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The Environmental Impact of Marine Fish Culture: Towards a Sustainable Future

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The environmental impact of marine fish-farming depends very much on species, culture method, stocking density, feed type, hydrography of the site and husbandry practices. In general, some 85% of phosphorus, 80–88% of carbon and 52–95% of nitrogen input into a marine fish culture system as feed may be lost into the environment through feed wastage, fish excretion, faeces production and respiration. Cleaning of fouled cages may also add an organic loading to the water, albeit periodically. Problems caused by high organic and nutrient loadings conflict with other uses of the coastal zone. The use of chemicals (therapeutants, vitamins and antifoulants) and the introduction of pathogens and new genetic strains have also raised environmental concerns.

Despite the high pollution loadings, results from various studies show that some 23% of C, 21% of N and 53% of P of feed input into the culture system is being accumulated in the bottom sediments and the significant impact is normally confined to within 1 km of the farm. The major impact is on the sea bottom, where high sediment oxygen demand, anoxic sediments, production of toxic gases and a decrease in benthic diversity may result. Decreases in dissolved oxygen and increases in nutrient levels in the water are also evident but are normally confined to the vicinity of the farm. Tributyltin (TBT) contamination and the development of antibiotic-resistant bacteria have been reported near fish farms. The stimulating effects of vitamins/fish wastes on growth of red tide species have been demonstrated in a number of laboratory studies. Nevertheless, there is no evidence to support the suggestion that the present use of therapeutants, vitamins and antibiotics and the introduction of pathogens and new genetic strains would pose a significant threat to the environment.

Marine fish culture can be a sustainable development, provided pollution loadings generated by fish farms are kept well below the carrying capacity of the water body. Effects can be significantly reduced by careful site selection, control of stock density, improved feed formulation and integrated culture (with macroalgae, filter-feeders and deposit-feeders). An example of the application of computer modelling in mariculture management is demonstrated. Environmental impact assessment and monitoring should also be carried out to ensure culture activities are environmentally sustainable.

Marine fish may be cultured in raceways, ponds or net cages in open waters. Salmonid fish culture predominates in western Europe, Scandinavia and North America, where artificial feed is commonly used. In Asian–Pacific waters (e.g. Japan, Korea, Hong Kong, Thailand, Malaysia and Singapore), a variety of non-salmonid species (e.g. groupers, sea breams, seabass, snappers and yellow tails) is commonly cultured and fed with trash fish. Marine fish culture in the coastal waters of many countries has grown dramatically in recent years, and further growth is expected in the coming decade (FAO, 1992). The rapid growth of marine fish-farming has already led to growing concerns over environmental impacts and conflicts with other coastal usage in Europe, North America, Australia and Asia (Hammond, 1987; Waldichuck, 1987a,b; Morton, 1989; Miki *et al*., 1992). Indeed, environmental concerns have led to a moratorium on new developments and tighter control in New Zealand, Denmark, Norway, Canada and Hong Kong (Duff, 1987; BC Ministry of Environment, 1990). In Scotland and Hong Kong, there is a general tendency to force marine fish-farming offshore (Aldridge, 1988).

This paper *1.* considers the pollutants generated from marine fish-farming and their environmental fates, *2.* reviews the impact of marine fish-farming on the marine environment, and *3.* examines practical mitigatory measures with a view to making marine fish-farming a sustainable development.

