and high stocking densities, respectively, occurred when the land cost was assumed as zero. A 50% increase or decrease in shrimp stocking density results in only marginal changes as compared with the baseline IRR.

4.4. Discussion

4.4.1. Wastewater characteristics

Briggs and Funge-Smith (1994) report that mean concentrations of SS, TN and TP in water passing through recently constructed, intensively managed shrimp ponds stocked at high densities were 31, 2.1 and 0.17 mg l^{-1} higher, than abstracted water. However, mean concentration increases for SS (109 and 130 mg l^{-1}), TN (6.8 and 8.2 mg l^{-1}) and TP (2.4 and 2.9 mg l^{-1}), predicted using the ADEPT model, for ponds stocked at both low and high densities, are higher than those reported by these authors. Budgets for solids and nutrients proposed by Briggs and Funge-Smith (1994) account for a range of inputs including lime, fertiliser, juvenile shrimp and inlet water, and consider dynamic processes occurring in culture ponds i.e. erosion, denitrification and sediment accumulation. In contrast, the ADEPT model treats the shrimp pond as a black box, only accounting for major inputs (feed) and outputs (wastewater and shrimp biomass). Consequently, the model simulates a culture system that may be a closer approximation to a lined pond, relying on commercial feed to sustain production, as opposed to the conventional earthen pond culture system described by Briggs and Funge-Smith (1994). Nitrogen and phosphorus may be retained in sediments that accumulate in shrimp culture ponds and nitrogen may be lost from the system through denitrification and volatilisation (Funge-Smith and Briggs, 1998).

The ADEPT model could be modified to account for the dynamics of waste fractions in the culture ponds, however, solids and nutrients that accumulate in sediments still require disposal and may ultimately be released to the receiving environment where they will require degradation and assimilation. Accounting for processes such as denitrification and volatilisation, where compounds are lost from the system, may be appropriate, although the extent and magnitude of these processes is likely to vary depending on management regimes, environmental conditions and climate.

The volume of water discharged from the shrimp ponds remains at low levels (0.4-8% d^{-1}) during the culture cycle, however, the ADEPT model assumes that the total waste loading from the culture system becomes entrained in water discharged from the culture facility. This may represent a reasonable assumption in flow-through systems where exchange rates exceed 100% d^{-1}, however, in the scenarios presented here, it may have been useful to adopt a modified approach. This could have simulated the composition of wastewater from the shrimp ponds to account for initial dilution and gradual accumulation of waste products in the culture system. However, unless the dynamic physical, chemical and biological processes that occur in shrimp ponds were considered, this approach may not have resulted in predictions that were more accurate. Furthermore, accounting for processes of dilution and accumulation in the process water would not change the total loading of excess nutrients and other waste fractions generated by the culture facility that may ultimately be discharged to the receiving environment.

4.4.2. Nutrient retention

Design criteria employed in this study meant that the capacity of the constructed wetland was sufficient to assimilate the total loading of N and P released from the culture system. However, Li (1997) demonstrated that in established mangrove forests in Shenzhen, south China, the assimilation of N and P by mangrove trees is in equilibrium with the return of nutrients to the ecosystem as litter. This finding indicates that established mangrove forests may not retain a significant proportion of the nutrients assimilated during primary production. However, the capacity of forests to permanently sequester nutrients from the ecosystem may be enhanced through regular cropping of accumulated wood; a simple mass-balance approach was employed here to demonstrate the potential of this management strategy.

The ADEPT model estimated that the proportion of N (42%) and P (30%) from wastewater discharged from intensively managed shrimp ponds, stocked both at high and low densities, and removed from the mangrove ecosystem through harvesting accumulated wood was significantly less than that assimilated in the mangrove plantation. The need to permanently remove accumulated biomass, and concomitantly nutrients from mangrove forests, was noted by Robertson and Phillips (1995) as being a condition of using mangroves as nutrient filters for shrimp pond wastewater. This investigation demonstrated that the actual area of mangrove required to assimilate the nutrients discharged from shrimp aquaculture is greater than the mangrove area required to assimilate the nutrients *per se*. The wetland design could be modified so that theoretically all nutrients released from the shrimp ponds could be assimilated into wood that could be harvested and removed from the system. However, such a strategy would most likely result in mangrove production further from the wastewater inlet becoming both P and N limited. With the design approach presented here, only 26% of the N that would usually be assimilated in the area of mangrove required to assimilate excess P was available, suggesting that N availability would limit overall production, and consequently P assimilation. Another N source may therefore be required to ensure that production in the mangrove does not become limited by this nutrient. Further work would be required to assess the possibility for other nutrients to limit mangrove production.

Mangrove ecosystems are capable of storing nutrients; Robertson and Phillips (1995) suggested that the retention and recycling of matter in sediments underlying mangrove forests represents a sink for nutrients. The ADEPT model could be modified to account for nutrient retention in mangrove sediments; however, the capacity of such sediments to retain nutrients is likely to be site specific, varying with the hydraulic regime, nutrient loading rate and age of the mangrove forest. Furthermore, the accumulation of nutrients in mangrove sediments does not represent an appropriate strategy for reusing nutrients discharged in shrimp culture wastewater.

4.4.3. Environmental goods and services associated with mangroves

Although the retention of assimilated nutrients in established mangrove communities may be limited (Li, 1997), the assimilation of nitrogen and phosphorus in complex organic matter such as leaf litter and fine root production may alleviate the problems of nutrient enrichment associated with shrimp aquaculture. Corea et al. (1995) provides an account regarding the degradation of water quality in the Dutch canal, Sri Lanka, due to the release of wastewater from shrimp farms. Establishing mangrove communities to assimilate bioavailable nutrients present in the wastewater may represent an important strategy in ameliorating the observed negative environmental and social impacts. Litter from the forest would represent an important source of detritus, the basis for a number of food chains in coastal environments. Sheridan (1997) reported that the abundance of benthic fauna in sediments underlying red mangrove (*Rhizophora mangle*) stands in Rookery Bay, Florida exceeded that in neighbouring seagrass and non-vegetated mud habitats. Sheridan

(1997) observed that surface deposit feeders and carnivores dominated the benthic fauna in sediment underlying the red mangrove, and that this habitat provided high densities of small prey items for consumers that exploit the inter-tidal zone during high tide. Furthermore, Larsson et al. (1994) estimated that 4 ha of mangrove forest is required to supply detritus to supplement feed inputs to 1 ha of semi-intensively managed shrimp culture ponds. Detritus production in mangroves established to treat wastewater from shrimp farming may compensate for the appropriation of detritus in the culture system.

The abundance of detritus and prey items in mangrove communities combined with the complex architecture of aerial roots and branches make mangroves ideal nursery areas for juvenile fish and shrimp. Primavera (1998) found that riverine mangroves in Guimaras, the Philippines played an important role as nursery areas for penaeid shrimp. Kautsky et al. (1997) estimated that 9.6-160 m^2 of coastal mangrove is required to supply shrimp post-larvae to stock 1 m^2 of a semi-intensively managed shrimp pond. Therefore, the mangrove forest required for nutrient assimilation in this study (12.8-25 ha) may represent an important nursery area for juvenile shrimp, producing a significant proportion of the post-larvae required from restocking and supplementing the supply of juvenile shrimp produced in natural mangrove ecosystems. In addition to assimilating nutrients, producing detritus and providing nursery areas, mangrove forests sequester carbon released during processes associated directly and indirectly with semi-intensive shrimp aquaculture. Kautsky et al. (1997) estimated that 0.8-2.5 m^2 of forest is required to sequester carbon emitted as a consequence of culturing shrimp in 1 m^2 of a semi-intensively managed ponds.

4.4.4. Management issues

Constraints and opportunities associated with the development of the mangrove production system extend further than financial and economic considerations; integrating mangrove production with commercial shrimp farming poses several practical problems. The capacity of shrimp farm operators may need strengthening as managing the constructed wetland may demand skills more closely allied to forestry, a discipline that has different demands to that of aquaculture. Although cultivating mangrove trees may not capitalise on existing knowledge to the same extent as culturing shellfish and seaweed, mangroves are aquatic habitats and knowledge of hydrology, water quality and aquatic ecosystems possessed by shrimp farmers may contribute to formulating appropriate management strategies.

Traditional management strategies for mangrove plantations are based on rotations of 20-30 years in which a single age class of plants is cultivated; plantations are thinned at 5-10 year intervals during the rotation and finally clear felled when the trees reach maturity (Ong, 1982; Johnston, Clough, Xuan and Phillips, 1999). Ong (1982) reports that in traditional management regimes for mangrove plantations in Malaysia, the stand is thinned after 15 years, again at 20-25 years and clear-felled at 30 years. Proposing a similar regime, with irregular intervals between planting, thinning and felling, would have serious implications for the management and economics of horizontally integrated mangrove plantations. Should substantial quantities of marketable wood only be produced after 15-30 years, the relatively long period preceding a financial return may dissuade potential developers.

The cropping pattern for mangroves used to develop the ADEPT model was based on extracting wood products from the mangrove plantation on an annual basis. Johnston et al. (1999) proposed that harvesting a proportion of mangrove plantations equal to the reciprocal of the rotation length from mixed aquaculture-silviculture farming systems annually would provide farmers with a regular annual income. Although this may be feasible when harvesting large established mangrove stands, when considering relatively small, newly established plantations, this may not represent the best management strategy. Possible problems may be associated with the fact that it is generally more difficult to manage numerous cohorts in small plantations compared with a single age class; that stands of mangrove trees in particular areas are likely to be of similar ages and that the length of time to establish such a rotation is twice that of traditional systems.

A possible solution that addresses problems associated with both traditional and modified harvesting strategies proposed in scenarios developed in this study would be the establishment of a regional management strategy. This would encompass mangrove stands of varying ages on different farms and permit the optimal management of single age class plantations on individual farms, whilst cash-flows generated from the harvest of mature trees would be distributed amongst those farmers managing their plantations cooperatively. This strategy would help disperse risks associated with long-term investment required for forestry and facilitate the establishment of reciprocal arrangements for labour demands associated with planting, thinning and harvesting. It would also provide the collective of operators greater negotiating power when sourcing inputs and trying to access markets demanding regular supplies of timber and wood products. However, a range of factors may constrain the development of cooperative agreements; operators may be reluctant to form cooperatives, especially where the agreement would last for 20-30 years; the capacity of local institutions may not be sufficient to support the formal implementation of agreements and insecurity of tenure may prevent operators from entering into such contracts.

4.4.5. Practical constraints

Robertson and Phillips (1995) noted several constraints regarding the management of mangroves treating wastewater from shrimp farms. Mangroves are not homogenous and hydrological conditions depend on a number of physical and environmental features e.g. sediment characteristics, root architecture, canopy thickness and the activity of burrowing animals. Consequently, it is unlikely that wastewater will be dispersed evenly throughout the mangrove wetland. Uneven wastewater dispersion in the mangrove may cause short-circuiting, reducing the treatment efficiency of the constructed wetland, while some areas of the wetland may receive excessive wastewater loadings, resulting in the assimilative capacity of certain areas being exceeded.

Robertson and Phillips (1995) raise questions regarding the feasibility of using mangroves to treat aquaculture wastewater. Continuous flooding of mangroves with wastewater contrasts with the intermittent flooding caused by tides and terrestrial run-off that characterises the hydrology of natural mangroves. Furthermore, these authors question the long-term assimilative capacity of mangrove sediments for phosphorus, the influence of sediment type on nitrogen retention and transformation, and which silviculture practices optimise forest growth and nutrient uptake. Excessive loadings of TAN, particulate organic matter and dissolved organic matter may lead to anaerobiosis in the mangrove sediments. This could result in tree mortality and negative impacts on benthic fauna, including the loss of burrowing animals, which could have a severe impact upon the functioning of the ecosystem. Evidently before mangroves are promoted for treating wastewater from commercial aquaculture, the queries presented here require consideration.

4.4.6. Social and economic issues

As noted above, mangrove ecosystems are important habitats for juvenile shrimp. Furthermore, studies have shown that mangroves act as nursery areas for a number of other aquatic species, including several that are economically important in near-shore and offshore fisheries (Collette, 1983; Louis, Bouchon and Bouchon-Navaro, 1995; Kenyon, Loneragan, Hughs and Staples, 1997; Ronnback, Troell, Kautsky and Primavera, 1999). Therefore, mangrove areas created to treat aquaculture wastewater could potentially harbour the juveniles of species, that on emerging from the mangrove supplement populations may be exploited by local fishermen. This would constitute an important social benefit, especially in areas where mangrove forests that once fringed the coast have disappeared through development or over-exploitation. Supplementing stocks in near-shore and offshore fisheries could benefit local fishermen, but this resource may be open to exploitation by commercial fisheries, in which case the benefits may not be realised by the local community. Ensuring that local communities benefit from the creation of mangrove areas to treat aquaculture wastewater could assist shrimp farming to become more acceptable, alleviating feelings of disenfranchisement in the local community and reducing antagonism between shrimp farmers and other stakeholder groups.

According to Primavera (1995), artisanal fishing communities obtain a range of food items from mangrove forests. In the Philippines, extensive aquaculture i.e. introducing rock mounds known as *amatong* to encourage the aggregation of fish, fish cage culture, crab fattening in cages, seaweed culture and mollusc culture on stakes, may be employed to increase the incomes of coastal communities. Permitting coastal communities access to constructed mangrove wetlands, to culture food species and collect medicine and construction materials, must be balanced against the need to manage the forest correctly, thereby securing the predicted financial returns required for the economic sustainability of the enterprise. Extensive aquaculture production systems that may be integrated with the mangrove plantation could include cultivating seaweed, shellfish or crabs, although the impact of these culture practices on water quality should be considered. Studies have demonstrated that cultivating seaweed is an economically attractive livelihood strategy for members of coastal communities in Southeast Asia (Padilla and Lampe, 1989; Firdausy and Tisdell, 1991). Siar, Samonte and Espada (1995) described how small-scale enterprises culturing the slipper oyster (*Crassostrea iredalei*) represented a significant source of employment for women in Western Visayas, the Philippines. Opportunities for employing women in extensive aquaculture practices associated with mangrove plantations may represent an important social benefit of using constructed wetlands to treat wastewater from shrimp farms.

The constructed wetlands proposed here for integration with 1 ha of shrimp ponds, create work for 4-6 people; the provision of employment is an important benefit of aquaculture in less developed countries. For communities close to shrimp farms in, for example, India, Bangladesh, Indonesia and the Philippines, the development of constructed mangrove wetlands may confer significant social benefits on the local community. Where rice fields have been converted to aquaculture, the need to generate employment opportunities may be great; Shiva (1995; source Primavera, 1997) states that a shrimp farm covering 40 ha requires only 5 employees as opposed to an equivalently sized rice farm that would employ 50 workers. With the advent of commercial aquaculture in coastal regions, particularly in the tropics, local communities have frequently been denied access to natural resources upon which they depended and deprived of tenure over productive costal areas. Permitting access to mangroves established primarily for wastewater treatment could therefore have a significant positive impact on local coastal communities that are often vulnerable, and sometimes resentful of previous aquaculture developments.

4.4.7. Financial viability

In addition to generating employment for poor people from the local community, the integration of extensive aquaculture practices, such as those described by Primavera (1995), amongst mangrove trees in constructed wetlands may have a positive impact on economic returns generated by the enterprise. Enander and Hasselstrom (1994) suggest that economic loss associated with low shrimp yields could be offset by horizontally integrated production of shellfish. However, findings from this study suggest that the financial demands associated with establishing horizontally integrated mangrove plantations to treat wastewater from intensively managed shrimp ponds may constrain development by operators of small-scale farming enterprises. Financial indicators developed in scenarios presented here do not fully support this hypothesis. Employing conventional financial measures of success, the IRR's generated over the first ten years of operation (1.9% and 3.2%) would not be considered sufficient to justify investment in this innovative management strategy.

Considering the financial performance of the horizontally integrated mangrove plantations over twenty years, the IRRs generated increased (10.3% and 11.2%), however, returns at the levels predicted are unlikely to persuade operators to invest in this treatment strategy, especially where the opportunity cost for financial assets may be significantly higher and risks over the medium to long-term become less predictable. Despite apparent conclusions that may be drawn from these findings, a number of mitigating circumstances may be proposed that would increase the economic viability and attractiveness of this wastewater treatment strategy to shrimp farm operators. The sensitivity analysis demonstrated that returns over ten years were relatively insensitive to a 20% change in key input variables. However, where land and earthworks retain their value over the assessment period, or land costs are close to zero, investment in such treatment systems generates more favourable returns (12-14%).

Scenarios developed in this investigation are based on commissioning a constructed mangrove wetland; potential sites for establishing mangrove plantations include land that is currently used for agricultural production, abandoned shrimp ponds and marginal areas. The cost of establishing the mangrove plantation is based on the acquisition of agricultural land at the full market value, ensuring that the viability of the proposed strategy does not depend on appropriating marginal land that local communities may consider a common resource. In contrast, developing derelict shrimp ponds that are unable to support other aquaculture activities may assist in regenerating coastal areas that have been degraded through the development of unsustainable aquaculture practices. Furthermore, reclaiming derelict ponds for use as constructed wetlands for aquaculture wastewater treatment would require less capital investment.

Acquisition of agricultural land is likely to be considerably more expensive than developing derelict and marginal land, and may have serious consequences for the financial viability of the proposed wastewater treatment strategy. Assessing the financial performance of constructed mangrove wetlands, as outlined in this study, demonstrates that this strategy for managing shrimp culture wastewater generates relatively small IRR during the first ten years of operation, and increases to only relatively modest levels over twenty years.

However, financial returns generated could be enhanced significantly if it were possible to capitalise on other values attributable to mangroves, and not depend solely on income generation through wood sales. Ruitenbeek (1994) outlines a wide range of sustainable production uses of mangroves; other wood based products include timber, firewood and charcoal. The extensive culture of fish, shellfish and crustaceans represents another important sustainable production function of mangroves. Traditional practices of hunting, fishing and gathering in mangroves yield greater benefits than might be expected; in addition to capturing fish, shellfish and crustaceans from natural populations, other products include tannins, nipa, medicines and honey (Ruitenbeek, 1994; Ronnback, 1999). Furthermore, the mangrove ecosystem acts as a repository for genetic resources, a function that still requires evaluation.

Production functions of mangrove ecosystems represent tangible benefits, however, benefits attributable to mangroves can be extended to include regulatory, carrier and information functions (Ruitenbeek, 1994; Gilbert and Janssen, 1998). Evaluating both marketed and non-marketed goods supplied by mangroves resulted in estimates ranging between \$30-11,561 ha^{-1} y^{-1}; the maximum value reported was attributed to sustained production of wood products in a Malaysian mangrove (Primavera, 1995). If the developer of a mangrove plantation were able to capitalise on marketed and non-marketed goods supplied by this habitat, the development of constructed mangrove wetland to treat wastewater from shrimp culture may become a financially viable proposition. However, it is unlikely that the value of both marketed and non-marketed goods will be transferred to operators of shrimp farms that establish new mangrove areas, although it may be argued that these shrimp farms are currently subsidised by existing mangrove ecosystems.

4.5. Conclusions

This study demonstrates that the stock model developed for the ADEPT model, based on anticipated feed inputs and expected feed conversion ratios, may be applied readily to simulate the growth of shrimp stocked at low and high densities in intensively managed ponds. Predictions using the ADEPT model for waste concentrations present in water discharged from intensively managed shrimp ponds stocked at low and high densities are within the range reported in the literature. The ADEPT model could be refined further to account for the internal dynamics of nutrients and waste fractions in culture ponds and variable water exchange rates, however, potential benefits associated with these developments are not clear. It is argued that those waste fractions not discharged, entrained in the process water during routine water exchange, may ultimately be released from the culture system and appropriate environmental services from receiving ecosystem areas.

Regarding the treatment efficiency of constructed mangrove wetlands, the k-C* models employed here are based on relationships derived largely from freshwater wetlands in temperate environments. Therefore, without validation it is difficult to define the level of confidence with which the findings may be considered. Information collected from existing mangrove stands receiving wastewater with a comparable composition may be employed to formulate k-C* models that relate directly to the dynamics of waste fractions in marine and brackish-water forested wetlands dominated by mangrove. Before proposing constructed mangrove wetlands as appropriate treatment systems for wastewater from shrimp ponds, monitoring of the treatment performance of pilot-scale systems would be advisable. Information collected regarding the wastewater composition, treatment performance, management demands and financial costs would enable the validation of scenarios developed in this study and permit the refinement of the ADEPT model

Employing a nutrient mass balance approach to dimension the constructed wetland results in an excessive capacity of the mangrove plantation to assimilate nitrogen. However, as nitrogen usually limits primary production in marine and brackish-water environments it may be desirable to refine the ADEPT model, dimensioning the wetland with respect to nitrogen loadings, ensuring that this limiting nutrient does not reach the receiving environment whilst permitting the discharge of potentially less damaging phosphorus. This refined procedure may assist in developing optimally designed horizontally integrated mangrove plantations that minimise costs whilst ensuring the achievement of a significant degree of environmental protection.

Financial indicators considered here i.e. ten and twenty year IRR's, suggest that the horizontal integration of mangrove plantations for wastewater treatment may not represent an attractive investment opportunity for operators of intensively managed shrimp ponds. Over ten years returns may be considered insufficient, whilst over twenty years returns increase to modest levels; however, uncertainty due to risks may be assumed to increase with time. Despite these economic indicators, the management of small-scale mangrove plantations continues in many coastal regions in Southeast Asia (Binh, Phillips and Demaine, 1997, Johnston et al., 1999).

Integration of mangroves with aquaculture ponds was once practised in Indonesia; dikes and tidal flats surrounding the ponds or *tambak* were planted with *Rhizophora* spp., *Avicennia* spp. and other mangrove species to provide firewood, fertiliser (decaying leaves) and protection against erosion (Schuster, 1952, cited in Primavera, 1995). Similarly, at the turn of the century, the embankments of aquaculture ponds in the

Philippines were planted with rows of *Rhizophora* sp., *Sonneratia* sp. and nipa palm to protect against soil erosion induced through wind and wave action (Adams, Montalban and Martin, 1932, cited in Primavera, 1995). Recent initiatives have attempted to systematically investigate the benefits and opportunities associated with these integrated shrimp aquaculture-mangrove silviculture systems (Fitzgerald, 1999; Johnston et al., 1999). However, earlier recognition by operators and managers of the benefits associated with these traditional practices may have averted some of the environmental degradation that has accompanied the emergence of commercial shrimp culture.

Further investigation of investment and labour demands, cropping strategies and financial returns associated with existing aquaculture-silviculture farming systems is required to ensure design criteria and management regimes proposed for horizontally integrated mangrove plantations meet the demands and expectations of potential operators. Horizontally integrated mangrove plantations that incorporate secondary extensive or semi-intensive production systems (e.g. seaweed, shellfish or crustacean culture) that contribute to financial returns generated by the farming enterprise and provide employment opportunities for members of poor coastal communities appear to have the greatest potential. However, the assessment of these systems will require the development of refined modelling scenarios and an increased understanding of the biological, managerial and financial demands associated with these diverse farming enterprises. Finally, developing horizontally integrated mangrove plantations removes the reliance of shrimp farms on environmental goods and services appropriated from supporting ecosystem areas, internalising costs associated with discharging waste to the receiving environment, whilst addressing concerns raised by environmentalists and stakeholder groups regarding the externalities associated with intensive shrimp culture.

Chapter Five

Comparison of conventional, rational and traditional strategies for lagoon-based wastewater treatment and reuse

5.1. Introduction

Lagoon-based wastewater treatment systems are commonly designed to produce discharge water that attains the recommended reduction in pathogens prior to reuse. However, a *rational* design approach proposed by Mara et al. (1993)[1] represents an attempt to simultaneously optimise wastewater treatment and fish production in lagoon-based systems. Here, the treatment lagoons are configured so that N is retained in the discharge water, increasing its value as a fertilizer, while dilution in the fishpond results in microbiologically safe culture conditions. These authors recommend that wastewater should be treated and retained for one day in a 2 m deep anaerobic lagoon, followed by a five day retention time in a 1.5 m deep facultative pond, before being discharged to fishponds. The main difference between the rational design and the *conventional* approach is the absence of maturation ponds. In conventional treatment, maturation ponds serve to lower the faecal coliform count to levels considered microbiologically safe prior to reuse. However, Mara et al. (1993) assert that an initial 99% reduction in pathogen numbers occurs in the fishpond through dilution, followed by a rapid die-off of faecal coliforms over the next 30 hours, accounting for a further 99% reduction. Therefore, the risk of fish

[1] Mara (1997) subsequently suggested a modified version of the model incorporating a verification step to assess the concentration of ammonia entering the fishpond.

being contaminated with pathogens and compromising the safety of cultured products is significantly reduced.

Traditional wastewater aquaculture practices at the Calcutta peri-urban interface have been described by a number of authors (Olåh, Sharangi and Datta, 1986; Ghosh, 1990; Kundu, 1994; Muir, Walker and Goodwin, 1994; Mukherjee, 1996; Jana, 1998). An account of the origins and evolution of these practices, together with an extensive review of their current status, including culture methods, production figures and management strategies is given by Kundu (1994). Fishponds managed for wastewater aquaculture at the Calcutta peri-urban interface cover 3,000 ha, with individual fisheries ranging in size from 0.4-162 ha (Mukherjee, 1996). It has been estimated that these ponds receive 550,000 m^3 d^{-1} of wastewater and produce an annual yield of 13,000 t of fish (Mara et al, 1993). However, despite its long history and continued operation, a number of problems have been reported with this system, including declining productivity and diminishing financial returns (Kundu, 1994; Mukherjee, 1996). Therefore, it was decided to develop a modified version of the ADEPT model suitable for assessing the treatment and financial performance of lagoon-based wastewater treatment approaches. The objective was to identify possible areas for improving current design and management approaches used in the Calcutta peri-urban system, and to highlight opportunities for improving engineering designs for lagoon-based wastewater treatment and reuse.

5.1.1. Study aims

Within the framework of the ADEPT model, this case study aims to compare the physical, financial, practical, economic and social consequences of adopting a rational design for lagoon-based wastewater treatment and aquaculture production, with the conventional design approach, and traditional reuse system operating in peri-urban Calcutta. Results are viewed from a systems perspective, considering implications for health, employment, environmental protection and nutrient recovery. Potential outcomes associated with adopting the rational design criteria are described with respect to the needs and expectations of stakeholders, while strategies with potential to facilitate appropriate modifications to existing practices to enhance benefits are discussed.

5.2. Method

Using the modified framework of the ADEPT model, rational design criteria proposed by Mara et al. (1993) were used in the first scenario to dimension anaerobic lagoons and facultative ponds with the capacity to treat 550,000 m^3 d^{-1} of wastewater, equal to the volume flowing to peri-urban fishponds in Calcutta. In the second, design criteria presented by Mara (1997) for a conventional lagoon-based treatment system were used. In both scenarios the predicted N concentrations following treatment were used to dimension the fishpond area. Levels of fish production were extrapolated from expected yields reported by Mara et al. (1993) and faecal coliform numbers in water entering the fishpond were used to assess public health risks. Financial indicators were estimated using the approaches described in previous chapters and systems designed using the rational and conventional design approaches were compared with those in peri-urban Calcutta.

5.2.1. Wastewater characteristics

The two modelling scenarios were developed using baseline data presented by Mara et al. (1993); these data were suggested as being representative of typical conditions encountered in West Bengal, India. These authors reported typical BOD and N concentrations for wastewater of 200 mg l^{-1} and 50 mg l^{-1}, respectively; faecal coliform numbers were 5 x 10^7 100 ml^{-1}, the pH was 8, and the mean temperature and net evaporation were 25°C and 5 mm d^{-1}, respectively.

5.2.2. Conventional and rational design approaches

In both scenarios, following the design criteria presented by Mara et al. (1993) an additional land area, equivalent to 55% of that required to accommodate the treatment lagoons and fishponds, was included to accommodate pond embankments, access roads and associated infrastructure.

Rational design approach

Detailed information on the rational design approach was given by Mara et al. (1993) and key design features are presented in Table 5.1. The mid-depth area of the anaerobic lagoon is found by multiplying the flow volume (m^3) by the retention time (1 day) and dividing the product by the depth of the pond (2 m). The volumetric BOD loading (λ_V, g m^{-3} d^{-1}) is used to validate the anaerobic lagoon design and may be calculated using the equation given by Mara (1997):

$$\lambda_V = L_i Q / V_a$$

where,	L_i	=	influent BOD concentration (g m^{-3})
	Q	=	flow rate (m^3 d^{-1})
	V_a	=	anaerobic lagoon volume (m^3).

At 25°C the maximum permissible design load for BOD is 300 g m^{-3} d^{-1} (Mara and Pearson, 1986; cited in Mara et al., 1993); the estimated loading on the pond in this case was 200 g m^{-3} d^{-1} and considered satisfactory.

The mid-depth area of the facultative lagoon is found by multiplying the flow volume (m^3) by the retention time (5 d) and dividing the product by the depth of the pond (1.5 m). As with the anaerobic pond, the design was validated using the surface BOD loading; at 25°C the maximum permissible loading for the facultative lagoon is 350 kg ha^{-1} d^{-1} (Mara, 1987; cited in Mara et al., 1993), here the estimated loading was 180 kg ha^{-1} d^{-1} and considered satisfactory.

Mara et al. (1993) discounted evaporation from the facultative lagoon due to a relatively short retention time, however, it is considered here as the ADEPT model could dimension large treatment systems with significant evaporation. Evaporation from the anaerobic lagoon is not considered because under normal operating conditions surface scum significantly reduces the rate of evaporation (Mara, 1997).

The fishpond was designed using an optimal N loading rate of ~4 kg ha^{-1} d^{-1} (Edwards, 1992), therefore, prior to dimensioning this component, the effects of the preceding treatment lagoons on the N concentration were estimated. Following the example of Mara et al. (1993) it was assumed that no N removal occurs in the anaerobic pond; however, N removal by the facultative pond is significant and was estimated using the relationship established by Reed (1985; cited in Mara et al., 1993):

$$C_e = C_i \exp\{-[0.0064(1.039)^{T-20}][Ø+60.6(\mathrm{pH}-6.6)]\}$$

where,	C_e	=	facultative pond effluent TN concentration (mg l^{-1})
	C_i	=	facultative pond influent TN concentration (mg l^{-1})
	T	=	temperature (°C)
	Ø	=	retention time (d)

Assuming the TN concentration in the in-flowing water (C_e) is 50 mg l^{-1}, at pH 8, the resulting concentration in the facultative pond effluent (C_i) is 24.9 mg l^{-1}. Therefore, with a discharge volume from the facultative lagoon of 540,900 m^3 the mass flow of N was estimated at 13,480 kg d^{-1}, sufficient to meet the desired N loading rate in 3,370 ha of fishponds.

The expected concentration of faecal coliforms in water used to culture fish was estimated using the relationship established by Marais (1974; cited in Mara et al., 1993):

$$N_p = N_i/(1+k_T Ø_a)(1+k_T Ø_f)(1+k_T Ø_p)$$

where,	N_p	=	faecal coliforms number (100 ml^{-1}) in fishpond
	N_i	=	faecal coliforms number (100 ml^{-1}) in untreated wastewater
	k_T	=	rate constant for faecal coliforms removal d^{-1} $(2.6(1.19)^{T-20})$
	$Ø_a$	=	anaerobic lagoon retention time (d)
	$Ø_f$	=	facultative lagoon retention time (d)
	$Ø_p$	=	fishpond retention time (d)

Table 5.1: Design parameters for lagoon-based wastewater treatment and aquaculture reuse.

Component	Design criteria	Conventional	Rational[1]
Anaerobic pond	Retention time (d)	1	1
	Depth (m)	4	2
	Volumetric BOD loading for validation	<300 g m^{-3} d^{-1}	<300 g m^{-3} d^{-1}
Facultative pond	Retention time (d)	4	5
	Depth (m)	1.5	1.5
	Aereal BOD loading for validation	<350 kg ha^{-1} d^{-1}	<350 kg ha^{-1} d^{-1}
Maturation ponds	Depth (m)	1.5	-
	Number of ponds (n) where: faecal coliform count (100 ml^{-1}) required in the maturation pond effluent is N_e.	$N_e = N_i(1+k_T Ø_m)^n$	-
	Minimum retention time (d)	3	-
	Aereal BOD loading for validation	< 350 kg ha^{-1} d^{-1}	-
Fishpond	Depth (m)	1	1
	Optimal N loading rate	4 kg ha^{-1} d^{-1}	4 kg ha^{-1} d^{-1}

[1]Refers to the rational design proposed by Mara et al. (1993)
[2]Ne for unrestricted irrigation recommended by WHO (1989) i.e. 1 x 10^3 100 ml^{-1}

Conventional design approach

Although the same general design criteria used in the rational design approach were employed to dimension the anaerobic lagoons and facultative ponds required for the conventional treatment systems, slight modifications were performed to replicate the design criteria proposed by Mara (1997). The depth of the anaerobic lagoon was increased to 4 m and the target retention time in the facultative lagoon reduced to 4 days. Excluding these minor alterations and the presence of maturation ponds, all other input variables were constant between the two simulations.

Design criteria for dimensioning the maturation ponds and predicting the treatment effect of these additional components was taken from Mara (1997). However, prior to dimensioning the maturation ponds, the required level of wastewater treatment was selected using the ADEPT model. The available options permitted the calculation of the optimum number of maturation ponds needed to reduce faecal coliform levels to $<1 \times 10^3$ and $<1 \times 10^2$ per 100 ml, guideline levels proposed by the World Health Organization for unrestricted and restricted irrigation, respectively (WHO, 1989). The number of maturation ponds required to achieve the desired treatment levels for the second and third options was derived from a modified version of the equation presented by Mara (1997) for increasing values of n (the number of maturation ponds):

$$N_e = N_i/(1+k_T Ø_m)^n$$

where,	N_i	=	faecal coliforms in untreated wastewater (100 ml^{-1})
	N_e	=	faecal coliforms in maturation pond discharge (100 ml^{-1})
	$Ø_m$	=	retention time (d) in single maturation pond in series (>3 d)
	k_T	=	areal rate constant for faecal coliform removal

Following the recommendation by Mara (1997) a sub-routine was developed in the ADEPT model to verify that the BOD loading on the first maturation pond was below that on the preceding facultative lagoon.

As with N removal in the facultative lagoon, the relationship established by Reed (1985; cited in Mara et al., 1993) was used to estimate the effect of the maturation ponds; removal in each was estimated sequentially. Having estimated N removal in the maturation ponds, the recommended N loading rate given above was used to dimension the fishponds.

5.2.3. Aquatic production and nutrient retention

Production in the fishponds was based on the cropping pattern proposed by Mara et al. (1993) where tilapia were cultured in ponds with an average size of 1 ha. It is possible to drain ponds of this size, permitting a complete harvest and enabling them to be quickly restocked (Mara et al., 1993). Adopting such a culture system, employing a relatively short grow-out phase of 4 months and stocking 20 g fingerlings at a rate of 3 m^{-2}, it should be possible to produce three crops of 200 g fish annually. Allowing for a 25% loss of fish through poaching, predation and mortality, an overall production rate of 13 t ha^{-1} y^{-1} was extrapolated. N and P retention in the fish biomass harvested from the ponds managed for wastewater aquaculture was calculated using a mass-balance equation. Average concentrations for N (14 g kg^{-1}) and P (5 g kg^{-1}) found in carcasses of freshwater fish were used to estimate nutrient retention through assimilation in fish biomass. Net production was used to account for N and P inputs in the form of fingerlings.

5.2.4. Financial inputs and implications

Although Mara et al. (1993) provided financial data for the development and operation of lagoon-based wastewater treatment systems in West Bengal, more recent information on financial costs and benefits was taken from sources that refer specifically to peri-urban

Calcutta (Kundu, 1994; Mukherjee, 1996 and Mukherjee, personal communication, 1998); these costs are summarised in Table 5.2.

The cost of land was assumed at £2,060 ha^{-1} and lagoon and fishpond construction was expected to cost £3,960 ha^{-1} (assuming a worker requires 30 days to develop 1 *katha* (720 ft^2) of land, and receives a wage of £0.88 d^{-1}; Mukherjee, personal communication, 1998). Control structures for regulating the flow of wastewater between treatment lagoons and fishponds were the only infrastructure costs included in the scenarios. Considering the likely scale of development it was assumed that one control structure would be required for each hectare of lagoons and fishponds; the cost of control structures was estimated at £50 per instillation (Mukherjee, personal communication, 1998).

Mara et al. (1993) include nets, boats, vans and bicycles as financial costs associated with the rational design approach, however, operators of fisheries at the Calcutta peri-urban interface do not provide vans and bicycles. Expenditure on boats and nets was expected to equate to £295 and £75 per employee, respectively; miscellaneous equipment required by the fisheries personnel was estimated to cost ~£5 per employee (Mukherjee, personal communication, 1998). Variable operating costs included the purchase of fingerlings at £0.01 each (Mukherjee, personal communication, 1998) and materials to maintain the system, assumed at 1% of initial costs. Fixed operating costs included wages to pay employees and a manager to coordinate the operation. Following the labour requirements outlined by Mara et al. (1993), it was assumed that 1 employee would be required for each hectare developed, whilst 1 manager would be required for every ten hectares. The cost of engaging an employee and manager was assumed at £410 y^{-1} and £825 y^{-1}, respectively (Mara et al., 1993). Income would be generated through the sale of fish and the expected market value was £0.37 kg^{-1} (Morrice, Chowdhury and Little, 1998).

Table 5.2: Financial parameters assumed for the two scenarios.

Parameter	Value	Units
Capital costs (CC)		
Land	2,060	£ ha^{-1}
Site development	3,960	£ ha^{-1}
Control structures	50	£ ha^{-1}
Boats, nets and miscellaneous equipment	375	£ ha^{-1}
Operating costs		
Maintenance	1% CC	
Employees	410	£ ha^{-1}
Management	825	£ 10 ha^{-1}
Fingerlings	0.01	£ each
Returns		
Fish sales	0.37	£ kg^{-1}

As in the previous two chapters, the financial implications of investing in each of the scenarios described above was assessed using a conventional discounted cash flow approach. The ten year NPV was calculated at discount rates of 5, 10, 15 and 20 per cent, whilst the IRR was calculated over ten years. The salvage value of materials and equipment was calculated as in the previous case studies, and cash flows during the financial assessment period included the replacement of infrastructure and equipment that has exceeded its economic life.

5.3. Results

5.3.1. Physical characteristics

Physical characteristics of the lagoon-based treatment and production systems designed using the conventional and rational design approaches are presented in Table 5.3. The depth of anaerobic ponds in the rational (2 m) and conventional (4 m) designs caused the area required for a retention time of 1 day, to be reduced from 27.5 ha in the rational design to 13.8 ha in the conventional. The area occupied by the facultative lagoon in the rational system was 182 ha, whilst in the conventional system the facultative lagoon occupied 146 ha. This results from the lower hydraulic retention time employed when designing conventional facultative lagoons.

Two maturation ponds required to produce an effluent suitable for unrestricted irrigation in the conventional system had an area of 215 ha and retention time of 6 days. In total the treatment basins required for the conventional system had an area of 374 ha, whilst in the rational system the area was 209 ha. The fishpond area under the conventional design was estimated at 856 ha, significantly less than the 3,370 ha required for the rational design. This difference in fishpond area arises from the N removal facilitated by maturation ponds in the conventional system. Land required for access roads and supporting infrastructure meant the total area required for the rational design system was ~5,550 ha, almost three times that required for the conventional design at 1,907 ha.

5.3.2. Water flow and evaporation

The greater area required for the rational design, as compared with conventional design, is a consequence of the greater fishpond area that could be supplied with optimal N loading. Fishponds in the conventional system generate a hydraulic retention time of 16 days, as compared with 62 days in the rational system. Overall, water passing through the conventional system was retained for 27 days, while in the rational system the retention period was 68 days. Variation in the surface area of the basins in the two scenarios results in the predicted difference in water loss through evaporation. Evaporation from the conventional and rational systems was estimated at 60,840 and 177,600 m^3 d^{-1}, respectively. Consequently the volume of water discharged to downstream users or the receiving environment from the conventional system (489,160 m^3 d^{-1}) would be considerably larger than that from the rational design system (372,400 m^3 d^{-1}).

Table 5.3. Physical characteristics of lagoon-based treatment systems dimensioned using conventional and rational design approaches.

Systems characteristics	Conventional	Rational
Anaerobic lagoon area (ha)	13.8	27.5
Facultative lagoon area (ha)	146	182
Maturation pond area (ha)	215	-
Fishpond area (ha)	856	3,370
Additional area required (%)	55%	55%
Total area required (ha)	1,907	5,550
Total hydraulic retention time (d)	27	68
Discharge volume (m^3 d^{-1})	489,160	372,400

5.3.3. Nitrogen dynamics

It was estimated that water discharged from the maturation ponds in the conventional system would contain ~6.4 mg l^{-1} of N, as compared with 25 mg l^{-1} in water discharged from the facultative lagoon. N removal in the treatment components of the conventional system was calculated at 87%, as compared to 50% in the rational design system. N removal by the facultative lagoon in the conventional and rational systems was comparable at 49.8% and 50.2%, respectively. However, the maturation ponds facilitated greater N removal in the conventional system; the influent concentration was reduced by ~50% in each pond.

Treated water discharged from facultative lagoons in the rational design system contains ~25 mg l^{-1} of N, equivalent to a mass loading of 13,480 kg d^{-1}, and sufficient to meet the optimal N loading rate for 3,370 ha of fishponds. Water discharged to the fishponds from the maturation ponds in the conventional system contains ~6 mg l^{-1}, ~26% of the N present in water from the facultative lagoons in the rational system. Accounting for evaporation from the maturation ponds, 532,000 m^3 d^{-1} of water flowing to fishponds in the conventional system contains 3,420 kg of N, sufficient to meet the optimal loading rate for 856 ha of fishponds.

5.3.4. Aquatic production and nutrient assimilation

Fish production and nutrient assimilation in the two systems are summarised in Table 5.4. Based on the predicted yield from fishponds managed for wastewater aquaculture of 13 t ha^{-1} y^{-1} (Mara et al., 1993), fish production in the two scenarios considered here is directly proportional to fishpond area. Consequently, gross fish production from the conventional and rational systems was estimated at 11,560 and 45,500 t y^{-1}, respectively. Net fish production from the conventional and rational systems was estimated at 10,400 and 40,950 t y^{-1}, respectively. Employing these production values the ADEPT model calculated that of 10,040 t y^{-1} of N discharged to the treatment system, 573 t y^{-1} or 5.7% was recovered in fish produced in the rational design system and 146 t y^{-1} or 1.5% was recovered in fish from the conventional system. The areal rate of N assimilation in fish biomass equates to 0.5 kg ha^{-1} d^{-1} in both systems. On a daily basis ~37 t of P was discharged to the treatment system, equating to a mass flow of 13,530 t y^{-1}. Fish biomass produced in the rational design system retains 197 t of P, equivalent to 1.5% of the annual mass flow, fish produced in the conventional system retain 49 t of P, or 0.4% of the annual mass flow. P assimilation in fish biomass equates to 0.16 kg ha^{-1} d^{-1} in both systems.

5.3.5. Faecal coliform levels

Faecal coliform levels in water flowing to the fishponds in the conventional and rational design scenarios were 7 x 10^2 and 2.2 x 10^5 100 ml^{-1}, respectively. The predicted removal of faecal coliform bacteria in fishponds results in water discharged to downstream users or the receiving environment from the conventional and rational systems containing levels of ~7 and 6 x 10^2 100 ml^{-1}, respectively.

Table 5.4. Production, nutrient assimilation and faecal coliform levels in fishponds.

Systems characteristics		Conventional	Rational
Gross fish production (t y^{-1})		11,560	45,500
Net fish production (t y^{-1})		10,400	40,950
N retention in fish biomass (t y^{-1})		146	573
N recovered from wastewater (%)		1.5	5.7
Areal rate of N recovery (kg ha^{-1} d^{-1})		0.5	0.5
P retention in fish biomass (kg y^{-1})		49	197
P recovered from wastewater (%)		0.4	1.5
Areal rate of P recovery (kg ha^{-1} d^{-1})		0.16	0.16
Faecal coliform level:	entering fishpond (100 ml^{-1})	7×10^2	2.2×10^5
	in final site discharge (100 ml^{-1})	7	6×10^2

5.3.6. Financial indicators

Rational design approach

Capital costs of £35.67 million associated with adopting the rational design are due mainly to the expense of developing the basin area, estimated at £21.97 million; a further £11.43 million was required to purchase the land required; costs associated with the development are detailed in Table 5.5. Additional capital costs of £2.27 million were associated with developing infrastructure at the site, purchasing nets, bicycles, boats and miscellaneous equipment. Operating costs associated with the rational design were £6.2 million y^{-1}; variable costs included £3.13 million to purchase fingerlings and £0.36 million y^{-1} for maintenance; fixed costs consisted of £2.28 million y^{-1} for full-time workers and £0.46 million y^{-1} for a manager.

Conventional design approach

Capital costs of £12.3 million associated with the conventional system are significantly lower (£23.4 million) as compared to adopting the rational design; land for the development was expected to cost £3.9 million and £7.6 million was estimated for basin construction. An additional £0.8 million would be required to purchase nets, bicycles and miscellaneous equipment and develop infrastructure. Operating costs for the conventional system amounted to £1.9 million y^{-1}; including £0.79 million to purchase fingerlings and £0.12 million for maintenance; employment of ~1,900 full-time employees represented a fixed cost of £0.78 million y^{-1}.

Table 5.5: Financial indicators for conventional and rational designs.

Systems characteristics		Conventional	Rational
Capital costs (£ million)		12.3	35.67
Operating costs (£ million y^{-1})		1.9	6.2
Profit excluding depreciation (£ million y^{-1})		2.4	10.6
Rate of return on initial capital cost (% y^{-1})		19.7	29.7
Rate of return on operating costs (% y^{-1})		130	171
Payback period		5	3.4
NPV (£ million) at:	5%	2.8	31.7
	10%	-0.3	17.2
	15%	-2.3	7.4
	20%	-3.7	0.6
IRR (%) over	10 years	9.5	20.1

Comparative performance

Accounting for capital costs, operating costs and income, general financial indicators were calculated. Excluding depreciation, the annual profits generated by the rational and conventional systems were estimated at £10.6 and £2.4 million, respectively. The rate of return on capital and operating cots for the rational design was 29.7% and 171%, respectively, whilst for the conventional design the rate of return on capital and operating costs was 19.7% and 130%, respectively. A payback period of 3.4 years was estimated for the rational design; for the conventional systems the estimated payback periods was higher at 5 years. The gross value of fish produced in the rational and conventional systems was estimated at £16.8 and £4.3 million, respectively.

The ten year NPV of the cash flow for the rational design system was estimated at £31.7, £17.2, £7.4 and £0.6 million at discount rates of 5, 10, 15 and 20 per cent, respectively. The IRR over ten years was estimated at 20.1%. The cash flow generated by the conventional system resulted in ten year NPV's of £2.8, -£0.3, -£2.3 and -£3.7 million at discount rates of 5, 10, 15 and 20 per cent, respectively. The IRR was estimated at 9.5%, over ten years.

5.3.7. Sensitivity analysis

As with previous chapters a sensitivity analysis was undertaken, investigating the impact of key variables on the ten year IRR generated by the different treatment approaches; findings are summarised in Table 5.6.

Increases and reductions of 20% in land, site development and labour costs had little impact on the returns generated, <5% for both the conventional and rational designs. However, in both cases, a 20% change in the value of fish resulted in a corresponding rise or fall in the IRR of ~10%; a 20% fall in the value of fish meant the conventional system was unable to generate a positive IRR. Assuming either that the land cost was zero or that it retained its value resulted in a slight increase in the IRR; assuming both the land and earthworks retained their value resulted in the second highest returns of 19% and 32.4% for the conventional and rational systems, respectively. However the best returns predicted resulted from decreasing the land area required for access and infrastructure to 6.5%.

Table 5.6. Sensitivity analysis of ten year IRR's associated with conventional and rational designs.

Sensitivity	Conventional	Rational
Baseline	9.5	20.6
Land cost (+20%)	8	18.8
Land cost (-20%)	11	22.5
Site development (+20%)	6.9	17.3
Site development (-20%)	12.6	24.4
Cost of labour (+20%)	7.6	18.8
Cost of labour (-20%)	11.4	22.3
Value of fish (-20%)	-	9.8
Value of fish (+20%)	17.8	30.2
Land retains its value	12.2	22
Land and earthworks retain their value	15.9	24.5
Land has a cost of zero	19	32.4
Additional land requirement reduced to 6.5%	23.3	36.4

5.4. Discussion

5.4.1. Physical characteristics

The area of fishponds required for the conventional (856 ha) and rational (3,370 ha) designs represent a significant difference between the two options. This variation is a consequence of N removal facilitated by the maturation ponds in the conventional system, and the fact that the fishponds in both scenarios are dimensioned using a fixed loading rate for N.