Sources, Kinds, Forms and Fates of Pollutants

High organic and nutrient loadings are mainly generated from feed wastage, fish excretion and faecal production. Most of our knowledge of feed wastage, organic and nutrient loadings and N excretion are, however, only based on salmonid fish cultured in land facilities fed with artificial feed (for a review, see Gowen & Bradbury, 1987; Handy & Poxton, 1993). Pollution loadings and effluent quality from land-based farms have been well documented (Alabaster, 1982), while those from open-sea cage farms and, particularly, those from non-salmonid species are poorly known (Handy & Poxton, 1993). Feed wastage is one of the most important sources contributing to organic and nutrient loadings (Ackefors & Enell, 1990; Seymour & Bergheim, 1991). Wastage may range from 1–38%, depending on the feed type, feed practices, culture method and species (Warren-Hansen, 1982; Ove Arup *et al.*, 1989; Thorpe *et al.*, 1990; Beveridge *et al.*, 1991). However, there are few published estimates of feed wastage and the limited available data are exclusive to salmonids cultured in ponds (Handy & Poxton, 1993). Deposition of organic waste was estimated at 3 kg m^{-2} yr^{-1} in the vicinity of a farm and 10 kg m^{-2} yr^{-1} or 1.8–31.3 kg C m^{-2} yr^{-1} underneath (Gowen & Bradbury, 1987). Fluxes and mass balances of C, P and N determined for a salmonid cage farm (rainbow trout fed with dry feed) indicated that 80% of C, 76% of N and 82% of P of feed input into the system were lost to the environment (Hall *et al.*, 1990; Holby & Hall, 1991; Hall *et al.*, 1992). The total environmental loss of N was estimated at between 95–102 kg N t^{-1} production (Hall *et al.*, 1992), and deposits from fish farms covered some 3.8 times the area of the farm itself (Hall *et al.*, 1990).

A comparison of feed wastage and pollution loadings from *1.* rainbow trout cultured in a raceway and fed with dry feed in Europe, *2.* salmonids cultured in ponds and fed with artificial feeds in Europe, and *3.* groupers and snappers cultured in open-sea cages and fed with trash fish in Hong Kong, is given in Figs 1 and 2. It is obvious that regardless of species and culture methods, marine fish-farming generates a very high pollution loading. Feed wastage and pollutant loadings are much higher in open-sea cage culture systems where trash fish is used as feed.

In a European salmonid farm using artificial feed, it was estimated that eventually some 80–84% of C, 52–95% of N and 82% of P in the feed may be lost to the environment (Table 1). Carbon may be lost into the environment through feed wastage and fish respiration (in the form of CO_2). Nutrients (N and P) may be lost through excretion (mainly in the form of ammonical N and urea), feed wastage and faecal production (mainly in organic forms).

Despite high organic and nutrient loadings deposited on the bottom sediment, only 10% of organic matter in the sediment underneath salmonid farms is broken down annually (Aure & Stigebrandt, 1990), and decomposition of farm deposits has been shown to be inversely related to net accumulation (Hansen *et al.*, 1990). Sediment fluxes of C, N and P are very low, and 79% of C, 88% of N and 95% of P in farm deposits (equivalent to 23% of C, 21% of N and 53% of P of the feed input) will be accumulated in the sediment and become unavailable (Table 1). Handy & Paxton (1993) also estimated that 59–66% of P in the feed is accumulated in the sediment. Phosphorus can be recycled to water by desorption and biological processes but the release becomes insignificant when the deposit is >7 cm thick (Hansen *et al.*, 1990). Since N mineralization mainly occurs in oxic surface sediments, the rate would be very slow in deposits underneath farms where the sediment is normally anaerobic and without bioturbation and epifaunal reworking (Rublee, 1982; Huettel, 1990). Indeed, Kaspar *et al.* (1988) failed to detect in-situ nitrification in sediment directly under a salmonid farm.

Vitamins (e.g. B_{12} and biotin), antibiotics (e.g. aureomycin, oxytetracycline, terramycin, furazolidone and nitrofurazone) and pigments are often added to artificial feed but such additions are not common in the tropics and sub-tropics where trash fish is predominantly used as a feed. Therapeutants (e.g. malachite green, formalin, copper sulphate and dipterex) are commonly used to treat fish diseases. Toxic chemicals (metals and sometimes TBT) are often used to treat cage netting to control fouling (Davies & McKie, 1987; Thrower & Short, 1991), although their use has been banned in many countries since 1990. Alabaster (1982) estimated that the use of therapeutants (aureomycin, oxytetracycline, terramycin, furazolidone and nitrofurazone) was between 70–2000 mg kg^{-1} feed. In Norway, the use of therapeutants for fish-farming (nitrofurazolidone, oxytetracycline, oxolinic acid, sulphamerazine and trimethoprim-sulphadiazine) has increased from 3.7 t in 1980 to 32.6 t in 1988 (Grave *et al.*, 1990); and it was estimated that some 430 g of antibiotics was used in producing 1 t of salmon (Rosenthal *et al.*, 1988).