The provision of additional land for access roads and supporting infrastructure in the base case scenarios means the total area required for the rational design (5,550 ha) is ~3 times that required by the conventional design (1,907 ha). However, this added demand for land may be difficult to justify in peri-urban areas where pressure on land resources is likely to be considerable. Reducing the area allocated for supporting infrastructure from an additional 55% to 6.5%, the proportion of land not occupied by ponds in the functional systems at the Calcutta peri-urban interface, may become more appropriate. The sensitivity analysis demonstrated that such a reduction has a beneficial impact on the ten year IRR generated by both systems. However, despite this, where land is available at a reasonable cost, the inclusion of an additional land area in the design would permit the future expansion of the treatment system if extra capacity were required.

When proposing the modification of the existing wastewater aquaculture system at the Calcutta peri-urban interface, the area required to implement the rational design would represent an important constraint, as suitable land for development may be limited. The land area required to accommodate the treatment lagoons and fishponds required for the rational design (5,550 ha) is greater than the existing 3,000 ha of fishponds managed for wastewater aquaculture (Kundu, 1994), without the addition of extra land for access roads and supporting infrastructure. Acquisition of suitable land at the urban fringe represents a major constraint to ecologically based wastewater reuse systems. Minimising the area required would help limit conflicts with other stakeholder groups; however, it should be ensured that design criteria are not compromised as this could lead to a systems failure.

Implementation of the rational design criteria would require formal arrangements for lagoon-based wastewater treatment. This would either require the construction of new lagoons at the urban fringe or conversion of existing fishponds to treatment lagoons. The availability of land close to the urban fringe may prevent construction of new lagoons specifically for treatment; operators of existing fisheries may be unwilling to convert their fishponds to treatment lagoons unless adequately compensated, although this could place unacceptable demands on public finances. It may be possible to raise sufficient funds by taxing other users of the wastewater resource or selling part of the land to a developer; revenue raised could also be used to subsidise the operation of the treatment lagoons. Irrespective of the strategy selected, appropriate institutional processes and structures would be required to ensure the equitable distribution of benefits.

5.4.2. Water conservation

Results of the modelling exercise presented here indicate that water conservation is more efficient in a conventional system than when the rational design criteria are used. The restricted loss of water from the conventional system (11%), as compared with the rational design system (32%), is a consequence of the lower surface area in the conventional system. The additional volume of water discharged from the conventional system may represent a significant advantage over the rational design when considering possible environmental requirements and the needs of downstream users. Water from fishponds could be used to irrigate agricultural crops and recreational areas, or to revive degraded watercourses; water discharged from the existing ponds managed for wastewater aquaculture at the Calcutta peri-urban interface is used downstream to irrigate rice fields (Mara, 1997). Creating wetland areas using the treated wastewater could confer a range of benefits to society and the environment e.g. flood protection, ground water recharge and wildlife habitat (Burbridge, 1994). However, to effectively plan and manage the distribution and exploitation of the wastewater resource the relative merits of the different reuse practices should be considered; to evaluate this, the ADEPT model could be developed further to evaluate the various reuse options.

5.4.3. Nutrient retention and reuse

As mentioned previously, N dynamics in the treatment lagoons have a significant influence on treatment system design; however, N recovery in fish biomass accounts for only a small proportion of the total mass flow entering the systems (1.5% with the conventional design and 5.7% with the rational design). P retention in the conventional and rational systems is less efficient (0.4% and 1.5%, respectively). Although N and P recovery in the rational design system is significantly higher than that in the conventional system, it is apparent that the efficiency of nutrient recovery through fish culture in ponds managed for wastewater aquaculture is still limited. Therefore, promoting wastewater aquaculture as an effective approach to the recovery of nutrients from wastewater may be inappropriate.

The recommended N loading for fishponds (4 kg ha^{-1} d^{-1}) is significantly higher than the rate at which N is assimilated in fish biomass (0.5 kg ha^{-1} d^{-1}). This difference between recommended nutrient loading and efficiency of resource recovery demonstrates the need for a shift in the way pond-based wastewater aquaculture systems are viewed. Intensification of nutrient flows within aquatic systems through the integration of additional culture practices may be a potential strategy. Linking the flow of nutrients within aquatic systems to terrestrial production offers another. The model system that may be considered is dike-pond farming, developed and refined in China over the past two thousand years.

N and P dynamics could be investigated using a range of approaches including field trials, pilot scale studies, laboratory experiments and mathematical modelling. Ruddle and Christensen (1993) employed the Ecopath II model to develop an energy flow model for mulberry dike-carp ponds in the Zhujiang Delta, Guangdong Province, China. Applying a similar approach to explore nutrient dynamics in wastewater aquaculture systems incorporating integrated production practices may assist in developing more efficient resource recovery strategies. Further development of the ADEPT model to account for production in integrated production systems i.e. dike-cropping, could have potential to increase the financial returns generated. However, the integration of further production practices may be constrained through increased management demands and requirement for greater support from local institutions and development agencies.

Currently there is no formal strategy for treating wastewater prior to reuse in fishponds at the Calcutta peri-urban interface. Operators rely on processes in feeder canals for passive treatment; however, the treatment effect has not been described. The hydraulic retention time in these canals has been estimated at ~0.5 days and it is assumed that there is no N removal (Edwards, personal communication, 1997). Kundu (1994) reported that 3,000 ha of ponds were managed for wastewater aquaculture in peri-urban Calcutta. Based on the predicted mass flow of N in untreated wastewater (27.5 t d^{-1}), it is estimated that these ponds are loaded with 9.2 kg N ha^{-1} d^{-1}, over twice the recommended rate (4 kg ha^{-1} d^{-1}). Excessive N loading in fishponds at the Calcutta peri-urban interface may be responsible for recent management problems encountered, in particular blue-green algae blooms (Ghosh, personal communication, 1998). Such blooms are often associated with off-flavours in fish and the production of toxic substances that can harm fish and pose a threat to consumers; this represents a potential constraint to the sustainability of the culture system. However, despite the model outputs and corroboratory evidence concerning algal blooms, more recent interviews with pond operators revealed a widespread perception that limited wastewater availability was restricting primary production and constraining fish growth (Kundu, personal communication, 2000). A likely explanation for these differing observations is a variation in the distribution and availability of wastewater to different fisheries, although further work is required to resolve these contradictions.

5.4.4. Financial indicators and productivity

Due to the difference in land and development costs, the conventional design requires significantly less capital investment, as compared with the rational design approach. This could be important, as implementing agencies may be more willing to invest in systems with lower initial start-up costs. Furthermore, comparing capital costs of the two systems raises questions regarding relative financial risks, the lower costs associated with the conventional system appear to represent a reduced risk. However, the higher rate of return on initial capital costs associated with the rational design (29.8%), as compared with the conventional design (19.7%), indicates that the performance of the rational design system is more resilient to increasing costs and decreasing financial returns.

During the first ten years of operation, implementation of the rational design generates an IRR (20.6%) that indicates financial viability, and could attract entrepreneurs from the private sector. The lower IRR generated by the conventional system during the same period (9.5%) indicates that this system may be less attractive to investors. As mentioned previously, from a public sector perspective, an IRR of 5% can be considered viable for projects that confer social benefits (Bojo, 1991). Furthermore, when evaluating wastewater treatment options it is important to consider the cost associated with alternatives to lagoon-based systems. Considering some of the alternatives i.e. aerated lagoons, oxidation ditches and conventional biofilter treatment, Mara (1997) estimated that ponds were the cheapest option when land costs were low.

The higher rate of return on capital invested in the rational design is a consequence of the larger proportion of the system being devoted to producing fish. Income generated through the sale of fish effectively subsidises capital and operating costs associated with constructing treatment components. The level of subsidy could be enhanced through the integration of additional production practices; however, suitable approaches to evaluating benefits generated in complex integrated systems are poorly developed. Therefore, modification of the ADEPT model to account for financial returns from integrated systems may represent an important area for future research. Enhancing the financial returns generated could increase the level of private sector interest in wastewater treatment combined with reuse, an important consideration in developing countries where authorities may be unable to implement wastewater treatment projects due to limited financial resources and inadequate institutional capacity.

Comparing the performance of the conventional and rational design approaches to the existing system in peri-urban Calcutta, a key difference is the achieved level of fish production. In the simulations it was assumed at 13 t ha^{-1} y^{-1}, however, yields reported for the Calcutta system may be estimated at ~4.3 t ha^{-1} y^{-1} (13,000 t y^{-1} from 3,000 ha of ponds). Providing the means to regulate the level of water in the existing system would allow ponds to be drained easily, permitting an efficient harvest at regular intervals. Draining ponds frequently would also allow the removal of predatory and small wild fish that can constrain production. Adapting existing management practices relating to stocking and harvesting strategies may represent the most promising approach to improving the finances of the existing system. Adopting management practices utilised in other large-scale fisheries e.g. the culture of tilapia in Israeli reservoirs (Fischer, 1997), may also represent a significant opportunity.

Establishing harvest strategies based around three distinct grow-out periods, as recommended by Mara et al. (1993), may improve the efficiency of fish production, however, fish are currently supplied to local markets on a daily basis, ensuring a continuous supply of fresh fish. Adopting the proposed three-harvest strategy could result in supplies of perishable fish exceeding demand in local markets. Operators would also require access to supplies of tilapia seed throughout the year to enable stocking in three distinct grow-out periods. Such a strategy is unlikely to be feasible unless facilities including hatcheries and holding ponds are developed, a process that may require support from local institutions. Furthermore, a fundamental question remains as to whether the expected production of 13 t ha^{-1} y^{-1} for tilapia in extensive ponds managed from wastewater aquaculture is feasible. Production from well managed systems such as those in Israel, in which supplementary feed and aeration are also provided, commonly average ~10 t ha^{-1} y^{-1} (Fischer, 1997). The effect of lower than predicted fish yields on the financial returns generated could be assessed through an extension of the sensitivity analysis presented, however, the sensitivity of financial returns to a decrease in fish values of 20%, with a fall in the IRR to 9.8%, suggests that lower productivity could have a significant negative influence on financial viability.

5.4.5. Social and economic issues

Employment represents a tangible benefit associated with wastewater reuse through aquaculture; Kundu (1994) reported that 8,000 workers were engaged in managing and operating ponds managed for wastewater aquaculture in peri-urban Calcutta. The estimated number of employees required to operate the treatment and production system dimensioned using the rational design criteria (~5,550) is significantly lower than the number currently employed. Therefore, should employment be considered an important benefit of lagoon-based wastewater reuse, then the continued operation of the existing system appears to represent the most attractive option. However, Mukherjee (1996) noted that only ~25% of the fishery workers in peri-urban Calcutta are engaged in full-time employment, the remainder being temporary employees. Furthermore, based on interviews Mukherjee (1996) reported that with average wages of Rs1,090 y^{-1} and Rs795 y^{-1} for unskilled permanent workers and temporary labourers, respectively, employees in the existing system found it difficult to cope. Therefore, the annual wage levels included in the scenarios presented here (£410 or Rs28,290 at an exchange rate of £1 to Rs69) would represent a significant improvement. Social benefits commonly attributed to waste reuse practices should be measured against the actual benefit derived by employees.

Access to income generating activities at the peri-urban interface of many towns and cities in developing countries is limited; therefore, jobs associated with operating lagoon-based wastewater treatment and aquaculture production systems may significantly enhance the livelihoods of poor people and marginalised individuals. Increased labour demands associated with managing and maintaining integrated production systems e.g. dike-cropping, may be viewed as a potential benefit, depending on the context within which the development is undertaken. The additional land area (55%) included when developing the base case scenarios could represent an important opportunity for integrating the production of vegetables, fruit trees and livestock. However, similar opportunities may be limited in the existing system as only a relatively small area of land (~6.5%) would be suitable for such production.

Viewing wastewater aquaculture from a social perspective, it is important to note that fish production achieved using the conventional design (11,560 t y^{-1}) is significantly lower than when employing the rational design (45,500 t y^{-1}). Fish production in ponds managed for wastewater aquaculture represents a source of small fish (100-200g) that command a relatively low price (£0.37 kg^{-1}) in the markets of metropolitan Calcutta, and therefore appeal to the poor (Morrice et al., 1998). Furthermore, the perennial supply of low cost fish from these systems may represent an important constituent in the diets of poor households. The estimated level of production based on the rational design approach is significantly higher than the estimated 13,000 t y^{-1} of fish produced in the existing system, and this increased production could contribute significantly to the availability of low cost fish in local markets servicing the poor. Assuming nutritional demands in West Bengal conform to those of typical households in South East Asia where households with 5 members consume on average 100 kg of fish annually (Edwards, Little and Yakupitiyage, 1997), production in the conventional system is sufficient to meet the demand for fish from 115,600 households. However, with the rational design the demand for fish from 455,000 households, a total of 2.28 million individuals, may be met. The population served by the drainage system supplying wastewater to fishponds managed for wastewater aquaculture was estimated at 4 million (Mara et al., 1993). Therefore, it can be estimated that production from the existing system and one developed using the rational design criteria would meet the demand for fish from 0.65 and 2.28 million people, respectively, or 16% and 57% of demand for fish from those served.

Lower faecal coliform numbers in water entering fishponds in the conventional system are largely due to die-off in the maturation ponds. Numbers of faecal coliform bacteria in water flowing to the fishponds in the rational design system (2.2×10^5 100 ml^{-1})

exceed WHO guidelines for the acceptable number of faecal coliforms in wastewater for use in aquaculture (1×10^4 $100ml^{-1}$; WHO, 1989). However, mitigating circumstances were suggested to justify the design criteria used i.e. the dilution and rapid die-off of pathogenic bacteria and viruses (Mara et al., 1993). In certain situations the optimal design criteria may be acceptable i.e. in well-mixed water bodies that are infrequently loaded with wastewater, and Mara et al. (1993) specify in their design criteria that ponds used to culture fish should be between 0.5 and 1 ha. Adopting these design criteria would increase the expected level of mixing, ensuring that the numbers of faecal coliform bacteria in the water column are rapidly attenuated. However, reducing the size of existing fishponds managed for wastewater aquaculture at the Calcutta peri-urban interface from tens of hectares to between 0.5 and 1 hectares, may be impractical.

If an attempt were made to transpose the design criteria proposed by Mara et al. (1993) to the wastewater aquaculture practices at the Calcutta peri-urban interface, the simple wastewater delivery and distribution system may result in pools of relatively concentrated wastewater close to the inlet. The constant introduction of wastewater containing excessive numbers of faecal coliform bacteria to fishponds may result in fish being continuously exposed to levels of faecal coliform bacteria in excess of the WHO guidelines. The problem could be exacerbated if fish congregate around the inlet to forage for food, such as particulate matter form the influent water or benthic organisms that proliferate in sediments receiving high organic inputs.

The concentration of faecal coliform bacteria in water entering fishponds following treatment according to the rational design approach exceeds guideline levels (WHO, 1989) and is a potential cause for concern. However, Pal and Das Gupta (1992) reported that water in fishponds at the Calcutta peri-urban interface contained *E. coli* bacteria at levels of $\sim 1 \times 10^5$ $100\ ml^{-1}$. Comparing this level with the number of faecal coliforms predicted when simulating the rational design approach, it is apparent that adopting such a design would represent an improvement on the current situation. Epidemiological evidence of negative health impacts associated with eating fish cultured in ponds managed for wastewater aquaculture at the Calcutta peri-urban interface is lacking. Local practices of thoroughly cooking fish may explain why consuming fish cultured in water containing numbers of faecal coliform bacteria exceeding the recommended safe levels does not result in elevated levels of illness in consumer groups. However, the potential for contamination with faecal coliform bacteria, other pathogens and persistent chemicals indicates that wastewater reuse systems require monitoring to ensure the safety of products destined for human consumption.

The need for monitoring is further illustrated when it is considered that wastewater aquaculture systems are open to contamination by pathogenic organisms that are more resilient to control measures such as cooking i.e. viruses and helminth eggs. Contamination of fish produced in the reuse system by persistent chemicals e.g. heavy metals and organic compounds, that are not removed by processing and cooking would represent a serious health hazard. Monitoring would need to consider the final product and should aim to ensure that the entire culture system is safeguarded against contamination. Adoption of a HACCP (Hazard Analysis Critical Control Point) framework could represent a significant contribution to safeguarding the quality of products from the existing system, however, the safety of producers must also be considered within whichever framework is adopted. Despite the potential benefits of monitoring, the adoption of such a programme would demand additional resources, require a local agency to take responsibility for implementation and could possibly lead to increased concerns being expressed by consumers.

5.5. Summary

The merits of comparing the performance of traditional waste reuse practices against highly engineered and designed systems may be debatable, especially as such traditional practices have evolved within, and been adapted to, local settings, including social, institutional, political, environmental and cultural facets. However, key indicators developed in this study have identified possible adaptations to current management strategies that have potential benefits for a range of stakeholders including operators, employees and consumers.

Relative merits associated with conventional and rational design approaches for lagoon-based wastewater treatment and aquaculture production have been discussed. However, the two approaches must be considered with respect to site-specific variables, including physical, social, institutional, financial and environmental factors. Outcomes associated with both should be considered with respect to the demands and expectations of potential operators, employees, stakeholders and consumers. Where social benefits associated with waste reuse practices are important, priority should be given to projects that create employment opportunities and supply low cost products to local markets.

Implementing the rational design criteria proposed by Mara et al. (1993) in place of existing wastewater aquaculture practices at the Calcutta peri-urban interface, would require extensive physical redevelopment and modifications to current management strategies. Conversion of the large fishponds that characterise the existing system to more manageable culture ponds of between 0.5 and 1 ha represents a considerable constraint. Furthermore, the area needed to accommodate the system would necessitate the acquisition of extra land.

Developing marginal land or converting existing wetlands to wastewater aquaculture may exclude poor people that previously had access to such areas as common property resources. Should such community resources become contaminated, this would prohibit their use for social purposes such as laundering clothes, bathing and washing utensils; people without access to alternative water sources would be severely affected. Land at the periphery of the existing wastewater aquaculture systems may already be under cultivation. Therefore, before converting already productive land to wastewater aquaculture, it would be prudent to conduct a comprehensive cost-benefit analysis, assessing whether conversion would generate greater net benefits.

Should sufficient land be available to develop the entire optimised system in a new location, this would permit reclamation of the existing system for development. Operators of the existing system could be compensated and relocated to manage the new system, as could employees currently engaged in activities associated with wastewater aquaculture. Although this scenario may be difficult to implement, provision for relocating wastewater reuse systems to locations further from the urban fringe may confer two distinct benefits. Firstly, reclaiming land occupied by the existing system would permit urbanisation to proceed, removing conflicts between developers and stakeholder groups intent on preserving the system. Secondly, revenue generated through the sale of land previously managed for wastewater aquaculture could be used to subsidise development of a new wastewater aquaculture system and relocate poor people displaced by urbanisation. The managed retreat of waste reuse practices from the peri-urban interface, and the transfer of benefits associated with urbanisation to displaced communities must be considered if development at the peri-urban interface is to be equitable.

Chapter Six

Development options: a Delphi investigation

6.1. Introduction

Despite numerous theoretical propositions, exploratory models and pilot-scale studies suggesting the potential technical and financial viability of horizontally integrated aquaculture, there are few operational systems developed in association with commercial aquaculture practices. This suggests that additional constraints may exist, therefore this chapter explores possible constraints and opportunities for horizontal integration identified through consultation with a stakeholder panel. The relative importance ascribed to alternative strategies for minimising negative impacts associated with discharging aquaculture wastewater is also investigated.

The Delphi technique has been developed to elicit expert opinion on a wide range of subjects (Rowe, Wright and Bolger, 1991; Woudenberg, 1991; Gupta and Clarke, 1996). Investigations relevant to aquatic systems include assessments regarding the recreational diver catch of spiny lobsters from waters off Florida (Zuboy, 1981), habitat impairments that impact on aquatic resources in lake Ontario (Busch and Lary, 1996), water resource planning in the Grand River basin, Ontario with respect to climate change (de Loe, 1995), the impacts of an improper effluent control system in Jeddah, Saudi Arabia (Mohorjy and Aburizaiza, 1997), the selection of treatment and discharge options for domestic wastewater in the Great Lakes basin (Ludlow, 1975) and the development of sustainability indicators for aquaculture (Caffey, 1998; Caffey and Kazmierczak, 1998).

Delphi investigations capitalise on the knowledge, analytical ability and predictive powers of experts in a specific subject area to produce consensus about possible future events. Unlike group discussion methods for reaching consensus e.g. committee meetings, the Delphi technique is conducted through an iterative series of questionnaires sent to individual participants or panelists. Caffey (1998) noted that the Delphi technique is founded on four key assumptions:

- expert opinion is a valid input to inexact research areas
- consensus amongst experts is more valid than the opinion of an individual
- joint meetings of experts induce a follow-the-leader bias
- ensuring the anonymity of participants compensates for inherent opinion biases.

The Delphi methodology was therefore chosen as a means of eliciting a more complete range of possible constraints and opportunities than the financial assessment developed in Chapters 3 and 4. Employing this approach it was also possible to reach a consensus amongst a stakeholder panel concerning the relative importance of the proposed factors.

The commitment of participants in responding to the iterative rounds of this Delphi investigation is appreciated greatly and correspondence with Dr Caffey contributed significantly to conducting the study.

6.2. Method

This study aimed to elicit responses from an expert stakeholder panel to identify factors having the greatest potential to influence decision-making in adopting horizontally integrated aquaculture. The panel proposed a range of factors, and then the relative importance of each was investigated and weighted. The importance of alternative strategies for limiting negative impacts associated with the discharge of aquaculture wastewater was also assessed.

6.2.1. Research questions and hypotheses

Research questions were formulated to guide the Delphi investigation:

- given the potential for horizontal integration in commercial aquaculture, which factor(s) are identified by a panel of experts as possible constraints and opportunities?
- which alternative management strategies do the panel believe could limit negative impacts associated with aquaculture wastewater?

Hypotheses were formulated to assist in analysing the generated data. It was postulated that participants would reach a consensus concerning the priority of various factors in influencing decision-making. It was further proposed that, due to the commercial nature of the aquaculture systems being considered, economic factors would represent the most important constraints, whilst due to growing concern over the negative impacts of discharging aquaculture wastewater, environmental attributes would present the greatest opportunities. Finally, it was expected that improved treatment technologies would represent the most promising alternative strategy for reducing negative impacts of wastewater.

6.2.2. Questionnaire formulation

The first-round questionnaire was written specifically for this Delphi investigation (Appendix 2a), and questions formulated to address the aims of the investigation. Prior to distributing the questionnaire a small number of aquaculture researchers were asked to provide feedback on the content and presentation. This demonstrated that asking for general constraints and opportunities elicited responses such as "environmental" and "financial". Therefore, six sub-category headings i.e. physical, environmental, managerial, institutional, economic and social were incorporated into the questionnaire to elicit responses that were more specific. These sub-category headings were invoked as they were found adequate to describe the responses from participants in the pilot-study. With respect to alternative strategies for minimising the impact of aquaculture wastewater, four sub-categories were adequate to describe the responses: institutional, managerial, socio-economic and technological.

6.2.3. Participant selection and instruction

The first-round questionnaire and the specific research aims associated with this Delphi investigation were presented to prospective panel members, allowing them to assess their own competence to answer the study questions. Furthermore, a statement outlining the aims and likely outputs of the study allowed potential panel members to judge possible benefits they may receive from participation. Stakeholders were approached using two strategies:

- individuals who had published work in the past ten years relating to horizontally integrated aquaculture were invited to participate in the study
- the first-round questionnaire was distributed to several electronic discussion groups; members who considered themselves sufficiently expert to engage in the process were asked to respond.

Although the statistical tests selected to assess the outcomes of this Delphi investigation require a minimum of three paired samples i.e. three panel members, previous studies employing this approach have commonly elicited the opinion of panels with 15-60 members (Hasson, Keeney and McKenna, 2000). It was anticipated that a similar number of participants would respond to this study.

To avoid leader bias, the identity of those responding was concealed from other panel members. The flow-chart presented in Figure 6.1 outlines the major steps in conducting the investigation; three rounds were found necessary to elicit consensus amongst the participants.

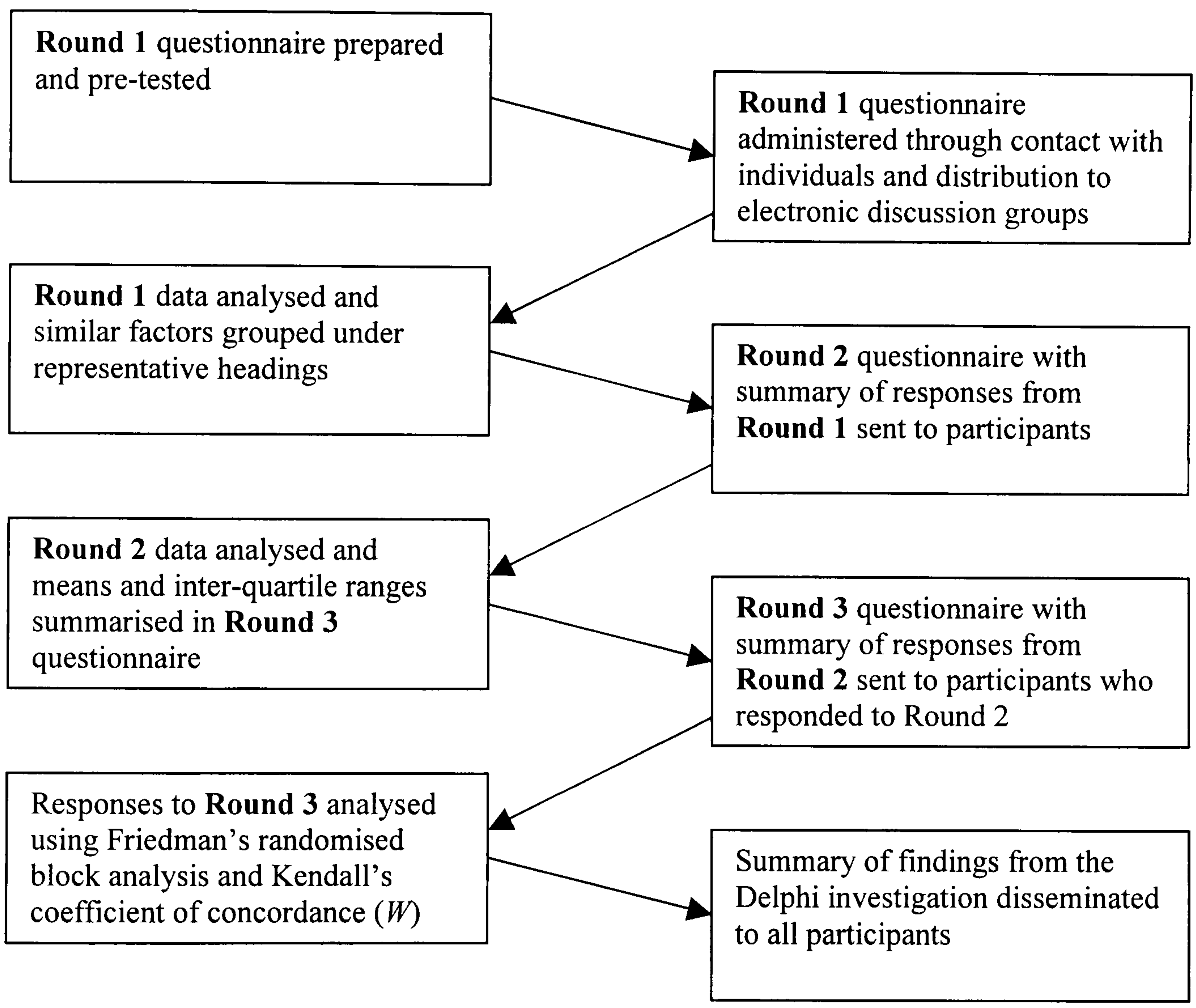

Figure 6.1. Flow-chart for data collection and feedback to participants during the iterative rounds of the Delphi investigation.

6.2.4. Round 1

Within the first-round questionnaire, panel members were presented with a brief explanation of the term *horizontally integrated aquaculture*, thus:

'the culture of aquatic species in wastewater from commercial aquaculture, reducing the concentration of pollutants and potentially conferring benefits to the operator, environment and stakeholder groups.'

Examples were presented, including the culture of seaweed and shellfish in wastewater from shrimp ponds and marine cage facilities, and the use of constructed wetlands planted with reeds or mangrove to treat wastewater from land-based freshwater and marine or brackish aquaculture, respectively. The purpose of providing a definition and examples was to focus the expertise of individual panel members on issues specifically relating to such systems.

When answering the questionnaire, panel members were required to consider factors they believed represented either *constraints* or *opportunities* and to list these factors under the six sub-category headings proposed following the pilot-study. The second section of the questionnaire requested panel members to suggest *alternative strategies* for limiting negative impacts associated with discharging aquaculture wastewater.

Once questionnaires were completed and returned, factors suggested as constraints, opportunities and alternative strategies during the first-round were grouped under representative headings; where possible wording used by participants was employed. In previous studies factors suggested by less than a specified proportion of the panel members, for example 20% of each stakeholder group (Caffey and Kazmierczak, 1998) were discarded. However, in this study no responses elicited during the first round were rejected. This approach was adopted to avoid biasing the study towards majority held beliefs and conceptions, and permitted individual participants to express unique points of view for evaluation by the other members of the stakeholder panel during subsequent rounds.

6.2.5. Round 2

The second-round questionnaire (presented in Appendix 2b), summarizing factors suggested by participants during the first-round, was sent to all respondents. Factors were listed under the respective category headings with the frequency with which they occurred. Participants were asked to assign a score between 1 and 10 for the importance ascribed to each factor, 1 being least and 10 being most important. Responses for the second round were collated and mean values and inter-quartile ranges for factor weights were calculated.

6.2.6. Round 3

Second-round responses were summarized in tables and sent to participants that replied during round 2 (Appendix 2c). Mean weights and inter-quartile ranges were given for the factors, together with their original narrative description. Participants were required to either accept the mean score or suggest an alternative score based on their own perception. Where participants wished to assign a weight outside the inter-quartile range, they were requested to provide a statement justifying their decision.

6.3. Analysis

Following the rationale presented by Caffey (1998), Friedman's randomized block analysis was employed to investigate the distribution of weights assigned to factors by participants in rounds 2 and 3. This nonparametric test is appropriate for identifying the presence of rank patterns in survey data generated by Delphi investigations where the non-random nature of participant selection excludes the use of parametric statistical tests. Friedman's null hypothesis (H_0) states that each ranking of random variables within a block is equally likely. However, the nature of the Friedman's test statistic means that the null hypothesis tends to be rejected in the presence of slight rank correlation, and the test provides no indication of the degree of agreement between the ranks assigned by participants (Caffey, 1998). Confidence in the degree of agreement within identified rank patterns was therefore elicited using Kendall's coefficient of concordance (*W*). This measure of rank convergence, or agreement, ranging from 0 to 1, has been recommended for interpreting data generated by Delphi investigations to provide information on the degree of consensus achieved and the level of confidence with which the mean ordinal ranks may be considered (Schmidt, 1997). Guidelines for interpreting the degree of agreement and confidence in ranks associated with values for Kendall's *W* are given in Table 6.1.

Table 6.1. Guidelines for interpreting the degree of agreement and confidence in ranks associated with discreet values for Kendall's coefficient of concordance (*W*).

Kendall's *W*	Interpretation	Confidence in ranks
0.1	Very weak agreement	None
0.3	Weak agreement	Low
0.5	Moderate agreement	Fair
0.7	Strong agreement	High
0.9	Unusually strong agreement	Very high

Adapted from Schmidt (1997)

Although it provides an indication of the degree of convergence observed in Delphi survey data, and the level of confidence with which ranks may be regarded, Kendall's *W* fails to provide any indication of the relative importance that participants ascribe to individual factors. Caffey (1998) employed a distance function approach to assess the actual consensus order of ranked preferences and indicators. However, his comparison with mean ranks calculated from cardinal weights of individual indicators showed only slight differences. In the light of this it was considered acceptable to use mean ranks to describe the relative importance attached to individual factors by participants.

6.4. Results

Results show the number of participants contributing to each round, the nature and distribution of constraints, opportunities and alternative strategies submitted in round 1, the mean weights assigned to factors following round 3 and the associated mean ranks of separate constraints, opportunities and alternative strategies. Patterns in rank distribution are identified and the degree of convergence in rank patterns following rounds 2 and 3 evaluated and discussed.

6.4.1. Survey participation

Due to the panel selection methods employed it is not known how many potential respondents viewed the initial questionnaire. However, approximately 40 individuals were contacted directly and a total of 24 individuals responded to the first-round questionnaire. During the second round 19 replies were elicited from those that had participated in the first round, a response rate of 79%; during the third round the response rate remained stable at 79%, with replies received from 15 participants. Of the initial respondents, 71% were researchers, 13% managed production facilities, 8% worked for regulatory bodies and 8% were consultants. In the geographical distribution, 7 were from the United States, 6 from Scotland, 2 from Brazil, with single representatives from Australia, Canada, Chile, Korea, Malaysia, New Zealand, Nicaragua, Singapore and Taiwan.

6.4.2. Constraints

First round participants responded with 84 short statements outlining possible constraining factors. A number of these were sufficiently similar to be aggregated into 29 distinct factors. Table 6.2 summarises the range of constraints suggested; eleven were mentioned by only 1 participant while one, "the availability of suitable land or water", was suggested

Table 6.2: Constraints participants associate with horizontally integrated aquaculture, the frequency of occurrence in round 1 (n) and both mean score (x) and mean ordinal rank following round 3.

Systems feature	n	x	Rank
Physical			
Facility designed primarily for original aquaculture species	1	4.7	28
Availability of suitable land or water	11	7.4	2
Wastewater supply e.g. nutrient flows not optimal or presence of chemicals, antibiotics or pathogens	4	7.1	6
Engineering of wastewater flows difficult e.g. high volumes and distance	2	5.4	22
Biological treatment inconsistent at removing nutrients throughout the growing season	1	5.8	16
Environmental			
Site specific design required e.g. climate vulnerability	4	5.7	14.5
Discharge requirements; by-products from integrated systems may not comply with regulations	2	6.5	7
Translocation of organisms suitable for integrated production restricted	1	4.3	29
External pollution could affect the integrated system	1	5.4	20
Managerial			
Lack of skills and training for existing managers and workers	4	6.0	14
Managers lack leadership, resist change and adopt a non-systematic approach	5	6.4	9
Decisions based on short-term financial appraisal	1	7.2	3
Limited knowledge-base regarding design and management e.g. optimal loading rates, harvesting strategies, disease management	6	7.0	5
Inadequate supplies of seed and inputs for integrated system	1	4.9	26
Institutional			
Absence of instruction and training support	1	5.9	18
Inertia i.e. a lack of research and development	1	6.0	11
Constrained by existing paradigms of food production i.e. no structures exist to facilitate systems thinking	2	5.8	17
Environmental laws and planning restrictions constrain integration	3	5.8	14.5
Lack of structures for monitoring and auditing integrated systems	1	5.6	19
Economic			
Market analysis and stimulation required	3	6.5	8
Limited revenue generated by the integrated system	4	5.0	25
Insufficient techniques for analysis of operating costs and accounting for broader issues such as opportunity costs	2	4.9	27
Holders of economic power not interested	1	5.3	21
Financial costs associated with development	8	7.3	1
Lack of funding, access to venture capital	2	6.9	4
Costs associated with management and operation	3	5.9	10
Social			
Public acceptance of produce, the perception of 'dirty' food	4	5.8	13
Conflicts with stakeholder groups e.g. user groups or environmentalists	4	5.3	23
Limited systems thinking education and information exchange	1	5.2	24

by 11 participants. Only 3 factors were submitted as possible social limitations, 4 factors identified as environmental constraints, 5 factors each as physical, managerial and institutional constraints and 7 submitted as economic constraints.

Results presented in Table 6.2 showed a wide variation in mean weightings following round 3, ranging from 4.3 for the "restricted translocation of organisms suitable for integrated production" to 7.4 for "the availability of suitable land or water". Lower mean weights represented lower ascribed importance. Mean ordinal ranks for each constraint are also presented in Table 6.2. These demonstrate that the 4 most important constraints were: "financial costs associated with development", "the availability of suitable land or water", "decisions based on short-term financial appraisal" and "lack of funding and access to venture capital".

6.4.3. Opportunities

Participants responded with 94 short statements concerning potential opportunities; as with constraining factors, a number were sufficiently similar to be aggregated, resulting in 27 distinct factors. Table 6.3 summarises the range of opportunities suggested; eight were mentioned by only 1 participant while one, "increased income generated from additional crops based on same inputs", was suggested by 12 participants. Only 3 factors were submitted as potential physical benefits; 4 factors identified as environmental opportunities and 5 factors each proposed as managerial, institutional, economic and social opportunities.

Mean weightings assigned to possible opportunities after round 3 ranged from 3.7 for "better transport around remote areas" to 8.5 for both "improved efficiency of resource use e.g. nutrients and water" in the physical category and "reduced impact on the environment and downstream users e.g. other farms" in the environmental category. These both received a mean rank of 1.5, indicating their equal importance to participants. Mean ranks presented in Table 6.3 show that the third and fourth most important opportunities, both with a mean weighting of 8.3, were "reduced sediment and nutrient concentration in wastewater" and "less energy consumed and waste generated in food production"; both factors occurred in the environmental category.

Table 6.3: Opportunities participants associate with horizontally integrated aquaculture, the frequency of occurrence in round 1 (n) and both mean score (x) and mean ordinal rank following round 3.

Systems feature	n	x	Rank
Physical			
Better transport around remote areas	1	3.7	27
Improved efficiency of resource use e.g. nutrients and water	7	8.5	1.5
More attractive in the landscape than conventional treatment systems	2	6.4	16
Environmental			
Reduced sediment and nutrient concentration in the wastewater	8	8.3	3
Less energy consumed and waste generated in food production	1	8.3	4
Increased habitat diversity, providing shelter to endemic species	3	6.5	17
Reduced impact on the environment and downstream users e.g. other farms	8	8.5	1.5
Managerial			
Meet expectations of managers regarding environmental protection and waste management	2	7.0	8
Increase opportunities for recirculation	1	7.0	11
Facilitate the development of a new paradigm for aquaculture and other agricultural sectors	2	6.6	13
Contribute to the skill base of managers	1	5.6	24
Reduce the potential for farms to self-pollute	2	7.2	6
Institutional			
Improved discharge standards that reduce penalties e.g. court action or closure	2	6.1	23
New opportunities to develop commercial partnerships	2	6.2	21
Potential site for research and development	2	6.5	18
Enabling more stringent discharge standards to be satisfied in the future	6	6.8	12
Help meet the standards for quality assurance or organic certification	1	7.2	10
Economic			
Reduce wastewater treatment costs	1	7.0	9
Increased income generated from additional crops based on same inputs	12	7.2	7
Represents a diversification reducing economic risks	5	6.5	14
Reduce the level of any potential pollution tax	6	6.1	22
Contribute to the regional economy, adding to tax base	2	5.2	25
Social			
Improved public perception; reconciling environmental and economic goals of different groups	6	7.6	5
Represents a good educational resource	1	6.3	20
Increased employment opportunities	5	6.5	15
Enhanced appeal of the primary aquaculture product to consumers	5	6.4	19
Better places to live	1	5.2	26

With the exception of institutional opportunities, each of the 5 remaining categories has a factor with a mean ranking equal to or above 7; the most important institutional factor with a mean rank of 10 was associated with "helping meet the standards for quality assurance or organic certification".

6.4.4. Alternative strategies for limiting impacts of aquaculture wastewater

First round participants responded with 51 short statements describing alternative strategies for limiting negative impacts associated with aquaculture wastewater. These factors were aggregated under 18 distinct headings and are summarized in Table 6.4. Five were mentioned by single panel members, whilst two proposed in the technological category "increased research and development into improved treatment technologies" and "develop systems for water reuse" were mentioned by 7 and 6 participants, respectively. The distribution of responses amongst the four strategy types showed that 4 factors were submitted in both the managerial and technological categories whilst 5 were submitted in both the institutional and socioeconomic categories.

Mean weightings assigned to the alternative strategies after round 3 ranged from 3.8 to 8.1. The lowest mean weight was associated with the "adoption of extensive as opposed to intensive management practices", the highest with two factors submitted in the management category: "management procedures that improve water quality e.g. careful feed management, de-sludging lagoons, aeration and harvesting strategies that minimise discharges" and "good planning prior to developing aquaculture facilities and improved site selection". The two management factors with mean weights of 8.1 received a mean rank of 1.5, indicating their equal importance to participants. The most important factors in the institutional, socioeconomic and technological categories received mean ranks of 3, 8 and 4, respectively, and were: "encourage collaboration between researchers and commercial enterprises"; "improved evaluation of benefits associated with management practices that reduce environmental impact" and "increased research and development into improved treatment technologies".

Table 6.4: Strategies proposed by participants to reduce negative impacts associated with aquaculture wastewater, the frequency of occurrence in round 1 (n) and both mean score (x) and mean ordinal rank following round 3.

Factor	n	x	Rank
Institutional			
Better education of farmers regarding water quality and environmental management	3	7.1	6
Increase and enforce discharge standards for wastewater or implement a pollution tax	3	6.9	10
Open some commercial operations for public tours	1	6.1	15
Encourage collaboration between researchers and commercial enterprises	3	7.9	3
Provide information and direction to government regarding opportunities for the innovative management of aquaculture wastewater	2	7.1	7
Managerial			
Adoption of a more holistic/systematic paradigm for aquaculture	2	6.7	12
Good planning prior to developing aquaculture facilities, improved site selection	3	8.1	1.5
Management procedures that improve water quality e.g. careful feed management, de-sludging lagoons, aeration and harvesting strategies that minimise discharges	5	8.1	1.5
Adopt extensive as opposed to intensive management practices	1	3.8	18
Socioeconomic			
Educate the public and managers regarding recycling systems	1	6.5	14
Look at the energy costs of typical intensive production systems	1	6.6	11
Government funding e.g. subsidies, grants and tax relief to encourage research and development	2	6.1	16
Improved evaluation of benefits associated with management practices that reduce environmental impact	2	7.0	8
The need to portray a positive image will necessitate improved waste management	1	6.9	9
Technological			
Increase research and development into improved treatment technologies	7	7.5	4
Develop systems for water reuse	6	6.6	13
Improved feed quality e.g. lower feed conversion ratios	5	7.3	5
Develop new vaccines and improve disease control	3	5.2	17

6.4.5. Confidence and convergence in rank patterns after round 2

Application of Friedman's test in assessing the distribution of factor weightings assigned by each participant identified similar rank patterns for constraints, opportunities and alternative strategies following round 2 of the investigation ($p<0.000$, two-tailed) (Table

6.5). However, despite the presence of rank patterns, Kendall's W calculated for ranks assigned to factors within each research area i.e. constraints, opportunities and alternative strategies at 0.143, 0.267 and 0.205, respectively, indicated that agreement in assigned ranks ranged from "very weak" to "weak" and that the confidence of defining rank patterns ranged from "none" to "low"; Table 6.5 presents a summary of results.

Table 6.5: Values for Friedman's X^2_F at probability levels (p) indicated and Kendall's W for weights assigned to factors during round 2.

Research area	Friedman's X^2_F	(P)	Kendall's W	Agreement	Confidence
Constraints	59.9	< 0.000	0.143	Very weak	None
Opportunities	111.2	< 0.000	0.267	Weak	Low
Alternatives	59.2	< 0.000	0.205	Weak-Very weak	Low-None

6.4.6. Confidence and convergence in rank patterns after round 3

Friedman's test on the distribution of factor weights following round 3 indicated rank patterns in all three areas of investigation (p<0.000, two-tailed). Application of Kendall's W indicated that the level of agreement in rank patterns was significantly higher than in round 2, and that the level of confidence in ranking had also increased (Table 6.6). A value for Kendall's W of 0.673 for ranking constraints, indicates "strong" agreement and a "high" level of confidence; a value of 0.837 for ranking opportunities indicates a "very strong" level of agreement and implies a "high" to "unusually high" level of confidence; a value of 0.733 for alternative strategies again implies a "strong" agreement amongst participants and a "high" level of confidence.

6.4.7. Rank patterns and consensus within categories

More detailed analyses of rank patterns in each of the sub-categories in the three research areas employing Friedman's test and Kendall's W are presented here. Friedman's test indicates the presence of similar rank patterns for responses from individual participants in all categories investigated ($p<0.000$, two-tailed).

Table 6.6: Values for Friedman's X^2_F at probability levels (p) indicated and Kendall's W for weights assigned to factors during round 3.

Research area	Friedman's X^2_F	(P)	Kendall's W	Agreement	Confidence
Constraints	263.7	< 0.000	0.673	Strong	High
Opportunities	304.6	< 0.000	0.837	Unusually strong-Strong	Very high-High
Alternatives	174.4	< 0.000	0.733	Strong	High

Constraints

Rank patterns observed in the physical and environmental categories result in values for Kendall's W of 0.87 and 0.884, respectively, indicating an "unusually strong" level of agreement and "very high" level of confidence (Table 6.7). Agreement in rank patterns observed in weights assigned to factors submitted as managerial and economic factors is "strong" and confidence is "high". Values for Kendall's W of 0.442 and 0.545 for rank patterns observed in institutional and social categories, respectively, indicate "moderate" agreement and that confidence is "fair".

Opportunities

Values for Kendall's W of 1 and 0.878 for rank patterns observed in physical and environmental categories, respectively, indicate an "unusually strong" level of agreement and that confidence in these findings is "very high" (Table 6.8). Rank patterns for weight assigned to factors in the managerial category result in a value for Kendall's W of 0.774 indicating a "strong" to "unusually strong" level of agreement and a "high" to "very high" level of confidence; Kendall's W for rank patterns relating to social factors is similar at 0.802 indicating an "unusually strong" to "strong" level of agreement and a "very high" to "high" level of confidence. Agreement in rank patterns observed in weights assigned to factors submitted as institutional and economic opportunities is "strong" and confidence is "high".

Table 6.7: Friedman's X^2_F at probability levels (p) indicated and Kendall's W for rank patterns in weights assigned to constraints following round 3.

Category for constraints	Friedman's X^2_F	(p)	Kendall's W	Agreement	Confidence
Physical	48.7	<0.000	0.87	Unusually strong	Very high
Environmental	37.1	<0.000	0.884	Unusually strong	Very high
Managerial	35.9	<0.000	0.641	Strong	High
Institutional	24.8	<0.000	0.442	Moderate	Fair
Economic	57.4	<0.000	0.684	Strong	High
Social	15.3	<0.000	0.545	Moderate	Fair

Alternative strategies

Rank patterns observed in the institutional and managerial categories for alternative management strategies result in values for Kendall's W of 0.697 and 0.765, respectively, indicating a "strong" agreement amongst participants and a "high" level of confidence (Table 6.9). Kendall's W for rank patterns observed in the technological category is slightly higher than that for the managerial category at 0.786, indicating a "strong" to "unusually strong" agreement amongst participants and signifying "high" to "very high" confidence. A value for Kendall's W of 0.579 for rank patterns in the socioeconomic category indicates that agreement between participants is moderate to "strong" and that confidence is "fair" to "high".

Table 6.8: Friedman's X^2_F at probability levels (p) indicated and Kendall's W for rank patterns in weights assigned to opportunities following round 3.

Category for opportunities	Friedman's X^2_F	(p)	Kendall's W	Agreement	Confidence
Physical	28	<0.000	1	Unusually strong	Very high
Environmental	36.9	<0.000	0.878	Unusually strong	Very high
Managerial	43.3	<0.000	0.774	Strong-Unusually strong	High-Very high
Institutional	38.8	<0.000	0.692	Strong	High
Economic	41.9	<0.000	0.749	Strong	High
Social	44.9	<0.000	0.802	Unusually strong-Strong	Very high-High

Table 6.9: Friedman's X^2_F at probability levels (p) indicated and Kendall's W for rank patterns in weights assigned to alternative strategies following round 3.

Category for alternative strategies	Friedman's X^2_F	(p)	Kendall's W	Agreement	Confidence
Institutional	39.1	<0.000	0.697	Strong	High
Managerial	32.1	<0.000	0.765	Strong	High
Socioeconomic	32.4	<0.000	0.579	Moderate-Strong	Fair-High
Technological	33	<0.000	0.786	Strong-Unusually strong	High-Very high

6.5. Discussion

An important aspect of the Delphi methodology is the contribution of participants to selecting factors for investigation. Open questions employed during the first round of enquiry provided respondents with an opportunity to suggest factors based on their unique understanding of the situation and their personal agendas. This may be in contrast to group discussions or workshops where opinionated individuals may have a forum to impose their ideas on others or the views and feelings of less vocal groups are ignored. In this study, where all factors proposed in the first round were put forward into the second round, all perspectives were given equal consideration. There are two distinct advantages over other consensus building strategies, firstly, permitting participants to select areas for investigation helps engender ownership of the process and secondly, requesting a variety of stakeholders to contribute results in a multidisciplinary assessment giving greater scope and depth than studies engaging only with sectoral experts. The wide range of factors submitted by participants in the first round confirms the advantage over employing a predefined questionnaire.