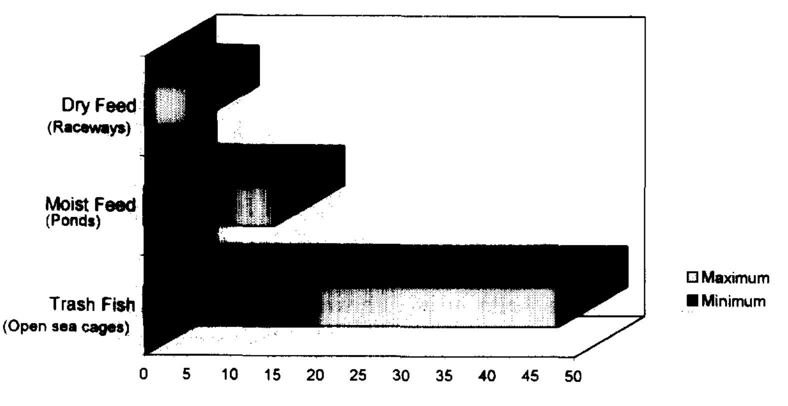

Fig. 1 Feed wastage resulting from the use of various feed types. Culture methods of each study are shown in parentheses. ([a]Ove Arup *et al.*, 1989; [b]Warren-Hansen, 1982.)

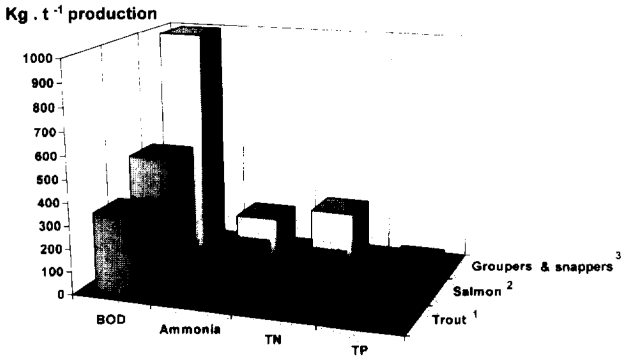

Fig. 2. Pollution loadings resulting from various types of culture and species (TN: total N; TP: total P; data from: [1]Warren-Hansen, 1982; [2]Alabaster, 1982; Handy & Poxton, 1993; [3]Ove Arup *et al.*, 1989).

Statistics on the use of chemical therapeutants in other countries is not available, but the quantity used is expected to be large (Chua, 1993). The quantities of therapeutic chemicals released into the environment, however, remain virtually unknown.

The P and N loading generated from fish-farming corresponds to a negligible fraction (0.6% and 0.2%, respectively) of overall loadings on coastal areas of Sweden, but local effects may be significant (Ackefors & Enell, 1990). Similarly, the impact of marine fish-farming was considered low compared with other waste generating activities along the west coast of Canada (BC Ministry of Environment, 1990). In Hong Kong, where open-cage culture is practised and trash fish is used, BOD and N, generated by the mariculture industry, constituted about 3% of total loading discharged into Hong Kong waters (Ove Arup *et al.*, 1989). The total discharge of P from a farm with a production of 50 t yr^{-1} would correspond to treated discharge from 7000 people, assuming 90% of P is removed from the discharge (Holby & Hall, 1991). However, it should be noted that fish-farm waste is not directly comparable to domestic sewage, mainly because of different C:N:P ratios and significant differences in settleable and soluble wastes (Rosenthal *et al.*, 1988).

Disposal of the fouling biomass on net cages into the water may occasionally add a high pollution loading to the environment. For example, the fouling biomass on fish cages in Hong Kong was estimated at 1.78 t (wet biomass) t^{-1} fish production yr^{-1} (equivalent to 31 kg BOD, 7.5 kg N and 70 g P) (Mak, 1982; Ove Arup *et al.*, 1989). Although it may be argued that nutrients and organic matter in the fouling biomass may be derived from, and represent part of, the nutrients released from fish farms, disposal of fouling organisms during cage cleaning will add a significant organic and nutrient input within a relatively short period and may cause problems.

Environmental Impacts

The environmental impact of fish-farming depends very much on species, culture method, hydrography of culture site, feed type and husbandry practices. Almost all studies on the environmental impact of marine fish-farming have been carried out in temperate regions where salmonid fish were cultured in ponds/cages and fed with artificial feed (Gowen & Bradbury, 1987). Only a single study has been carried out to assess the environmental impact of open-water cage farming in sub-tropical conditions where non-salmonid fish were

TABLE 1

Major forms and estimated loss of C, N and P to various environmental compartments and sediment flux in salmonid fish farms (data expressed as % of total feed input into the system).