Previous studies have emphasised the importance of selecting a panel of experts for Delphi investigations. However, as this study dealt with an innovative management strategy it was difficult to identify sufficient experts in the field. It was therefore decided to try and identify stakeholders with experience or expertise in policy making, regulation, environmental protection, commercial aquaculture production, research and development. This widened the debate, elicited a more diverse range of factors for consideration and tested the Delphi methodology in generating consensus amongst diverse stakeholder groups. Hasson et al. (2000) discuss the validity of the term "expert" as applied to participants in Delphi investigations and instead suggest using "a panel of informed individuals", a term used by McKenna (1994) to describe participants in a Delphi investigation regarding nursing practices. In addition to obtaining a broader perspective of the subject matter through engaging stakeholders as opposed to experts, it should also be noted that stakeholders who may not consider themselves experts can nevertheless be highly influential, or have agendas in which aquaculture is not of prime importance, but which may constrain or enhance prospects for aquaculture development.

Considering the diverse backgrounds and specialisations of participants in this study, prospects for reaching a consensus may have been considered poor. However, following the third round, rank patterns had emerged in each of the three research areas with a level of agreement ranging from "strong" to "unusually strong", and confidence in this finding considered "high" to "very high".

At the start of the study it was hypothesised that economic factors would constitute the most important constraints to development; that the greatest opportunities would relate to environmental concerns and that improved treatment technologies would represent the most promising strategy for reducing negative impacts associated with aquaculture wastewater. Mean rankings following round 3 indicated that the most important constraint was indeed submitted in the economic category. Assigning the greatest importance to the "financial costs associated with development" may have been expected considering the focus on commercial production systems in this study. However, when considering potential opportunities the prospect of "increased income generated from additional crops based on same inputs", which could offset the costs associated with development, was only considered the 7th most important factor. This suggests perhaps that the participants are pessimistic concerning the likely financial returns from horizontal integration or that opportunities such as the "improved efficiency of resource use…", "reduced impact on the environment and downstream users…", "reduced sediment and nutrient concentrations in wastewater" and "less energy consumed and waste generated…", although not generating a direct income, are of more value than the financial returns from integration. However, the question remains whether commercial producers are in a position to capture these benefits, especially when opportunities such as "help[ing] meet the standards of quality assurance or organic certification" and "enhanced appeal of the primary aquaculture product to consumers" were ranked 10th and 19th most important, respectively.

The second most important constraint, "the availability of land or water" appears to be a decisive limiting factor, although presumably this would depend on the setting of the primary culture facility. However, it may be possible to tailor the design, type and management of the secondary culture facility to permit effective horizontal integration. The 3rd and 4th most important constraints "decisions based on short term financial appraisal" and a "lack of funding, access to venture capital" may reflect a perception that horizontal integration is costly, whilst generating low returns. This may make financial assessments based on conventional approaches unfavourable, dissuading prospective financial backers. Other constraints, e.g. "environmental laws and planning restrictions constrain integration" or the negative "public acceptance of produce, the perception of 'dirty' food", although not discussed in depth, may demand detailed consideration as in particular locations they could represent insurmountable obstacles to horizontal integration.

Although one of the two most important opportunities, a "reduced impact on the environment and downstream users e.g. other farms" occurred in the environmental section, the second, "improved efficiency of resource use e.g. nutrients and water" was proposed as a physical benefit, so only partially satisfying the initial hypothesis. Limiting the environmental impact of aquaculture is undoubtedly desirable and avoiding conflicts with downstream users could represent a tangible benefit to operators. However, the importance of horizontal integration in achieving "improved discharge standards that reduce penalties e.g. court action or closure" and to "reduce the level of any potential pollution tax" with ranks of 23 and 22, respectively, suggests that environmental protection and altruistic motives are more influential than possible future cost-savings. Potentially influential and beneficial opportunities from the perspective of a production system manager are "improved public perception..." and "reduced potential for farms to self-pollute", ranked 5th and 6th most important. Enhanced public perception may add value to products from the primary aquaculture facility, although potential benefits from quality assurance and organic certification were given less importance. Avoiding the problems of self-pollution outlined in Chapter 1 could have a direct and positive impact on both production and financial returns.

It may be argued that the 3rd and 4th most important opportunities, "reduced sediment and nutrient concentrations in wastewater" and "less energy consumed and waste generated..." could have been brought together following the first round. However, interpreting these statements as alluding to waste reduction and waste avoidance, respectively, suggests that participants considered these two fundamentally different approaches to waste management to be equally important, which has significant implications for production system management.

The hypothesis that improved wastewater treatment technologies would represent the most promising alternative approach was rejected, the greatest importance being associated with management factors. "Good planning prior to developing aquaculture facilities..." and the adoption of "management procedures that improve water quality..." were regarded as the most important strategies. Effective planning and appropriate site selection for aquaculture developments is widely regarded as a prerequisite for financial and ecological sustainability, and it has been demonstrated that refined management practices can represent an efficient and cost effective approach to waste reduction and improved resource-use efficiency. However, as access to suitable sites for aquaculture declines and the returns from improved management strategies begin to diminish, the prospects for improved wastewater management may improve. The perception of horizontal integration may therefore be modified by future trends, which suggests the need for a continued reassessment of its potential. Furthermore, it has been shown that for both constraints and opportunities for horizontal integration, the most important 3-4 factors were distributed in more than one category, reconfirming the need to consider the development of innovative management strategies and technological development employing a systems approach.

This investigation demonstrates that the practical application of outputs from the modelling scenarios in the preceding chapters may demand greater attention to other factors not explicitly stated or evaluated. Thus, possible constraints such as the "availability of land or water" or potential opportunities e.g. "reduced impact on the environment and downstream users e.g. other farms" may need to be more explicitly evaluated. To assess the full range of possible limitations and potential benefits, researchers and commercial managers may need to develop case studies or site specific assessments involving a range of factors, perhaps suggested by a stakeholder panel with a particular interest in the sector or proposed location e.g. local planners or environmental groups. Engaging local stakeholders may elicit factors that were only ranked as moderately important here but may require greater attention in a particular setting e.g. "environmental laws and planning restrictions constrain integration" or "attractiveness in landscape compared to conventional treatment systems". The need to consider agriculture and aquaculture development within a systems perspective has been recognised (Spedding, 1988; Muir, 1996), and Delphi investigations conducted with local stakeholder groups may represent a promising approach to achieving this objective.

Although the range of factors considered in this study is diverse, the ADEPT model has the capacity to enable a number of the most important constraints identified to be investigated e.g. costs associated with development, the area of land or water required, the effect of varying the time horizon for financial appraisal and optimal design strategies and management regimes. Furthermore, some of the most important opportunities associated with horizontally integrated aquaculture may be investigated e.g. exploring the efficiency of resource use in the integrated culture system and assessing changes in the concentration of solids and nutrients in treated wastewater. Therefore, despite the limitations described above, ADEPT model scenarios could be a valuable resource from which more comprehensive situation analyses could be developed. Modelling outputs should be used to identify high potential systems and focus resources on developing more comprehensive case studies that address the needs, priorities and expectations of operators and stakeholders.

Although good agreement was identified in rank patterns observed in all three research areas and the majority of categories within these, it was notable that the "moderate" agreement and "fair" confidence regarding constraints from institutional and social factors was less defined than in the other categories. This may reflect differing social settings and institutional arrangements encountered by participants from different countries, and it could be argued that a more focused study would be required to consider these categories more fully in constraining prospects of horizontally integrated aquaculture. Areas of ambiguity identified during a Delphi process are likely to require further investigation; possible tools for achieving a better understanding of the social setting may include focus groups, consumer acceptance and market surveys or the development of a more refined Delphi investigation. Institutional arrangements may be investigated further through reviewing appropriate policy and regulatory documents and undertaking an institutional analysis.

The preparation and facilitation of a stakeholder Delphi investigation represents a resource demanding exercise that is unlikely to be feasible in the majority of cases where small-scale innovations are being proposed on individual farms. However, where issues with potentially wide-reaching impacts are being addressed, such as the formulation of policy or regulation or the preparation of an Environmental Impact Assessment for a large-scale development, the Delphi method may provide a suitable approach to encourage a constructive dialogue between diverse stakeholder groups.

The initial focus of this Delphi investigation, both with respect to the area of research and selection of participants, has useful implications for interpreting the results. It could be argued that selecting participants from a particular geographical region or having experience with a certain culture system may have improved the focus of this study. The limited extent of knowledge and experience regarding horizontally integrated aquaculture necessitated the inclusion of participants from a wide range of countries and experience with a diverse array of culture systems. However, it was still possible to reach a general consensus regarding the importance of factors proposed by all participants. As knowledge and experience of these systems develops it may be possible to undertake more focused studies, but this investigation has provided a valuable insight into the nature and diversity of factors that may influence decision-making.

As investigations become more focused, the demands and expectations of participants are also likely to become more personal and value-laden. A criticism of other traditional methods for consensus building, which the Delphi methodology is designed to avoid, is the fact that open debates regarding emotive issues may become distorted by the opinions of participants with markedly different agendas. Investigations involving stakeholders with differing agendas may cause concerns and grievances to become explicit, leading to conflict. The Delphi technique aims to avoid this issue as the quasi-anonymity of participants is ensured (McKenna, 1994) and emotive language is tempered through the aggregation of statements submitted in the first round, although, value-laden statements are not excluded. Eliminating conflict using the Delphi technique and avoiding confounding influences such as leader-bias helps develop a constructive dialogue amongst participants with differing perspectives and assists in formulating mutually acceptable outcomes.

A potential constraint to invoking the Delphi methodology is the problem of attrition; participants contributing value-laden statements and weights in the initial rounds may decide not to contribute in subsequent rounds. To avoid this problem it is recommended to engage sufficient participants to reduce the influence of any one individual; efforts should also be made to ensure that as few participants retire as possible, possibly meaning extra correspondence and time allocated to following up queries and unreturned questionnaires. Sumsion (1998) suggests that a response rate of 70% is desirable in each round to ensure the rigour of the investigation; this criterion was met in this investigation.

A key factor governing response rates in Delphi investigations is the commitment of participants, which has been related to their interest and involvement with the question under investigation (Hasson et al., 2000). As mentioned previously, the purposive sampling methodology that permitted panel members to submit factors ensured that issues of interest to individual participants were evaluated. Furthermore, making the objectives of the study explicit to potential participants in the introductory text that accompanied the first-round questionnaire ensured that only those with an interest in the outcome of the research would respond. The success of this approach is shown by response rates greater than 70% in both the second and third rounds. Having demonstrated a commitment to the research being undertaken, it is hoped that participants that contributed to this study gained an insight to the perceptions of others regarding horizontally integrated aquaculture, and that agreement on possible constraints and potential opportunities has provided an indication of the most significant factors that require further consideration, or hold the greatest promise for realising the benefits associated with horizontally integrated aquaculture.

Findings from this study indicate that rank patterns were identifiable following round two. However, agreement and confidence in the distribution of these ranks was only regarded as reliable after round 3. Despite the diverse backgrounds and experiences of participants it was possible to reach consensus on the nature and importance of key constraints and opportunities. The findings of this research also indicate where the priorities for subsequent research and development lie, and may contribute to focusing resources on formulating design and management plans that meet the demands and expectations of the diverse stakeholder groups that exert an influence on the decision-making process. This *post hoc* research may be best facilitated through invoking methodologies such as focus groups and structured questionnaires. However, the role of the Delphi investigation in providing guidance on the nature and diversity of factors requiring further consideration may be critical.

Chapter Seven

Discussion

7.1. Overview

This thesis has assessed the potential of horizontally integrated aquaculture from a systems perspective. Problems associated with the discharge of wastewater from commercial aquaculture operations and limitations to current waste management strategies were described in Chapter 1. In Chapter 2 the potential role of horizontally integration in ameliorating negative impacts associated with aquaculture wastewater was discussed. A definition for these production systems was developed and management strategies that facilitate horizontal integration reviewed. The development and application of a bioeconomic model suitable for assessing the potential of a constructed wetland and recreational fishery as horizontally integrated production systems was presented in Chapter 3. Outputs generated by the ADEPT model were then validated against observations from a commercial aquaculture facility and a constructed wetland with a recreational fishery, functioning under comparable operating conditions to those envisaged for horizontally integrated systems. The model and approach developed though this case study was applied in Chapter 4 to assess the potential of using constructed mangrove wetlands to treat shrimp culture wastewater. The same strategy was used in Chapter 5 to evaluate a rational design approach to lagoon-based wastewater treatment and reuse through aquaculture, as compared to conventional and traditional practices. Finally, Chapter 6 presented the findings of a Delphi investigation that explored the nature and importance of key factors that stakeholders perceive as having a significant role in influencing the decision-making process when considering horizontal integration. This chapter presents an assessment of the appropriateness of the methodologies adopted in this study, together with a review of the major findings from the preceding chapters. Based on this review, important emerging thematic issues are discussed and appropriate recommendations formulated to guide future research and development work regarding horizontally integrated aquaculture.

7.2. Discussion of methodology

Bioeconomic modelling

Adoption of a bioeconomic modelling approach enabled key physical, managerial and financial aspects of the proposed horizontally integrated systems to be assessed simultaneously. From the review presented in Chapter 2 it was apparent that the majority of studies dealing with horizontally integrated aquaculture had focused on addressing key technical constraints or optimising the treatment performance or productivity of the integrated components. Few studies had, however, considered the practical and financial implications for the operator of the primary aquaculture activity.

Through consultation with the managers of a commercial aquaculture facility it was possible to identify some of the key issues that determine the wastewater management strategy employed. Primarily, these factors related to cost, reliability and the required treatment standard as stipulated by the regulatory authorities. These issues required consideration when developing the proposed bioeconomic model to ensure that the resulting outputs were practical, and of use to commercial producers. Ease of operation was a key requirement for the model; therefore, the input variables required by the model relate to information that should be readily available to managers. It was envisaged that using a proprietary spreadsheet package to develop the model would increase the prospects for the use of the product. It was anticipated that using the ADEPT model to simulate waste outputs and treatment performance of conventional treatment strategies could assist managers of commercial operations to identify periods in the culture cycle when there is a risk of the farm discharge contravening discharge consent standards. Refined management strategies e.g. restricted feeding, increasing wastewater treatment capacity at the site or partial harvesting, could then be adopted during high-risk periods.

Although developing the model in consultation with commercial operators helped inform the modelling process, there were constraints to formulation and testing of the model. Relationships and assumptions used to develop the model did not always relate specifically to the production or treatment system under consideration, however, where possible, only information relating to comparable systems was used. A major constraint was the absence of information relating to the likely treatment effect of a trout fishery. The k-C* models for surface-flow wetlands proposed by Kadlec and Knight (1996) were therefore tested in this context. Although the model outputs were tested, and deficiencies identified, there is a need to further refine and calibrate the model in order that reliable and robust outcomes are assured. However, despite its limitations, the ADEPT model provides an initial step for assessing the potential of horizontal integration for commercial aquaculture and identifying high potential strategies worthy of further and more intensive study.

The next phase in such an assessment would probably involve the development of a pilot-scale system, although, as was evident from the literature review, these are often developed outside the context of commercial aquaculture. However, if the commercialisation of horizontally integrated approaches to aquaculture wastewater is to be an objective of such studies, they must be undertaken in close association with commercial partners. Otherwise, potentially useful findings may be only narrowly disseminated and consequently have a limited impact.

Testing the predicted performance of selected approaches to horizontal integration against observations from comparable commercial systems demonstrated the potential of the chosen strategies, but there were also limitations to this approach. The systems monitored were large in scale and it was not possible to control the prevailing operating conditions. As these systems were open to a range of external variables, it was often not possible to know which were the most influential in dictating the behaviour of the system. Testing the model outputs against pilot-scale systems, where more variables could be regulated could help clarify this. An alternative approach would be to test the findings from this study with observations from a wider range of comparable systems. This would be the first step in developing robust, generalised models for key variables, such as the treatment performance of extensive open-water systems.

The outputs of the modelling exercises presented here require consideration from a more general perspective. Although proposed as an approach that would assess a range of issues such as treatment performance, productivity and financial returns, these represent only a part of the systems context for developments in the aquaculture industry. Other facets of the systems in which aquaculture functions e.g. social, economic, environmental and legislative aspects, also require consideration.

Delphi investigation

To better understand the nature and importance of factors that could influence the adoption of horizontally integrated aquaculture, a Delphi investigation with stakeholders was undertaken. This approach elicited a wide range of factors considered by the participants to represent key benefits and constraints associated with horizontally integrated aquaculture. However, due to the relatively small number of participants involved and the limited focus of the study, the findings should only be regarded as general indicators.

Ideally, variations in the perceptions of respondents would have been assessed with respect to their background and area of interest, as demonstrated by Caffey (1998). The priority different groups gave to the various constraints and opportunities based on whether they were researchers, operators, regulators or environmentalists, and which production systems they were involved with would have provided useful information. Although the Delphi investigation was used here to facilitate an assessment of the systems features affecting the prospects for horizontally integrated aquaculture, key areas of potential or conflict in particular settings may have been lost.

A more informative approach could have been to specify a particular aquaculture practice, such as smolt production, with which to consider the prospects for horizontal integration. However, such a targeted approach would limit the numbers of potential participants considering themselves adequately qualified to engage in the process. Where time and resources are available it may be beneficial to identify and engage with selected groups; the outputs from such studies could highlight key areas of opportunity or conflict with respect to particular settings. However, caution must be exercised so as not to restrict too tightly the views and experience of participants, as this risks losing the general perspective gained through the Delphi investigation.

The Delphi investigation provided the opportunity for the assessment of stakeholders' perceptions of the relative merits of wastewater management strategies other than horizontal integration. In this investigation it was evident that approaches other than treating wastewater were given priority, e.g. improved feed management and husbandry practices. It would not have been possible to obtain this perspective from a modelling exercise carried out in isolation, therefore, as a complementary approach for use with a more focused modelling exercise, the Delphi could place the modelling output into context. Delphi investigation outputs would also be useful in guiding more targeted research, focusing resources on strategies that have good prospects for developing appropriate systems that are adopted by commercial operators.

The approaches employed in this study proved useful in highlighting possible constraints and potential opportunities associated with the development of horizontally integrated aquaculture. However, as with other modelling approaches, the findings should be interpreted with caution; they may best be used to guide the development of pilot-scale systems in association with commercial aquaculture, prior to the development of full-scale commercial systems. Such an approach would facilitate the testing of the proposed strategy under a range of operating conditions and permit an assessment of the likely risks associated with the system. When dealing with ecologically based wastewater treatment approaches the possible impact of unexpected perturbations must be considered, and where possible, included in the planning of the proposed system. The development of pilot-scale systems would also permit other aspects of horizontally integration, such as the perceptions of the commercial operators, to be assessed and consumer acceptance of products tested.

7.3. Implications of findings

From the review presented in Chapter 1 it is apparent that concerns regarding the impact of aquaculture wastewater have been addressed through several approaches. Effluent loads have been significantly reduced through careful feed management, the formulation of energy-dense and low pollution diets, and through more efficient sludge collectors and mechanical filters. Furthermore, farm managers have developed guidelines to promote good management practices that reduce negative environmental impacts associated with aquaculture (Gavine et al., 1996). However, the potential benefits of improved management strategies in reducing waste discharges are constrained by practical considerations. The effectiveness of current approaches to aquaculture wastewater treatment, such as settlement and mechanical filtration, is limited, and should stricter discharge standards be introduced it is unlikely that they could be met. Conventional approaches to treatment also result in the production of sludge that requires disposal and may ultimately represent a source of pollution.

As well as meeting the demands of regulators, aquaculture producers should also be conscious of the concerns expressed by the public regarding the origins of products they consume and their environmental credentials. Wasteful producers may find themselves at a disadvantage while products from operators employing environmentally friendly options may command a premium; it has been suggested that market perceptions and environmental attributes will increasingly drive future growth within the aquaculture industry (Young et al, 1999). Therefore, the proactive development of management approaches that address problems associated with discharging aquaculture wastewater could confer a significant advantage on operators.

The prospect of stricter discharge consents, pollution taxes and increased preference of consumers for environmentally friendly goods was seen as justification for further research into improved treatment strategies for aquaculture wastewater. Furthermore, there is a growing recognition that nutrients entrained in aquaculture wastewater represent a potentially valuable resource. Treatment options that facilitate the exploitation of this resource to culture aquatic species, whilst ameliorating the negative impacts associated with aquaculture wastewater, could be attractive to both consumers and operators. This was the rationale proposed in support of *horizontally integrated aquaculture*.

The review in Chapter 2 identified a range of strategies investigated or demonstrating potential for horizontal integration with semi-intensive and intensive aquaculture. However, few examples of full-scale systems associated with commercial aquaculture were noted. From this, it was inferred that although technically viable in a number of settings, addition financial, economic, managerial, social, institutional and environmental factors might be constraining commercialisation.

The development and application of the ADEPT model permitted some of these potential constraints to be evaluated, including the treatment performance and reliability of horizontally integrated systems, and the managerial and financial demands associated with particular case studies. The first of these considered prospects for the integration of a constructed wetland and trout fishery to treat the wastewater from a commercial smolt unit in Scotland. Outputs from the modelling exercise were in general agreement with the treatment potential observed in the comparable system, however, further work is required to calibrate the model.

Despite the potential limitations to adopting the integrated trout fishery and reedbed, it was concluded that such an approach to horizontal integration may have a role to play in facilitating the reuse of wastewater from a commercial smolt farm in Scotland: achieving higher treatment standards, generating additional income and creating employment opportunities. However, the specific nature of the scenario developed suggests that the potential of this particular approach is likely to be limited for operators in remote locations where demand for angling is low. A further limitation was also identified: the reduction of DO levels in surface-flow wetlands to low levels may represent an intractable problem. A potential solution would be to reconfigure the design so that the reedbed precedes the fishery, although, this could in turn result in elevated SS levels in the final discharge. However, if it were possible to convince the regulatory authorities that the overall impact of the wetland treatment systems was positive, and that SS generated in the wetland were mainly complex organic matter that would only have a limited impact on the receiving environment, then such an approach may still be feasible. Further work is required to substantiate this argument.

Although horizontally integrated aquaculture may offer several advantages, two constraints identified in the case studies were the land required and the initial capital expenditure. Where land and capital are available for development, in most cases the financial returns generated by the most promising systems were only considered marginally attractive. Small returns predicted through the financial assessment could be subsidised by other potential benefits, such as a premium for products from the primary aquaculture activity, the sale of additional treatment capacity through a tradable permit scheme or a reduction in pollution tax. However, where such additional benefits were perceived, the task of assessing their likely impact on the financial returns in the proposed case studies requires further investigation. The ADEPT model provides a useful framework from which to develop such assessments, although the actual estimation of the likely magnitude of any potential benefits would require a more wide ranging assessment undertaken at a regional or national level.

Expanding the diversity of farmed aquaculture products by the adoption of horizontal integration could disperse financial, environmental and disease risks associated with intensive monoculture-based aquaculture. Culturing a range of products with varying production cycles presents opportunities to market products throughout the year, contributing to a more even distribution of cash-flows and making more efficient use of infrastructure, the workforce and marketing channels.

Horizontal integration does however, result in additional management demands and increased workloads; in the scenarios developed here, this factor has been accounted for by costing additional labour into the model. However, on existing farms, managers may be reluctant to employ more staff and the existing workforce may be required to perform additional tasks. The distribution of this labour demand may not be uniform and potential synergies and conflicts with workloads associated with the operation of the primary aquaculture facility would require further investigation. Where generated financial returns are sufficient, managers may be more willing to engage extra employees to operate and maintain the system; such an increase in the size of the workforce could represent an important source of employment in remote or deprived areas.

Another area requiring further consideration is the assessment of risks associated with horizontal integration, in particular where systems depend on ecologically based wastewater treatment processes that may be susceptible to external factors, such as environmental perturbations. However, examples such as the traditional approaches developed in peri-urban Calcutta suggest that ecologically based systems can represent a viable and resilient wastewater treatment and reuse option.

The limited scope of previous studies considering horizontal integration was considered here as a major constraint to developing commercial systems. This failing was addressed in this study through the application of the Delphi technique with a range of stakeholders. The findings of this assessment would also be useful in indicating where the priorities for subsequent research and development lie, and could contribute to focusing resources on formulating design and management plans that meet the demands and expectations of the diverse stakeholder groups that exert an influence on the decision-making process. The role of the Delphi investigation in providing guidance on the nature and diversity of factors requiring further consideration was regarded as a critical aspect of the work presented in this thesis.

7.4. Conclusions and recommendations

This study presents an assessment concerning the potential of horizontally integrated aquaculture, with outcomes assessed from a systems-based perspective. The development and application of the ADEPT bioeconomic model to assess the potential of using a constructed wetland and trout fishery to treat the wastewater from a commercial smolt unit in Scotland was generally effective in predicting the composition of wastewater outputs from the farm, and the effect of the selected treatment strategies. The model was also successfully applied to two further case studies: the potential of treating wastewater from shrimp farms in Thailand using a constructed mangrove wetland, and the possible advantage of a rational design approach to lagoon-based wastewater treatment and reuse, as opposed to a conventional design and traditional practices developed in peri-urban

Calcutta. For all case studies, it was possible to simulate the general treatment and financial performance of the proposed scenarios with a relatively good degree of agreement between the model outputs and observed data.

A Delphi investigation permitted the modelling outputs, and horizontally integrated aquaculture in general, to be considered from a systems-based perspective. It was concluded that horizontal integration could have a role to play in developing more sustainable aquaculture practices, however, further work will be required to assess the prospects for such an approach based on the local setting and production system in question.

From this study several key recommendations may be suggested. Although the prospects for specific approaches to horizontal integration were considered here, further alternatives require investigation as the likely demand for the specific products and services generated by horizontal integration proposed in this thesis e.g. angling, may be limited. The diverse environments in which aquaculture operates suggests that a reassessment of the strategies proposed here would be required to truly test their prospects in particular settings. Assumptions made for the case studies presented here may be expected to change over time, as might the results of a Delphi investigation with stakeholders. Therefore, the assessment of horizontally integrated systems should not be static, but an ongoing process. This process should become more refined through the assimilation of additional information and validation of assumptions, leading to more robust assessments that have greater potential to facilitate the development of appropriate and sustainable approaches to horizontal integration.

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Inputs

Summarises input variable required:

input information required by the ADEPT model

Stock model

Describes the stocks biomass over one year:

if known, the exact biomass held on the farm can be entered on a monthly basis

Wastewater output

Calculates the concentrations of waste fractions in the wastewater from the smolt unit:

if known, exact concentrations can be entered

Wastewater treatment

Calculates the percentage reduction in waste fractions achieved by the ADEPT system:

the performance criterion for the user defined system should be entered here.

Horizontal integration

Estimates the biomass and value of products from the ADEPT system:

the cost of seed, rates of production or the value of products and angling can be entered if known

Financial considerations

Calculates the capital costs and operating costs of adopting the ADEPT system:

the majority of these costs are based on information provided in sections 6 and 8 of the Inputs page

Cost return

Presents an outline of the initial costs, annual operating costs, income generated and some general financial indicators:

no inputs are required here

Cash flow

Displays the projected cash flow during the first ten years of operation:

discounted cash flow is used to calculate the Net Present Value (NPV) and Internal Rate of Return (IRR)

Outputs

Presents a summary of outputs from the scenario modelled:

outputs include treatment effect, physical characteristics and financial and economic indicators

Aquatic Downstream Ecological Production and Treatment for aquaculture wastewater: inputs

User inputs required by the ADEPT model:

1. Culture unit management: select and define parameters to describe the stock

Primary aquaculture facility: smolt unit (1) or shrimp farm (2)	1
Number of juveniles stocked in the culture facility (No.)	100000
Expected mortality (%)	0%
Average biomass (g) of stock at harvest	70.0
Change in stock biomass based on a calculation from the feeding rate and FCR (0) or observation (1)	0
Expected Feed Conversion Ratio (FCR)	1.15
Maximum feeding rate (% body weight/d)	4.0%
N content of the feed (gN/kg feed)	72.0
P content of the feed (gP/kg feed)	12.0
Dry-matter content of the feed (gDM/kg feed)	950.0

2. Physical and economic characteristics of the culture site: define the key economic and physical variables

Estimated capital cost of the culture facility (£ '000)	1000
Estimated annual operating costs for the culture facility (£ '000/y)	50
Hydraulic volume of culture unit (m^3)	1000
Land area occupied by culture facility (m^2)	5000
Average flow rate through smolt unit or maximum through shrimp farm (m^3/d)	3400
Percentage of culture water flowing to treatment	100%

3. Wastewater composition: select the required method, *only applies to smolt production

SS:	based on literature (0), observation (1), stock biomass (2)* or mass-balance equation (3)	3
TN:	based on literature (0), observation (1) or mass-balance equation (2)	2
TAN:	based on observation (0) specific growth rate (1)* or feed input (2)*	2
TP:	based on literature (0), observation (1), stock biomass (2)* or mass-balance equation (3)	3
BOD:	based on literature (0), observation (1), stock biomass (2)* or feed input (3)*	3
Oxygen consumption:	based on observation (0) or temperature (1)*	1

4. Background and target concentrations: insert values for abstracted water and permissible change

Mean concentrations (mg/l) in abstracted water:	SS	1.75
	TAN	0.07
	TN	0.00
	TP	0.00
	BOD	1.25
	DO (% saturation)	100%
	pH	7.2
Permissible change in concentrations (mg/l) i.e. discharge consent standards:	SS	5.00
	TAN	0.10
	BOD	4.00

5. Proposed treatment system configuration: select the position of the water treatment components

Treatment:	absent (0) or present (1)	1
Drum filters:	absent (0) or position in treatment stream (1,2,3,etc.)	1
Fate of sludge produced by the filter:	storage tanks (0)* or flowing to other treatment units (1)	0
Settlement pond:	absent (0) or position in treatment stream (1,2,3,etc.)	0
Constructed wetland:	absent (0) or position in treatment stream (1,2,3,etc.)	0
Recreational fishery:	absent (0) or position in treatment stream (1,2,3,etc.)	0

*Note: where storage tanks are stipulated, the filtrate will flow to the ADEPT system

6. Horizontally integrated aquaculture: select the required characteristics of the production/treatment system

Recreational fishery: trout (1) or coarse fish (2)	0
Catch release (0) or restocking required (1)	0
Cost of stock fish: literature (0) or user defined (1)	0
Expected or actual stocking density in the fishery (no./m^2): recommended (0) or actual (1)	0
Expected value of angling (£/day): literature (0) or user defined (1)	0
Type of macrophyte produced in constructed wetland: common reed (1) or mangrove (2)	0
Macrophyte production in wetland based on: literature (0) or observation (1)	0
Market value of macrophyte biomass based on: literature (0) or user defined (1)	0

7. Economic considerations for operation: define the key economic parameters concerning operation

Expected value of land (£/ha)	£7,400
Expected cost for site development (£/ha)	£49,500
Treatment units connected via.: pipework (0) or open channels (1)	0
Value of time for manager of a smolt unit (£/h) or shrimp farm (£/y)	£20.0
Cost of farm worker at a smolt unit (£/h) or shrimp farm (£/y)	£10.0
Infrastructure depreciation period (5, 10 or 15 y)	15
Plant and equipment depreciation period (5, 10 or 15 y)	10

Aquatic Downstream Ecological Production and Treatment for aquaculture wastewater: outputs

Outputs produced by the ADEPT model:

1. A physical description of the treatment system: describes the system as defined by current inputs

Average water flow through the treatment system (m^3/h)	141.7
Average hydraulic retention time within treatment system (d)	0.0
Land area required (ha)	0.0
Mean temperature (°C) at the site	8.5

2. Concentrations of waste discharged from the site: mean concentration (mg/l) after deducting background levels

Mean concentration (mg/l) in site discharge:	SS	-0.57
	TAN	0.18
	TN	0.15
	TP	0.02
	BOD	1.04
	DO	-2.34

3. Treatment performance: indicates the mean percentage change in the various parameters

Mean percentage change in waste concentration due to treatment:	SS	-111.9%
	TAN	0.0%
	TN	-46.5%
	TP	-53.5%
	BOD	-10.4%
	DO	0.0%

4. General indicators: physical indicators of the treatment systems performance

Ratio of culture unit to treatment system area		0.0
Ratio of culture unit to treatment system hydraulic retention time		0.0
Estimated N retention in:	harvestable macrophyte biomass (kg/y)	0
	extensive aquaculture system (kg/y)	0
Percentage of N released from culture present in harvested biomass		0%
Estimated P retention in:	harvestable macrophyte biomass (kg/y)	0
	extensive aquaculture system (kg/y)	0
Percentage of P released from culture present in harvested biomass		0%

5. Ecological downstream production: describes production in the treatment system

Maximum area required for the constructed wetland (ha)	0.0
Maximum area required for the extensive aquaculture system (ha)	0.0
Macrophyte biomass production (t/y)	0.0
Value of macrophyte biomass produced (£ '000/y)	0.0
Coarse fish production (kg/y)	0.0
Value of coarse fish production (£ '000/y)	0.0
Number of anglers visiting the site (no./y)	0.0
Value of angling provided (£ '000)	0.0
Annual crayfish production within the aquaculture production/treatment system (kg/y).	0.0
Net value of the crayfish produced within the aquaculture production/treatment system (£ '000).	0.0

6. General economic indicators: likely effects on costs and returns of selected options

Capital cost (£'000)	16.0
Depreciation on capital investment (£ '000/y)	1.5
Annual operating costs (£'000/y)	1.6
Ratio of culture unit to treatment system capital costs	0.0
Ratio of culture unit to treatment system operating costs	0.0

7. Economic performance: broad economic indicators of performance

Net income (£'000/y)		0.0
Annual profit (£ '000)		-1.6
Profit generated per unit area (£ '000/ha)		-301.4
Annual rate of return on initial capital cost		-9.9%
Annual rate of return on operating costs		-100.0%
Pay-back period (y)		na
Residual 10 year NPV:	5%	-25.9
	10%	-22.8
	15%	-20.5
	20%	-18.6
IRR over:	ten years	na
	twenty years	na
	fifty years	na

Stock model for: smolts

	April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March	Max.	Ave.
Days.	30	31	30	31	31	30	31	30	31	31	28	31	31.0	30.4
Mean temperature.	7.1	10.3	12.0	14.4	14.6	12.1	8.6	5.6	4.1	3.6	4.4	5.0	14.60	8.48
Stock model														
Feed input (% body wt. day)	1.00%	4.00%	4.00%	4.00%	3.00%	1.25%	1.00%	0.70%	0.20%	0.25%	0.75%	0.75%	4.00%	1.74%
Feed input (kg)	33	171	343	725	1130	824	904	777	271	357	1034	1354	7924	
FCR	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15	1.15
Average fish weight (g) at start of month	1.15	1.45	3.01	6.14	12.76	23.08	29.15	37.01	43.77	46.13	49.24	58.23	58.2	25.9
Average fish weight (g) at end of month	1.45	3.01	6.14	12.76	23.08	29.15	37.01	43.77	46.13	49.24	58.23	70.00	70.0	31.7
Total stock biomass (kg) at start of month	109	138	286	585	1215	2198	2915	3701	4377	4613	4924	5823	5823	2574
Total stock biomass (kg) at end of month	138	286	585	1215	2198	2915	3701	4377	4613	4924	5823	7000	7000	3148
User defined stock model														
Total stock biomass (kg) at start of month	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0	0
Total stock biomass (kg) at end of month	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0	0
Summary														
Total stock biomass (kg) at start of month	109	138	286	585	1215	2198	2915	3701	4377	4613	4924	5823	5823	2574
Total stock biomass (kg) at end of month	138	286	585	1215	2198	2915	3701	4377	4613	4924	5823	7000	7000	3148
Mean biomass (kg)	123	212	436	900	1707	2557	3308	4039	4495	4768	5373	6411	6411	2861

Waste output from the culture unit

	April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March	Max	Ave
Days	30	31	30	31	31	30	31	30	31	31	28	31	31.0	30.4
Mean temperature	7.1	10.3	12.0	14.4	14.6	12.1	8.6	5.6	4.1	3.6	4.4	5.0	14.6	8.5
Feed input (% body wt./d)	1.0	4.0	4.0	4.0	3.0	1.3	1.0	0.7	0.2	0.3	0.8	0.8	4.0	1.7
Feed input (kg/d)	1.2	8.5	17.4	36.0	51.2	32.0	33.1	28.3	9.0	11.9	40.3	48.1	51.2	26.4
FCR	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2
Total stock biomass at start of month (kg)	109	138	286	585	1215	2198	2915	3701	4377	4613	4924	5823	5822.8	2573.7
Total stock biomass at end of month (kg)	138	286	585	1215	2198	2915	3701	4377	4613	4924	5823	7000	7000.0	3147.9
Mean biomass on the farm (kg)	123	212	436	900	1707	2557	3308	4039	4495	4768	5373	6411	6411.4	2860.8
Water passing through the culture unit (m^3/d)	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400.0	3400.0
Water flowing to treatment (m^3/d)	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400.0	3400.0
Conversion factor: g/kg/d to mg/l i.e. (l/kg/d)	27.5	16.0	7.8	3.8	2.0	1.3	1.0	0.8	0.8	0.7	0.6	0.5	27.5	5.2

Change in wastewater composition (mg/l)

SS	April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March	Max	Ave
Literature	14	14	14	14	14	14	14	14	14	14	14	14	14.0	14.0
Observation	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0
Stock biomass	0.06	0.10	0.21	0.44	0.83	1.25	1.62	1.97	2.19	2.33	2.62	3.13	3.1	1.4
Mass-balance equation	0.24	1.22	2.53	5.16	8.04	6.06	6.43	5.71	1.93	2.54	8.15	9.63	9.6	4.8
Wastewater concentration	0.24	1.22	2.53	5.16	8.04	6.06	6.43	5.71	1.93	2.54	8.15	9.63	9.6	4.8
TN														
Literature	1.400	1.400	1.400	1.400	1.400	1.400	1.400	1.400	1.400	1.400	1.400	1.400	1.4	1.4
Observation	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.0	0.0
Mass-balance equation	0.014	0.070	0.145	0.297	0.463	0.349	0.370	0.329	0.111	0.147	0.469	0.555	0.6	0.3
Wastewater concentration	0.014	0.070	0.145	0.297	0.463	0.349	0.370	0.329	0.111	0.147	0.469	0.555	0.6	0.3
TAN														
Observation	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.00	0.00
Specific growth rate	0.002	0.003	0.006	0.011	0.022	0.032	0.042	0.051	0.057	0.061	0.068	0.081	0.08	0.04
Feed input	0.009	0.045	0.093	0.190	0.297	0.223	0.237	0.211	0.071	0.094	0.300	0.355	0.36	0.18
Wastewater concentration	0.009	0.045	0.093	0.190	0.297	0.223	0.237	0.211	0.071	0.094	0.300	0.355	0.36	0.18
TP														
Literature	0.125	0.125	0.125	0.125	0.125	0.125	0.125	0.125	0.125	0.125	0.125	0.125	0.1	0.1
Observation	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.0	0.0
Stock biomass	0.001	0.002	0.004	0.007	0.014	0.021	0.027	0.033	0.036	0.038	0.043	0.052	0.1	0.0
Mass-balance equation	0.002	0.012	0.025	0.052	0.080	0.061	0.064	0.057	0.019	0.025	0.081	0.096	0.1	0.0
Wastewater concentration	0.002	0.012	0.025	0.052	0.080	0.061	0.064	0.057	0.019	0.025	0.081	0.096	0.1	0.0
BOD														
Literature	8	8	8	8	8	8	8	8	8	8	8	8	8.0	8.0
Observation	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0
Stock biomass	0.04	0.07	0.14	0.30	0.56	0.84	1.09	1.33	1.48	1.58	1.78	2.12	2.1	0.9
Feed input	0.05	0.37	0.77	1.59	2.26	1.41	1.46	1.25	0.40	0.53	1.78	2.12	2.3	1.2
Wastewater concentration	0.05	0.37	0.77	1.59	2.26	1.41	1.46	1.25	0.40	0.53	1.78	2.12	2.3	1.2
Oxygen consumption														
Observation	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0
Temperature	0.10	0.26	0.69	2.03	3.97	4.13	3.22	2.54	2.28	2.25	2.84	3.70	4.1	2.3
Wastewater concentration	0.10	0.26	0.69	2.03	3.97	4.13	3.22	2.54	2.28	2.25	2.84	3.70	4.1	2.3

Summary

	April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March	Max	Ave
SS	0.24	1.22	2.53	5.16	8.04	6.06	6.43	5.71	1.93	2.54	8.15	9.63	9.63	4.80
TAN	0.01	0.04	0.09	0.19	0.30	0.22	0.24	0.21	0.07	0.09	0.30	0.36	0.36	0.18
TN	0.01	0.07	0.15	0.30	0.46	0.35	0.37	0.33	0.11	0.15	0.47	0.55	0.55	0.28
TP	0.00	0.01	0.03	0.05	0.08	0.06	0.06	0.06	0.02	0.03	0.08	0.10	0.10	0.05
BOD	0.05	0.37	0.77	1.59	2.26	1.41	1.46	1.25	0.40	0.53	1.78	2.12	2.26	1.17
DO	-0.10	-0.26	-0.69	-2.03	-3.97	-4.13	-3.22	-2.54	-2.28	-2.25	-2.84	-3.70	-0.10	-2.34

Dimensioning and treatment performance of the ADEPT system components

Month	April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March	Max	Avg
Days	30	31	30	31	31	30	31	30	31	31	28	31	31	30
Mean temperature	7.1	10.3	12.0	14.4	14.6	12.1	8.6	5.6	4.1	3.6	4.4	5.0	15	8
Water flowing to treatment (m^3/d)	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400

Dimensioning of treatment components

Summary

Area required by the treatment system components (m^2)

Drum Filter	50	50	50	50	50	50	50	50	50	50	50	50	50	50
Configuration complete	na	na	na	na	na	na	na	na	na	na	na	na	0	0
Configuration complete	na	na	na	na	na	na	na	na	na	na	na	na	0	0
Configuration complete	na	na	na	na	na	na	na	na	na	na	na	na	0	0
Total area required (ha)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Hydraulic retention time (d)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01

Drum filter

Treatment volume (m^3/d)	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400	3400
Sludge production (l/m^3 of influent)	20	20	20	20	20	20	20	20	20	20	20	20	20	20
Total sludge production (m^3/d)	68	68	68	68	68	68	68	68	68	68	68	68	68	68
Area for filter and drying beds (m^2)	50	50	50	50	50	50	50	50	50	50	50	50	50	50
Hydraulic residence time (min)	10	10	10	10	10	10	10	10	10	10	10	10	10	10

Settlement pond

Hydraulic loading (m^3/d)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Length (m)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Breadth (m)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Depth (m)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Area required (m^2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydraulic volume (m^3)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydraulic retention time (h)	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Constructed wetland

Treatment volume (m^3/d)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SS	na	na	na	na	na	na	na	na	na	na	na	na		
TAN	na	na	na	na	na	na	na	na	na	na	na	na		
BOD	na	na	na	na	na	na	na	na	na	na	na	na		
N assimilation	na	na	na	na			na	na	na	na				0
P assimilation	na	na	na	na			na	na	na	na				0
Area required (m^2) Max	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Depth (m)	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30
Proportion occupied by plants (%)	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10
Hydraulic loading rate ($m^3/m^2/d$)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Hydraulic residence time (d)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Extensive fishery

Treatment volume (m^3/d)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Target hydraulic loading ($m^3/m^2/d$)	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.050	0.050
Area required (m^2)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Depth (m)	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20
Hydraulic residence time (d)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

SS

Month		April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March		Sum
Mass flow from culture (kg)		24.6	128.1	257.6	543.9	847.8	618.3	677.8	582.9	203.5	268.1	775.5	1015.3		5943.3
Concentration (mg/l)														Max	Avg
Wastewater (+C*)		1.99	2.97	4.28	6.91	9.79	7.81	8.18	7.46	3.68	4.29	9.90	11.38	11.38	6.55
Drum Filter		0.36	0.53	0.77	1.24	1.76	1.41	1.47	1.34	0.66	0.77	1.78	2.05	2.05	1.18
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
No. of treatment modules	1														
Treatment discharge (-C*)		-1.39	-1.22	-0.98	-0.51	0.01	-0.34	-0.28	-0.41	-1.09	-0.98	0.03	0.30	0.30	-0.57
Final site discharge		-1.39	-1.22	-0.98	-0.51	0.01	-0.34	-0.28	-0.41	-1.09	-0.98	0.03	0.30	0.30	-0.57

Drum filter

Influent (mg/l)	1.99	2.97	4.28	6.91	9.79	7.81	8.18	7.46	3.68	4.29	9.90	11.38	11.38	6.55
Removal efficiency	82%	82%	82%	82%	82%	82%	82%	82%	82%	82%	82%	82%	0.82	0.82
Sludge (mg/l)	81.63	121.57	175.28	283.32	401.52	320.27	335.40	306.06	150.92	176.05	405.73	466.71	466.71	268.71
Filtrate (mg/l)	0.36	0.53	0.77	1.24	1.76	1.41	1.47	1.34	0.66	0.77	1.78	2.05	2.05	1.18

Settlement pond

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Removal efficiency	47%	47%	47%	47%	47%	47%	47%	47%	47%	47%	47%	47%	0.47	0.47
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

Constructed wetland

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
C*	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
k	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000		
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

Extensive fishery

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
C*	na	na	na	na	na	na	na	na	na	na	na	na		
k	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000		
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

TAN (mg/l)

Month		April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March		Sum
Mass flows from culture (kg)		0.91	4.72	9.50	20.05	31.25	22.79	24.99	21.49	7.50	9.88	28.59	37.43		219.09
Concentration (mg/l)														Max	Avg
Wastewater composition (+C*)		0.079	0.115	0.163	0.260	0.367	0.293	0.307	0.281	0.141	0.164	0.370	0.425	0.425	0.247
Drum Filter		0.079	0.115	0.163	0.260	0.367	0.293	0.307	0.281	0.141	0.164	0.370	0.425	0.425	0.247
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
No. of treatment modules	1														
Treatment discharge (-C*)		0.009	0.045	0.093	0.190	0.297	0.223	0.237	0.211	0.071	0.094	0.300	0.355	0.355	0.177
Final site discharge		0.009	0.045	0.093	0.190	0.297	0.223	0.237	0.211	0.071	0.094	0.300	0.355	0.355	0.177

Drum filter

Influent (mg/l)	0.079	0.115	0.163	0.260	0.367	0.293	0.307	0.281	0.141	0.164	0.370	0.425	0.425	0.247
Removal efficiency	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.000	0.000
Sludge (mg/l)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Filtrate (mg/l)	0.079	0.115	0.163	0.260	0.367	0.293	0.307	0.281	0.141	0.164	0.370	0.425	0.425	0.247

Settlement pond

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Removal efficiency	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.000	0.000
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000

Constructed wetland

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
C*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		
k	10.9	12.3	13.2	14.5	14.6	13.2	11.5	10.2	9.6	9.5	9.8	10.0		
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000

Extensive fishery

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000

TN (mg/l)

Month		April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March		Sum
Mass flows from culture (kg)		1.42	7.38	14.84	31.33	47.26	36.80	39.04	33.58	11.72	15.44	49.45	56.60		344.85
Concentration (mg/l)														Max	Avg
Wastewater composition (+C*)		0.014	0.070	0.145	0.297	0.463	0.349	0.370	0.329	0.111	0.147	0.469	0.555	0.555	0.277
Drum Filter		0.007	0.037	0.078	0.159	0.248	0.187	0.198	0.176	0.060	0.078	0.251	0.297	0.297	0.148
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
No. of treatment modules	1														
Treatment discharge (-C*)		0.007	0.037	0.078	0.159	0.248	0.187	0.198	0.176	0.060	0.078	0.251	0.297	0.297	0.148
Final site discharge		0.007	0.037	0.078	0.159	0.248	0.187	0.198	0.176	0.060	0.078	0.251	0.297	0.297	0.148
Drum filter															
Influent (mg/l)		0.014	0.070	0.145	0.297	0.463	0.349	0.370	0.329	0.111	0.147	0.469	0.555	0.555	0.277
Removal efficiency		46.5%	46.5%	46.5%	46.5%	46.5%	46.5%	46.5%	46.5%	46.5%	46.5%	46.5%	46.5%	0.465	0.465
Sludge (mg/l)		0.323	1.627	3.382	6.911	10.772	8.118	8.612	7.653	2.586	3.407	10.909	12.901	12.901	6.433
Filtrate (mg/l)		0.007	0.037	0.078	0.159	0.248	0.187	0.198	0.176	0.060	0.078	0.251	0.297	0.297	0.148
Settlement pond															
Influent (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Removal efficiency		0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.000	0.000
Filtrate (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Constructed wetland															
Influent (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
C*		1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5		
k		11.7	13.7	14.9	16.7	16.9	15.0	12.6	10.9	10.1	9.9	10.3	10.6		
Filtrate (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Extensive fishery															
Influent (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
C*		1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5		
k		11.7	13.7	14.9	16.7	16.9	15.0	12.6	10.9	10.1	9.9	10.3	10.6		
Filtrate (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000