	1 Fish harvest	2 Loss to water	3 Loss to sediment	4 Loss to environment (2+3)	5 Sediment flux	6 Net sediment accumulation
C						
Major form		CO_2 from fish respiration	Organic C from feed wastage and faecal production		CH_4, DOC, CO_2	Organic C
%	20*	51*	29*	80*	6*	23*
	16†	40†	44†	84†	–	–
N						
Major form		Ammonia and urea from fish excreta	Organic N from feed wastage and faecal production		DON, ammonia nitrate and urea	Organic N
%	20†	52†	28†	80†	3‡	21‡
	24‡	51‡	25‡	76‡	–	–
	–	14–50	14–81‖	52–95‖		
P						
Major form		Particulate and dissolved P	Organic P from feed wastage and faecal production		Phosphate	Organic P
%	16§	28§	56§	82§	3§	53§
						59–66§¶

* Hall *et al.* (1990), estimated by the flux method.
† Gowen & Bradbury (1987).
‡ Hall *et al.* (1992).
‖ Handy & Poxton (1993).
§ Holby & Hall (1991).
¶ Enell (1987).

cultured and fed with trash fish (Wu *et al.*, 1994). The general picture emerging from the existing studies indicates that the major impact is on the sea bed and, to a lesser extent, on water quality.

Organic matter settled on the sea bed may lead to the development of anoxic and reducing conditions in the sediment and the production of toxic gases (e.g. ammonia, methane and hydrogen sulphide). Sediment oxygen demand (SOD) of bottom sediments enriched by fish-farming activities can be some two to five times higher than that of the controls (Wu, 1990a; Wu *et al.*, 1994), while total sediment metabolism has been reported as being 10 times higher than that of the control (Holmer & Kristensen, 1992). Changes in macrobenthic community structure along a farm waste-enriched gradient have been documented. An azoic zone was typically found underneath the cages and a decrease in benthic diversity occurred in the vicinity of the farm (Ritz *et al.*, 1989; Tsutsumi *et al.*, 1991; Wu *et al.*, 1994). Benthic assemblages were normal between 25 to 150 m away from the cages in which salmonids were fed with artificial feed (Brown *et al.*, 1987; Weston, 1990) while the affected area may extend to 1 km where trash fish are used and flushing is poor (Wu *et al.*, 1994).

Carbon dioxide produced by fish respiration is unlikely to create any adverse effect in the marine environment. Unionized ammonia is acutely toxic to marine life and toxicity is dependent upon salinity, temperature and pH, while nitrate and nitrite are not significantly toxic to fish, except in the context of promoting algal blooms. Phosphorus is not important in promoting algal growth in the marine environment and, therefore, unlikely to have a significant effect (for a review, see Handy & Poxton, 1993).

A decrease in dissolved oxygen and increases in BOD, nutrients (P, organic and inorganic N and total C) have been generally found in the water column around fish farms (Muller & Varadi, 1980; Bergheim *et al.*, 1982; Beveridge & Muir, 1982; Enell, 1982, 1987; Penczak *et al.*, 1982; Enell & Lof, 1983; Beveridge, 1985; Phillips & Beveridge 1986; Molver *et al.*, 1988). Dissolved oxygen values returned to normal 30 m away from salmonid farms (Gowen & Bradbury, 1987) but an oxygen sag may extend to 1 km where trash fish is used and culture conditions are poor (Wu *et al.*, 1994). Changes in suspended solids, light extinction coefficient, chlorophyll-*a* and phaeopigment were considered to be either insignificant or localized (Beveridge *et al.*, 1994; Wu *et al.*, 1994).

Nitrogen is considered to be the limiting nutrient for primary production in coastal areas (Gundersen, 1981). Ammonia and urea excreted by fish can be readily taken up by phytoplankton and hence may stimulate their growth. It should be noted that fish excreta and waste food have a N:P ratio close to 7:1 w/w (the Redfield ratio) (Aure & Stigebrandt, 1990), and hence provide well-balanced nutrients for phytoplankton requirement. However, it appears unlikely that marine fish-farming may cause eutrophication on a large scale, although the possibility of localized eutrophication occurring in areas of poor flushing cannot be excluded (Gowen & Bradbury, 1987; Aure & Stigebrandt, 1990; Wu *et al.*, 1994).