TP

Month		April	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March		Sum
Mass flow from culture (kg)		0.25	1.28	2.58	5.44	8.48	6.18	6.78	5.83	2.04	2.68	7.75	10.15		59.43
Concentration (mg/l)														Max	Avg
Wastewater composition (+C*)		0.002	0.012	0.025	0.052	0.080	0.061	0.064	0.057	0.019	0.025	0.081	0.096	0.096	0.048
Drum Filter		0.001	0.006	0.012	0.024	0.037	0.028	0.030	0.027	0.009	0.012	0.038	0.045	0.045	0.022
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
No. of treatment modules	1														
Treatment discharge (-C*)		0.001	0.006	0.012	0.024	0.037	0.028	0.030	0.027	0.009	0.012	0.038	0.045	0.045	0.022
Final site discharge		0.001	0.006	0.012	0.024	0.037	0.028	0.030	0.027	0.009	0.012	0.038	0.045	0.045	0.022
Drum filter															
Influent (mg/l)		0.002	0.012	0.025	0.052	0.080	0.061	0.064	0.057	0.019	0.025	0.081	0.096	0.096	0.048
Removal efficiency		53.5%	53.5%	53.5%	53.5%	53.5%	53.5%	53.5%	53.5%	53.5%	53.5%	53.5%	53.5%	0.535	0.535
Sludge (mg/l)		0.064	0.325	0.675	1.380	2.152	1.621	1.720	1.529	0.517	0.680	2.179	2.577	2.577	1.285
Filtrate (mg/l)		0.001	0.006	0.012	0.024	0.037	0.028	0.030	0.027	0.009	0.012	0.038	0.045	0.045	0.022
Settlement pond															
Influent (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Removal efficiency		0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.000	0.000
Filtrate (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Constructed wetland															
Influent (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
C*		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		
k		12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0		
Filtrate (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
Extensive fishery															
Influent (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000
C*		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		
k		12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0	12.0		
Filtrate (mg/l)		na	na	na	na	na	na	na	na	na	na	na	na	0.000	0.000

BOD (mg/l)

Mass flows (kg/d) from smolts		0.19	1.27	2.61	5.40	7.68	4.79	4.96	4.24	1.35	1.79	6.04	7.21	7.68	3.96
Wastewater composition (+C*)		1.30	1.62	2.02	2.84	3.51	2.66	2.71	2.50	1.65	1.78	3.03	3.37	3.51	2.42
Drum Filter		1.24	1.54	1.92	2.70	3.33	2.53	2.57	2.37	1.56	1.69	2.88	3.20	3.33	2.29
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
No. of treatment modules	1														
Treatment discharge (-C*)		-0.01	0.29	0.67	1.45	2.08	1.28	1.32	1.12	0.31	0.44	1.63	1.95	2.08	1.04
Final site discharge		-0.01	0.29	0.67	1.45	2.08	1.28	1.32	1.12	0.31	0.44	1.63	1.95	2.08	1.04

Drum filter

Influent (mg/l)	1.30	1.62	2.02	2.84	3.51	2.66	2.71	2.50	1.65	1.78	3.03	3.37	3.51	2.42
Removal efficiency	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	5%	0.05	0.05
Sludge (mg/l)	3.26	4.06	5.05	7.10	8.77	6.65	6.77	6.24	4.12	4.44	7.57	8.43	8.77	6.04
Filtrate (mg/l)	1.24	1.54	1.92	2.70	3.33	2.53	2.57	2.37	1.56	1.69	2.88	3.20	3.33	2.29

Settlement pond

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Removal efficiency	41.7%	41.7%	41.7%	41.7%	41.7%	41.7%	41.7%	41.7%	41.7%	41.7%	41.7%	41.7%	0.42	0.42
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

Constructed wetland

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
C*	na	na	na	na	na	na	na	na	na	na	na	na		
k	16	19	21	25	25	21	17	15	13	13	14	14		
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

Extensive fishery

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
C*	na	na	na	na	na	na	na	na	na	na	na	na		
k	16	19	21	25	25	21	17	15	13	13	14	14		
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

DO (mg/l)

Mass flows (kg/d) from smolts		-0.33	-0.90	-2.36	-6.91	-13.49	-14.06	-10.95	-8.65	-7.74	-7.64	-9.67	-12.59	-0.33	-7.94
Background oxygen concentration		12.11	11.19	10.75	10.17	10.12	10.72	11.67	12.59	13.10	13.28	13.00	12.79	13.28	11.79
Culture outputs (+C*)		12.02	10.93	10.05	8.14	6.16	6.59	8.45	10.05	10.82	11.03	10.15	9.09	12.02	9.46
Drum Filter		12.02	10.93	10.05	8.14	6.16	6.59	8.45	10.05	10.82	11.03	10.15	9.09	12.02	9.46
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Configuration complete		na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
No. of treatment modules	1														
Treatment discharge (-C*)		-0.10	-0.26	-0.69	-2.03	-3.97	-4.13	-3.22	-2.54	-2.28	-2.25	-2.84	-3.70	-0.10	-2.34
Final site discharge		-0.10	-0.26	-0.69	-2.03	-3.97	-4.13	-3.22	-2.54	-2.28	-2.25	-2.84	-3.70	-0.10	-2.34

Drum filter

Influent (mg/l)	12.02	10.93	10.05	8.14	6.16	6.59	8.45	10.05	10.82	11.03	10.15	9.09	12.02	9.46
Removal efficiency	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0.00	0.00
Sludge (mg/l)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Filtrate (mg/l)	12.02	10.93	10.05	8.14	6.16	6.59	8.45	10.05	10.82	11.03	10.15	9.09	12.02	9.46

Settlement pond

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Removal efficiency	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0.00	0.00
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

Constructed wetland

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

Extensive fishery

Influent (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00
Filtrate (mg/l)	na	na	na	na	na	na	na	na	na	na	na	na	0.00	0.00

Horizontal integration

	Unit cost	Units	Value
Constructed wetland: **na**			
Dimensions of the constructed wetland			
Maximum area occupied by the constructed wetland (ha)			0.00
Market value of macrophyte biomass (£/kg)			0.000
Market value of macrophyte biomass (£/kg)			0.00
Production in the constructed wetland			
Expected production (kg/m²)			0.00
General production based on the literature (kg/m²)			2.0
Macrophyte biomass production (kg/ha/y)			20000
Total macrophyte biomass production (t/y)			0.0
Gross value of the macrophyte biomass produced (£ '000)			**0.00**
Fishery: **na**			
Stocking the fishery			
Maximum area occupied by the fishery (ha)			0.00
Recommended stocking density (fish/m²)			0.10
Stocking density (fish/m²)			0.00
Total fish stocked or seeded			0
Cost of stock (£/fish)			£2.00
Cost of stock (£/fish)			£0.00
The initial stocking cost for the fishery (£ '000)			0.00
Expected income generated by the extensive fishery			
Number of anglers expected at the fishery (anglers/d)			0
Angling days per year (one closed week per year)			358
Total number of angling days provided per year			0
Cost of a days angling from literature (£/angler)			£25
Cost of a days angling (£/angler)			£0
Gross value of angling provided (£ '000/y)			0.00
Catch limit for anglers (fish/d)			2
Fish required to maintain the stocking density (fish/y)			0
Cost of the fish required to maintain the stocking density (£ '000/y)			0.00
Net value of the angling provided (£ '000/y)			**0.00**

Financial considerations: variable and fixed construction and operating costs

1.Variable construction costs		Unit cost (£)	Total units	Total cost (£'000)
Land requirements and basin construction				
Estimated cost of land (ha)		7400		
Estimated cost for site development (ha)		49500		
Maximum flow rate through the treatment system (m³/h)			141.67	
Maximum area (m²) required for:	drum filter		50	
	settlement pond		0	
	constructed wetland		0	
	extensive aquaculture		0	
Maximum area required for basins (ha)			0.0	
Additional land required for access			5.0%	
Total land area required (ha)			0.0	
Total capital cost of land		7400	0.0	0.0
Basin construction		49500	0.0	0.26
Site development: pipework and infrastructure				
Electricity supply		500	1.00	0.50
Drum filters (@ 167 m³/h)		10000	1.00	10.00
Conical settling tank for 167 m³/h drumfilter		1500	1.00	1.50
Sludge storage tank for 167 m³/h drumfilter		1500	1.00	1.50
Sludge pump for 167 m³/h drumfilter		200	1.00	0.20
Concrete, brickwork and control structures (individual structures)		1000	2	2.00
Total cost of site infrastructure and pipework				15.70
Plant and equipment required				
Seine net and fish handling equipment		1500	na	0.00
Miscellaneous equipment		5	na	0.00
Total cost of equipment required				0.00
Seed required to establish fishery				0.00
Total cost of land, construction and infrastructure				16.00
2. Variable operational costs				
Annual electricity consumption				
Drum filter (per 167 m³/h unit)		200	1.00	0.20
Sludge pump (per 167 m³/h drumfilter)		50	1.00	0.05
Total electricity consumption				0.25
Labour requirements				
Drumfilter maintenance i.e. clean pre-filter and pressure wash screens		10.0	91.3	0.91
Part-time staff required to run a fishery >5 ha		10.0		0.00
Harvest and stocking fisheries <3 ha		10.0		0.00
General maintenance of constructed wetland		10.0		0.00
Harvest of constructed wetland (if fishery <3 ha)		10.0		0.00
Total labour requirement				0.91
Contract harvest of biomass from the reedbed (ha)		41	na	0.00
Stock for extensive aquaculture				0.00
Materials and equipment for infrastructure maintenance				0.16
Total variable operating costs				1.32
3. Fixed operating costs (excluding depreciation)				
Operators labour:	10 h + 3 h for each ADEPT component (h/y) (if fishery <3 ha)	20.0	13.00	0.26
Full-time manager:	for fisheries >3 ha	10.0	na	0.00
Total fixed operating costs				0.26

Costs and returns

1. Initial Costs and annual depreciation:

Cost	Cost (£ '000)	Economic life	Salvage value	Depreciation (£ '000/y)
Land	0.04	na	na	-
Basin construction	0.26	na	na	-
Seed/fish	0.00	na	na	-
Water control structures	2.00	15	0	0.13
Electricity supply	0.50	15	0	0.03
Drum filter, settlement and storage tanks	13.00	10	0	1.30
Pump	0.20	10	0	0.02
Seine nets	0.00	10	0	0.00
Total	16.00			1.49

2. Annual operating costs:

	Quantity	Unit price (£)	Cost (£ '000)
Variable costs			
Seed/fish required			0.00
Labour requirements (h)	91	10	0.91
Reed harvesting	0.0	41	0.00
Electricity			0.25
Maintenance			0.16
Sub-total			1.32
Fixed costs			
Tax			0.00
Operators labour			0.26
Fisheries manager (fishings >3ha)			0.00
Total labour cost			0.26
Sub-total			0.26
Total cost			1.58

3. Income:

	Production	Unit price (£)	Income (£ '000)
Angling (d/y)	0.00	25.00	0.00
Coarse fish (kg)	0.00	0.00	0.00
Crayfish (kg)	0.00	0.00	0.00
Macrophyte biomass (kg)	0	0.00	0.00
Total income			0.00

4. Indicators:

Profit including depreciation (£ '000)	-3.1
Profit excluding depreciation (£ '000)	-1.6
Rate of return on initial cost	-9.9%
Rate of return on operating cost	-100%
Pay-back period (y)	na

Cash flow and economic indicators

Cash flow projections for the first ten years of operation (£ '000)

	Year									
	0	1	2	3	4	5	6	7	8	9
Cash inflow										
Gross income (1)		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Operators equity	16.0									
Total cash inflow (2)	16.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cash outflow										
Investment cost (3)	16.0					0.0				
Operating cost (4)		1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Total cash outflow (6)	16.0	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Net cash flow (2-6)	0.0	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6
Cost/Benefit										
Financial benefit (1)		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cost (3+4) = (7)	16.0	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Net benefit (1-7)	-16.0	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6

Cash flow projections for years 10-19 of operation (£ '000)

	Year									
	10	11	12	13	14	15	16	17	18	19
Cash inflow										
Gross income (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Operators equity										
Total cash inflow (2)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cash outflow										
Investment cost (3)	13.2					2.5				
Operating cost (4)	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Total cash outflow (6)	14.8	1.6	1.6	1.6	1.6	4.1	1.6	1.6	1.6	1.6
Net cash flow (2-6)	-14.8	-1.6	-1.6	-1.6	-1.6	-4.1	-1.6	-1.6	-1.6	-1.6
Cost/Benefit										
Financial benefit (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cost (3+4) = (7)	14.8	1.6	1.6	1.6	1.6	4.1	1.6	1.6	1.6	1.6
Net benefit (1-7)	-14.8	-1.6	-1.6	-1.6	-1.6	-4.1	-1.6	-1.6	-1.6	-1.6

Cash flow projections for years 20-29 of operation (£ '000)

	Year									
	20	21	22	23	24	25	26	27	28	29
Cash inflow										
Gross income (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Operators equity										
Total cash inflow (2)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cash outflow										
Investment cost (3)	13.2					0.0				
Operating cost (4)	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Total cash outflow (6)	14.8	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Net cash flow (2-6)	-14.8	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6
Cost/Benefit										
Financial benefit (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cost (3+4) = (7)	14.8	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Net benefit (1-7)	-14.8	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6

Cash flow projections for years 30-39 of operation (£ '000)

	Year									
	30	31	32	33	34	35	36	37	38	39
Cash inflow										
Gross income (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Operators equity										
Total cash inflow (2)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cash outflow										
Investment cost (3)	13.2					0.0				
Operating cost (4)	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Total cash outflow (6)	14.8	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Net cash flow (2-6)	-14.8	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6
Cost/Benefit										
Financial benefit (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cost (3+4) = (7)	14.8	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Net benefit (1-7)	-14.8	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6

Cash flow projections for years 40-49 of operation (£ '000)

	Year									
	40	41	42	43	44	45	46	47	48	49
Cash inflow										
Gross income (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Operators equity										
Total cash inflow (2)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cash outflow										
Investment cost (3)	13.2					2.5				
Operating cost (4)	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6	1.6
Total cash outflow (6)	14.8	1.6	1.6	1.6	1.6	4.1	1.6	1.6	1.6	1.6
Net cash flow (2-6)	-14.8	-1.6	-1.6	-1.6	-1.6	-4.1	-1.6	-1.6	-1.6	-1.6
Cost/Benefit										
Financial benefit (1)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cost (3+4) = (7)	14.8	1.6	1.6	1.6	1.6	4.1	1.6	1.6	1.6	1.6
Net benefit (1-7)	-14.8	-1.6	-1.6	-1.6	-1.6	-4.1	-1.6	-1.6	-1.6	-1.6

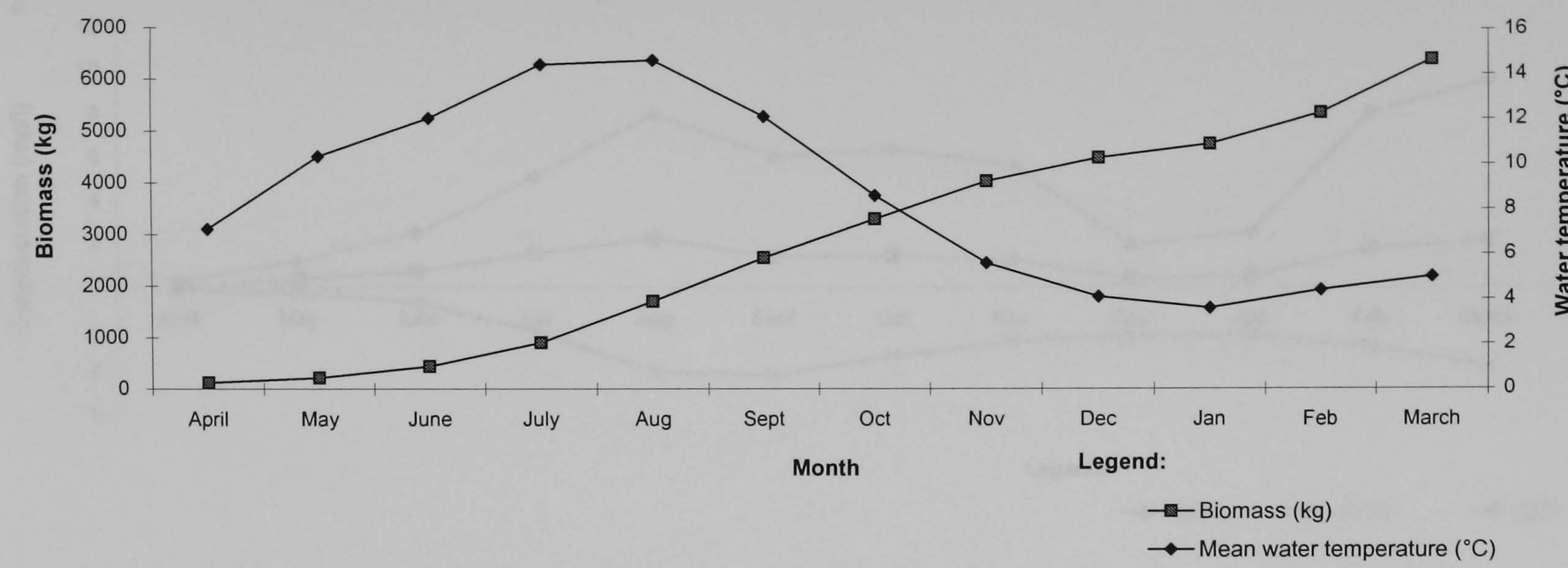

Figure 1a. Stock model (kg) and mean temperature (°C) of abstracted water

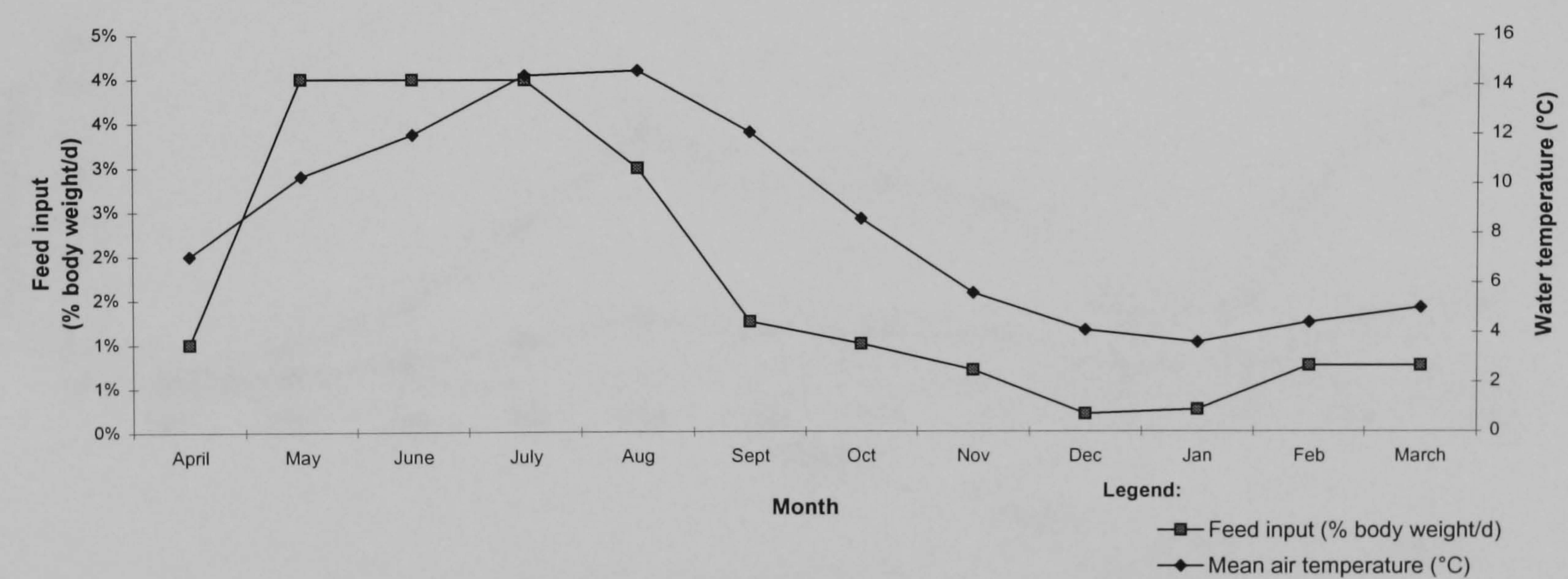

Figure 1b. Feed input (% body weight/d) and mean temperature (°C) of abstracted water

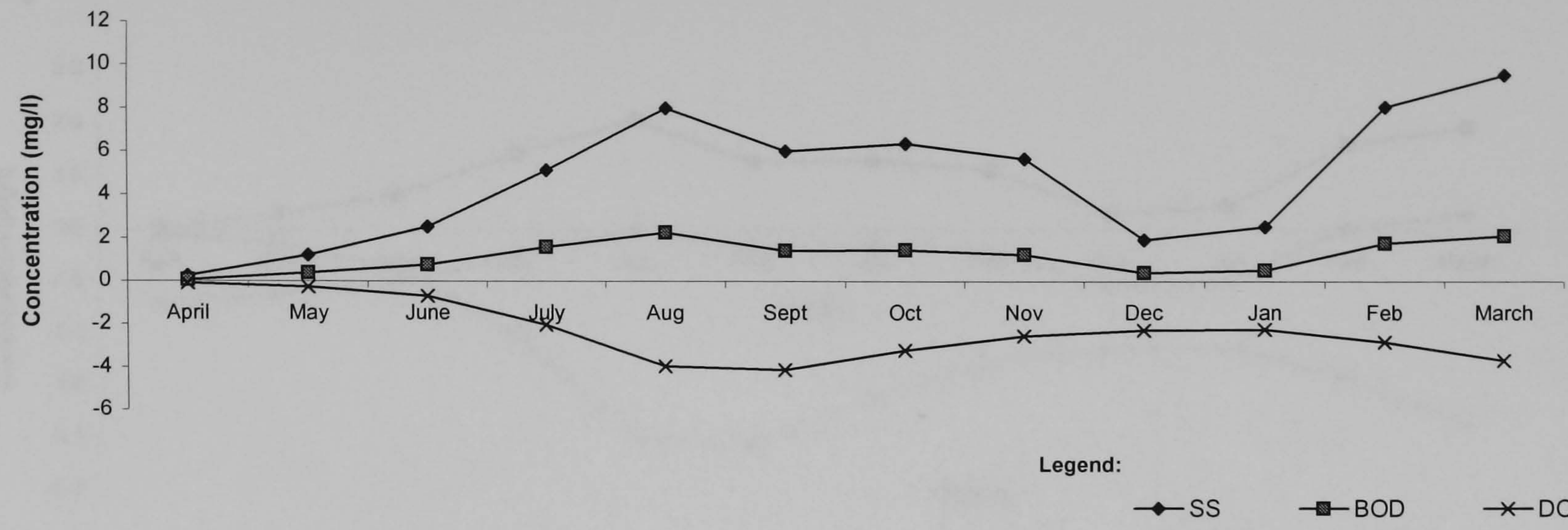

Figure 2a. Mean change in concentration (mg/l) of SS, BOD and DO in smolt unit wastewater

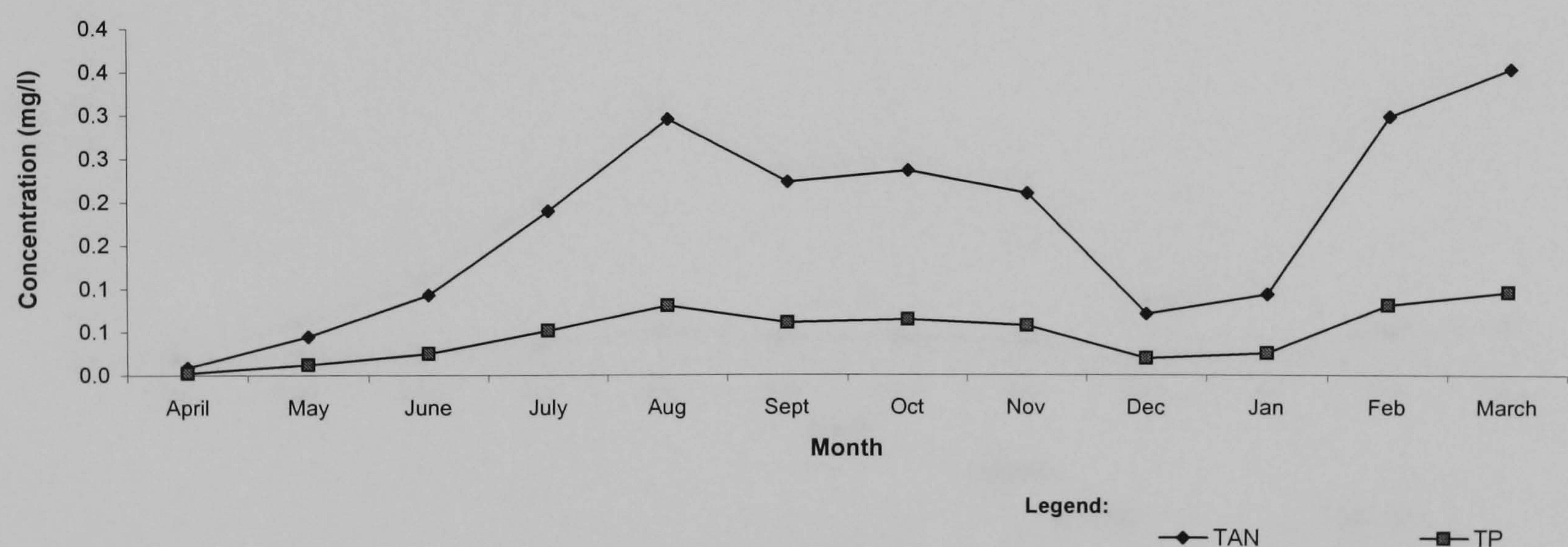

Figure 2b. Mean change in concentration (mg/l) of TAN and TP in smolt unit wastewater

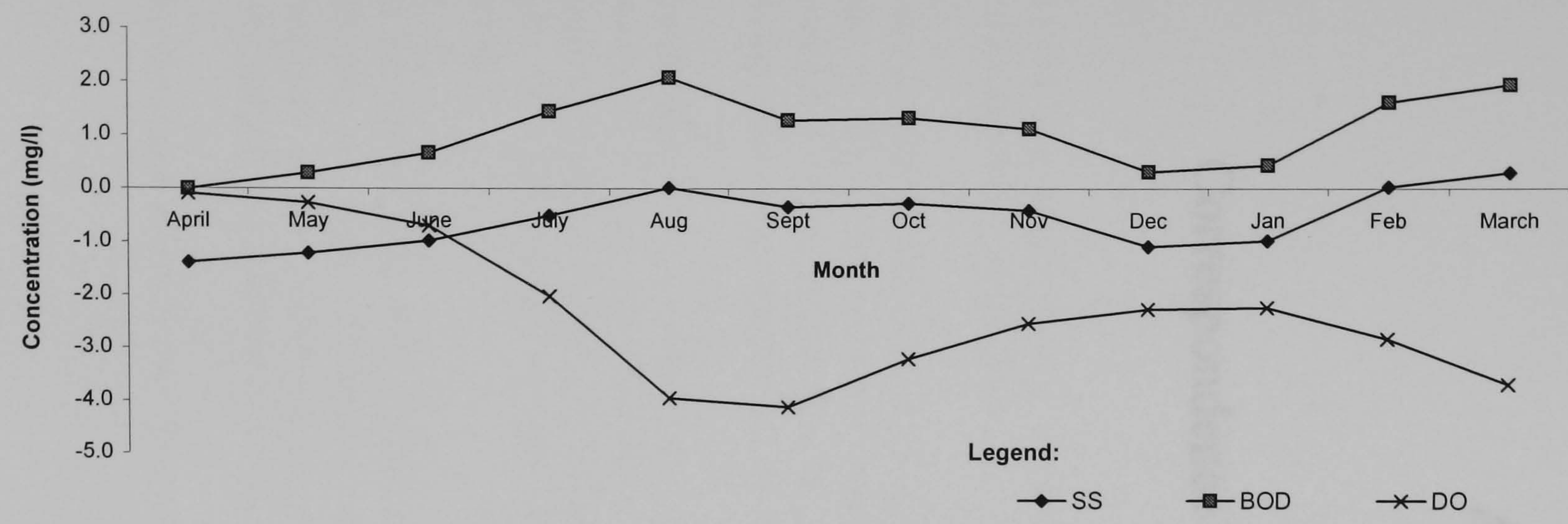

Figure 3a. Mean SS, BOD and DO concentrations (mg/l) in water treated using a drumfilter (minus background levels)

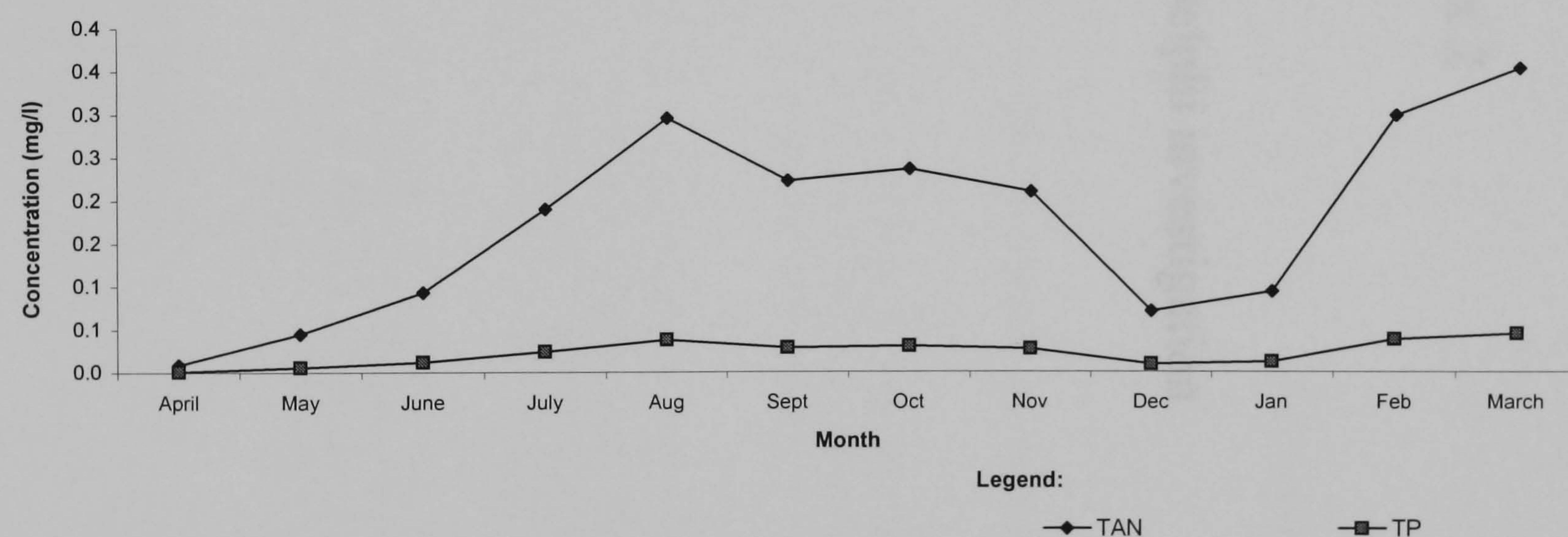

Figure 3b. Mean TAN and TP concentrations (mg/l) in water treated using a drumfilter (minus background levels)

Appendix 2

Correspondence for the Delphi investigation

Appendix 2a

First round questionnaire

Institute of Aquaculture
University of Stirling
Stirling FK9 4LA
Scotland
Fax: 01786 472133
email: swb1@stir.ac.uk

4th September 1999

Dear

I would like to invite you to participate in this investigation regarding the management of wastewater from aquaculture.

There are two sections to this questionnaire; **Section 1** asks for your opinion on the current status of aquaculture wastewater treatment and management. **Section 2** represents the first round of a Delphi investigation into the potential of horizontally integrated aquaculture in treating wastewater from commercial aquaculture.

I would ask everyone who wishes to participate to complete **Section 1;** but to only complete **Section 2** if they are willing to participate in the Delphi investigation. This will entail completing up to 3 more questionnaires over the following 2-3 months. Following a pilot-study, it is anticipated that the first questionnaire should take approximately 15 minutes to answer.

The results of this study will be distributed to all participants helping focus interest and resources on high potential systems, and that the publication of results will help raise awareness of horizontally integrated aquaculture, including the real world constraints to developing these systems and their potential.

Your time and effort in carrying out this research is much appreciated.

Yours sincerely,

Stuart Bunting

Horizontally integrated aquaculture:

For the purposes of this questionnaire the term 'horizontally integrated aquaculture' refers to the culture of aquatic species in the wastewater from commercial aquaculture, reducing the concentration of pollutants and potentially conferring benefits to the operator, environment and stakeholder groups.

Examples include the culture of seaweed and shellfish in the wastewater from shrimp ponds and marine cage facilities and the use of constructed wetlands planted with reeds or mangroves to treat the wastewater from landbased freshwater of marine/brackish aquaculture, respectively.

Section 1

1.1. From the perspective of which country and aquaculture sector would you like to answer this questionnaire?

a. country .... b. aquaculture sector ....

1.2. In general, what impact does the discharge of wastewater from the aquaculture sector selected above have on the environment?

a. very negative

b. negative

c. none

d. positive

e. very positive

1.3. There is a need to develop improved technologies to treat the wastewater discharged from aquaculture sector selected above.

a. agree strongly

b. agree

c. neither agree nor disagree

d. disagree

e. disagree strongly

1.4. Do you consider the regulations governing the discharge of wastewater from the aquaculture sector selected above to be:

a. very excessive

b. excessive

c. correct

d. inadequate

e. very inadequate

f. no regulations exist

1.5. Do you believe that the wastewater from the aquaculture sector selected above could be utilised for production in horizontally integrated systems?

a. yes

b. unsure

c. no

1.6. Do you believe that any economic, social or environmental benefits could be gained from developing horizontally integrated aquaculture systems in association with the aquaculture sector selected above?

a. yes

b. unsure

c. no

1.7. How would you describe your current position?

a. research

b. commercial

c. regulatory

d. stakeholder

e. other (please specify)

If you would like to participate in the Delphi investigation please proceed to **Section 2**; alternatively, please return your answers to **Section 1** by the **30th of September**.

Section 2: Delphi investigation

The Delphi technique is employed here in an attempt to develop consensus amongst stakeholders with an interest in wastewater discharged from aquaculture, i.e. regulators, commercial operators, researchers, environmental groups, etc. The survey is conducted over three or four rounds, during the first round all participants are asked to express their individual opinions; these opinions then form the basis of the subsequent rounds.

The research aims of the Delphi investigation are to:

- identify the factors that a panel of experts believe have the greatest potential to influence the decision-making process when considering horizontally integrated aquaculture to treat and utilise the wastewater from commercial aquaculture

- highlight the relative importance of horizontally integrated aquaculture with respect to alternative strategies for treating wastewater from commercial aquaculture

Answering the questionnaire

Based on your experience of commercial aquaculture, please consider the following questionnaire.

- In parts **1** and **2**, please state any factors that you consider as potential constraints or benefits, respectively, to developing horizontally integrated aquaculture. Please try and state the factors succinctly under the appropriate category i.e. physical, environmental, management, economic, regulatory, institutional and social.

- In part **3** please suggest any alternative strategies that you feel are appropriate to limiting the environmental impact of wastewater from aquaculture.

If you encounter any problems in answering this questionnaire please contact me for more information. Please return the completed questionnaire by the **30th of September.**

Delphi questionnaire

1. Please indicate any potential **constraints** or **problems** to developing horizontally integrated aquaculture with commercial aquaculture.

System features	Factors
Physical	
Environmental	
Managerial	
Institutional	
Economic	
Social	

2. Please indicate any potential **benefits** or **opportunities** to developing a horizontally integrated aquaculture with commercial aquaculture.

System features	Factors
Physical	
Environmental	
Managerial	
Institutional	
Economic	
Social	

3. Please indicate other factors that could help commercial aquaculture limit any negative environmental impacts associated with the discharge of wastewater.

Strategic area	Factors
Management	
Technology	
Institutional	
Economic/ Social	

Thank you for participating in the first round of this Delphi investigation. Please return your completed questionnaire by the **30th of September**. Once the results have been collated, I will return a summary of the responses received during this round and the second questionnaire.

Appendix 2b

Second round questionnaire

Dear

Having compiled the responses of participants from the first round, the next stage of the investigation is to assign relative values to the importance of the wide range of factors suggested. Below you will find a list of the responses from the first round together with an indication of the number of respondents (*n*) suggesting each factor. Please now assign a score between 1 and 10 relating to the importance that you assign to each factor (1 = not important, 10 = very important).

Your continued support in this work is appreciated.

Yours sincerely,

Stuart Bunting

PS: Please try and reply to this round by the 31st of November.

a. Constraints associated with horizontally integrated aquaculture

	n	Score
Physical		
Facility designed primarily for original aquaculture species	1	
Availability of suitable land or water	11	
Wastewater supply e.g. nutrient flows not optimal or presence of chemicals, antibiotics or pathogens	4	
Engineering of wastewater flows difficult e.g. high volumes and distance	2	
Biological treatment inconsistent at removing nutrients throughout the growing season	1	
Environmental		
Site specific design required e.g. climate vulnerability	4	
Discharge requirements; by-products from integrated systems may not comply with regulations	2	
Translocation of organisms suitable for integrated production restricted	1	
External pollution could affect the integrated system	1	
Managerial		
Lack of skills and training for existing managers and workers	4	
Managers lack leadership, resist change and adopt a non-systematic approach	5	
Decisions based on short-term financial appraisal	1	
Limited knowledge-base regarding design and management e.g. optimal loading rates, harvesting strategies, disease management	6	
Inadequate supplies of seed and inputs for integrated system	1	
Institutional		
Absence of instruction and training support	1	
Inertia i.e. a lack of research and development	1	
Constrained by existing paradigms of food production i.e. no structures exist to facilitate systems thinking	2	
Environmental laws and planning restrictions constrain integration	3	
Lack of structures for monitoring and auditing integrated systems	1	
Economic		
Market analysis and stimulation required	3	
Limited revenue generated by the integrated system	4	
Insufficient techniques for analysis of operating costs and accounting for broader issues such as opportunity costs	2	
Holders of economic power not interested	1	
Financial costs associated with development	8	
Lack of funding, access to venture capital	2	
Costs associated with management and operation	3	
Social		
Public acceptance of produce from integrated systems, perception of "dirty" food	4	
Conflicts with stakeholder groups e.g. user groups or environmentalists	4	
Limited systems thinking education and information exchange	1	
Total	84	
Count	29	

b. Opportunities associated with horizontally integrated aquaculture

	n	Score
Physical		
Better transport around remote areas	1	
Improved efficiency of resource use e.g. nutrients and water	7	
More attractive in the landscape than conventional treatment systems	2	
Environmental		
Reduced sediment and nutrient concentration in the wastewater	8	
Less energy consumed and waste generated in food production	1	
Increased habitat diversity, providing shelter to endemic species	3	
Reduced impact on the environment and downstream users e.g. other farms	8	
Managerial		
Meet expectations of managers regarding environmental protection and waste management	2	
Increase opportunities for recirculation	1	
Facilitate the development of a new paradigm for aquaculture and other agricultural sectors	2	
Contribute to the skill base of managers	1	
Reduce the potential for farms to self-pollute	2	
Institutional		
Improved discharge standards that reduce penalties e.g. court action or closure	2	
New opportunities to develop commercial partnerships	2	
Potential site for research and development	2	
Enabling more stringent discharge standards to be satisfied in the future	6	
Help meet the standards for quality assurance or organic certification	1	
Economic		
Reduce wastewater treatment costs	1	
Increased income generated from additional crops based on same inputs	12	
Represents a diversification reducing economic risks	5	
Reduce the level of any potential pollution tax	6	
Contribute to the regional economy, adding to tax base	2	
Social		
Improved public perception; reconciling environmental and economic goals of different groups	6	
Represents a good educational resource	1	
Increased employment opportunities	5	
Enhanced appeal of the primary aquaculture product to consumers	5	
Better places to live	1	
Total	94	
Count	26	

c. Strategies for reducing negative impacts associated with the discharge of wastewater

	n	Score
Institutional		
Better education of farmers regarding water quality and environmental management	3	
Increase and enforce discharge standards for wastewater or implement a pollution tax	3	
Open some commercial operations for public tours	1	
Encourage collaboration between researchers and commercial enterprises	3	
Provide information and direction to government regarding opportunities for the innovative management of aquaculture wastewater	2	
Managerial		
Adoption of a more holistic/systematic paradigm for aquaculture	2	
Good planning prior to developing aquaculture facilities, improved site selection	3	
Use management procedures that improve water quality e.g. careful feed management, de-sludging of lagoons, aeration and harvesting strategies that minimise discharges	5	
Adopt extensive as opposed to intensive management practices	1	
SocioEconomic		
Educate the public and managers regarding recycling systems	1	
Look at the energy costs of typical intensive production systems	1	
Government funding e.g. subsidies, grants and tax relief to encourage research and development	2	
Improved evaluation of benefits associated with management practices that reduce environmental impact	2	
The need to portray a positive image will necessitate improved waste management	1	
Technological		
Increase research and development into improved treatment technologies	7	
Develop systems for water recirculation	6	
Improved feed quality .e.g. improve feed conversion ratios	5	
Develop new vaccines and improve disease control	3	
Total	51	
Count	18	

Appendix 2c

Third round information

Dear

Firstly, thank you for your patience and responses during the second round.

Having compiled responses from the second round the final stage of the investigation is to see whether those participating generally agree on the mean values assigned to the factors being considered.

Below you will find the list of factors first presented in the second round, in addition, the mean value (x) for scores assigned during round two are given together with the 25% and 75% percentiles to provide an indication of the distribution of responses suggested.

If you are willing to accept the mean value (x) then please leave the space for a new score blank, however, if you feel that the mean value is not appropriate please enter your preferred score in the space provided.

If your preferred score lies outside the 25% and 75% percentile values stated, would you please provide a short statement to justify your opinion?

I would like to thank you for your continued support and hope that you have found this exercise interesting.

Yours sincerely,

Stuart

PS: Although there is no date by which to submit you responses I would encourage you to respond as soon as possible so that the outcome of this study may be assessed and the findings disseminated to all participants.

Delphi third round

Please remember: 1 = not important, 10 = very important

a. Constraints associated with horizontally integrated aquaculture

	x	25%	75%	Score
Physical				
Facility designed primarily for original aquaculture species	4.6	3.0	5.0	
Availability of suitable land or water	6.9	5.0	9.5	
Wastewater supply e.g. nutrient flows not optimal or presence of chemicals, antibiotics or pathogens	6.8	5.0	8.5	
Engineering of wastewater flows difficult e.g. high volumes and distance	5.2	3.0	7.0	
Biological treatment inconsistent at removing nutrients throughout the growing season	5.6	5.0	6.5	
Environmental				
Site specific design required e.g. climate vulnerability	5.7	4.0	7.8	
Discharge requirements; by-products from integrated systems may not comply with regulations	6.1	5.0	8.0	
Translocation of organisms suitable for integrated production restricted	4.3	2.3	6.0	
External pollution could affect the integrated system	5.4	4.0	6.8	
Managerial				
Lack of skills and training for existing managers and workers	5.6	3.5	8.0	
Managers lack leadership, resist change and adopt a non-systematic approach	6.2	4.0	8.0	
Decisions based on short-term financial appraisal	6.8	5.0	8.0	
Limited knowledge-base regarding design and management e.g. optimal loading rates, harvesting strategies, disease management	6.7	5.5	8.0	
Inadequate supplies of seed and inputs for integrated system	4.7	2.5	6.5	
Institutional				
Absence of instruction and training support	5.4	3.0	7.0	
Inertia i.e. a lack of research and development	5.9	4.3	7.8	
Constrained by existing paradigms of food production i.e. no structures exist to facilitate systems thinking	5.5	3.5	8.0	
Environmental laws and planning restrictions constrain integration	5.9	5.0	7.8	
Lack of structures for monitoring and auditing integrated systems	5.2	3.0	7.0	
Economic				
Market analysis and stimulation required	6.5	4.5	8.0	
Limited revenue generated by the integrated system	4.6	3.0	6.0	
Insufficient techniques for analysis of operating costs and accounting for broader issues such as opportunity costs	4.9	3.0	6.8	
Holders of economic power not interested	5.3	2.0	8.0	
Financial costs associated with development	7.2	6.0	8.5	
Lack of funding, access to venture capital	6.9	5.0	9.0	
Costs associated with management and operation	5.8	5.0	7.5	
Social				
Public acceptance of produce from integrated systems, perception of "dirty" food	5.7	4.0	8.0	
Conflicts with stakeholder groups e.g. user groups or environmentalists	5.5	3.5	7.5	
Limited systems thinking education and information exchange	5.5	4.0	7.0	
Total	167			
Count	29			

b. Opportunities associated with horizontally integrated aquaculture

	x	25%	75%	Score
Physical				
Better transport around remote areas	3.8	2.0	5.5	
Improved efficiency of resource use e.g. nutrients and water	8.4	8.0	9.5	
More attractive in the landscape than conventional treatment systems	6.4	5.0	8.0	
Environmental				
Reduced sediment and nutrient concentration in the wastewater	8.3	8.0	9.0	
Less energy consumed and waste generated in food production	8.2	7.5	9.0	
Increased habitat diversity, providing shelter to endemic species	6.3	5.0	8.0	
Reduced impact on the environment and downstream users e.g. other farms	8.4	7.0	9.5	
Managerial				
Meet expectations of managers regarding environmental protection and waste management	6.9	5.5	8.0	
Increase opportunities for recirculation	6.8	6.0	8.0	
Facilitate the development of a new paradigm for aquaculture and other agricultural sectors	6.6	5.0	7.8	
Contribute to the skill base of managers	5.6	4.0	7.0	
Reduce the potential for farms to self-pollute	6.9	6.0	8.0	
Institutional				
Improved discharge standards that reduce penalties e.g. court action or closure	6.0	5.0	7.0	
New opportunities to develop commercial partnerships	5.9	4.5	8.0	
Potential site for research and development	6.3	5.0	8.0	
Enabling more stringent discharge standards to be satisfied in the future	6.6	5.0	8.0	
Help meet the standards for quality assurance or organic certification	7.2	6.0	9.0	
Economic				
Reduce wastewater treatment costs	6.9	5.0	8.0	
Increased income generated from additional crops based on same inputs	7.3	6.0	8.8	
Represents a diversification reducing economic risks	6.4	5.0	8.5	
Reduce the level of any potential pollution tax	6.1	5.0	8.0	
Contribute to the regional economy, adding to tax base	5.2	3.0	7.5	
Social				
Improved public perception; reconciling environmental and economic goals of different groups	7.6	6.5	8.5	
Represents a good educational resource	6.2	4.5	8.5	
Increased employment opportunities	6.4	5.0	8.0	
Enhanced appeal of the primary aquaculture product to consumers	6.1	4.5	8.0	
Better places to live	5.2	2.0	7.5	
Total	178			
Count	27			

c. Strategies for reducing negative impacts associated with the discharge of wastewater

	x	25%	75%	Score
Institutional				
Better education of farmers regarding water quality and environmental management	7.0	5.5	8.0	
Increase and enforce discharge standards for wastewater or implement a pollution tax	6.5	5.5	8.0	
Open some commercial operations for public tours	5.6	4.5	8.0	
Encourage collaboration between researchers and commercial enterprises	7.5	6.5	9.0	
Provide information and direction to government regarding opportunities for the innovative management of aquaculture wastewater	6.8	5.0	8.5	
Managerial				
Adoption of a more holistic/systematic paradigm for aquaculture	6.3	5.0	8.0	
Good planning prior to developing aquaculture facilities, improved site selection	7.6	6.0	10.0	
Use management procedures that improve water quality e.g. careful feed management, de-sludging of lagoons, aeration and harvesting strategies that minimise discharges	8.0	8.0	9.0	
Adopt extensive as opposed to intensive management practices	4.1	2.0	5.0	
SocioEconomic				
Educate the public and managers regarding recycling systems	6.3	4.5	8.5	
Look at the energy costs of typical intensive production systems	6.4	5.0	8.0	
Government funding e.g. subsidies, grants and tax relief to encourage research and development	5.9	5.0	7.0	
Improved evaluation of benefits associated with management practices that reduce environmental impact	6.9	6.0	8.0	
The need to portray a positive image will necessitate improved waste management	6.7	5.0	8.5	
Technological				
Increase research and development into improved treatment technologies	7.4	6.5	8.0	
Develop systems for water recirculation	6.3	5.0	8.0	
Improved feed quality .e.g. improve feed conversion ratios	7.2	6.5	9.0	
Develop new vaccines and improve disease control	5.1	2.0	7.0	
Total	118			
Count	18			