The environmental effects of pigments and vitamins are poorly known. Biotin has been shown to stimulate growth of certain phytoplankton species and is implicated in the toxicity of the dinoflagellate *Gymnodinium aureoles* (Gowen & Bradbury, 1987). Vitamin B_{12} has been shown to be one of the growth-promoting factors of the alga *Chrysochromulina polylepis* (which caused massive kill of caged culture salmon in Scandinavian waters) and the dinoflagellate *Heterosigma akashiwo* (Graneli *et al.*, 1993; Honjo, 1993). Fish meat and faeces have been shown to stimulate the growth of the red tide species *Gymnodinium* type 65 and *Chatonella antigua* in laboratory culture (Nishimura, 1982). Despite these laboratory results, there is no good scientific evidence to relate the field occurrence of red tides to fish-farm wastes.

The use of antibiotics in fish farms may lead to the development of resistance in bacterial pathogens of fish, and the possibility of transfer of resistance to human pathogens has also raised concern (Aoki, 1989; Dixon, 1991). The development of a resistant bacterial population in the sediment has been documented (e.g. Austin, 1985; Homer, 1992). For example, up to 100% of oxytetracycline-resistant bacteria have been recorded from marine sediment near fish farms after medication; and resistance persisted for more than 13 months afterwards (Torsvik *et al.*, 1988; Samuelsen *et al.*, 1992). On the other hand, furazolidone can be rapidly degraded by microbes and hence antibacterial activity was not detectable in sediments (Torsvik *et al.*, 1988; Samuelsen *et al.*, 1991). Inhibition of sulphate reduction in sediment underneath cages after antibiotic treatment has been reported (IOE, 1992). The effects of vitamins on the marine environment are still not well known. In oxic environments, however, the half-lives of antibiotics and vitamins is short (e.g. <7 days in seawater and 32–64 days in fish-farm sediments for oxytetracycline and biotin) (Samuelsen, 1989), and the accumulation of vitamins and antibiotics in the environment is highly unlikely. Although oxytetracycline may be very persistent in anoxic fish-farm sediments (up to 419 days), it is not biologically available in such cases (Bjoerklund *et al.*, 1990).

TBT contamination has been identified from the tissues of culture fish (Davies & McKie, 1987; Waldichuck, 1987b), and water (Balls, 1987; Thrower & Short, 1991) where TBT-treated net pens were used. However, no significant changes in mortalities and growth rates were observed in TBT-contaminated fish (Thrower and Short, 1991). Imposex has also been reported in dogwhelks from sea lochs in Scotland where TBT was used in treating fish cages (Davies *et al.*, 1987).

Despite growing concern regarding the spread of disease from farmed fish to wild stock (Hill, 1991), there have been very few documented examples. In most cases, disease identified in one population cannot be positively traced as having spread from another population (Brackett, 1991).

Culture species may be less adaptable to the natural environment, and escaped culture fish may inter-breed with the wild stock, thereby altering the gene pool of the latter. However, there is insufficient evidence to ascertain the ecological impact of escaped stock. It is conceivable that the introduced gene in wild stock might be eliminated by natural selection very quickly.

Coastal habitats (e.g. mangroves) are often destroyed for marine fish-farming, resulting in the losses of nursery and spawning grounds for marine animals. Conflicts with other uses of the coastal environment (e.g. navigation, recreation and fishing) and reductions in current speed have been well documented.

Towards a Sustainable Future

Keep stocking density and pollution loadings under environmental capacity

It has been demonstrated that environmental impacts of marine fish-culture vary considerably between sites and are highly dependent on water circulation, stocking density, husbandry practices and feed types (Wu *et al.*, 1994). At one site in Hong Kong with good flushing and low stocking density, benthos (and even corals) could be found underneath cages. The results from this study clearly indicated that marine fish culture can be a sustainable industry provided that stocking density does not exceed the carrying capacity of the water.

The carrying capacity of the water depends on tidal flushing, current and assimilative capacity of the water body to pollutants. Oxygen consumption of culture species ranges from 83 to >400 g O_2 t^{-1} h^{-1} (Wu, 1990b; McLean *et al.*, 1993). Assuming that dissolved oxygen in seawater is 7 mg O_2 l^{-1}, at least 17–57 m^3 of fresh seawater would be required to compensate for the oxygen consumption alone of 1 t of culture fish; not to mention the additional oxygen demand exerted by wastes from the farming activities. Indeed, water use in land-based rainbow trout farms was very high (210 000–252 000 m^3 t^{-1} production) (Beveridge *et al.*, 1994). In open-water cage culture systems, it has been suggested that an annual production of 200 t fish would require 1 m^3 s^{-1} of current flow (Tervet, 1981). In areas where water exchange is poor, the fish stock should be reduced.

Having quantified the values of organic and nutrient wastes produced per tonne of fish, and defined acceptable limits for water and sediment quality parameters to support marine life, the maximum permissible stock that the defined water/sediment quality should not exceed can be estimated by water quality modelling techniques. To this end, an attempt has been made to construct a water quality model to determine the carrying capacity of water in relation to culture stock at Three Fathoms Cove, Hong Kong (Wu *et al.*, in prep.). In summary, the model consists of *1.* a 2D, two-layer hydrodynamic model, to simulate tidal flows at a fish culture site, using conservation of mass equation for each layer, and *2.* a 3D (segmented and layered) tide-averaged water quality model, to allow for an assessment of organic and nutrient input on the quality of the receiving water. The hydrodynamic model calculates tidal flow and generates the necessary hydrodynamic input for the running of the water quality model. The water quality model quantifies the relationships between major biotic components (i.e. bacteria, phytoplankton, zooplankton, macroalgae, benthos and fish) and abiotic components (i.e. salinity, organic carbon, dissolved oxygen, nitrogen, phosphorous and oxygen) at a fish culture site, and also describes the prevailing major biological and chemical processes (including oxidation of organic carbon, nutrient and phytoplankton dynamics, hydrolysis and oxidation of organic and inorganic N, growth, photosynthesis and respiration and decay of plant carbon, SOD, fish respiration and BOD). Fish biomass and the resulting pollutants (organic waste and nutrients) generated from various activities (e.g. food wastage, fish faeces and excreta, etc.) were quantified and input into the model. Based on the input organic and nutrient loadings from a given stocking density, the model calculates the resulting levels of NH_3, NO_2 NO_3, total organic N, dissolved oxygen and BOD in the receiving water.

The results of the simulation are in good agreement with those from a field study of the same site reported upon earlier (Wu *et al.*, 1994). By comparing the output of water quality data under different scenarios of stocking density, the model can serve as an effective tool to help management decisions on the maximum fish stock permissible at a particular fish culture site so that acceptable water quality objectives can be met for the sustainable development of the industry.

Improved artificial feed formulation

About 80–84% of C, 52–95% of N and 82% of P in feed input to the culture system may be lost through feed wastages, excretion and faecal production (Table 1), and such losses can be minimized by improved feed technology.

Feed wastage can be reduced by increasing the stability and reducing the sinking rate of feed, and providing the fish with an optimal size of feed at different stages of development. Ammonia excretion by fish is a function of protein intake and can be kept to a minimum; with a highly digestible feed an optimal protein/energy ratio can be provided for each species and its developmental stages. The energy requirements of fish can be satisfied by carbohydrates and fat, so that protein can be spared for body tissue construction. It has been shown that protein retention in *Sparus aurata* can be increased from 24.3 to 31.3% by increasing the dietary lipid by 37% (Kissil & Lupatsch, 1992). Obviously, reduction of N in the diet can only be achieved if artificial feed is used. There is little doubt that formulated artificial feed is superior to trash fish, in terms of its nutritional value, storage, supply and pollution loading. The reason that trash fish is still widely used in Asia (e.g. Japan, Hong Kong, Singapore and Thailand) appears to be largely due to our poor understanding of the nutritional requirements of the various non-salmonid species cultured. This points to the urgent need for research into the nutritional requirements of these non-salmonid species.

More recently, attempts have been made to minimize feed usage by video monitoring of feed behaviour

(Guldberg *et al.*, 1993) or installation of feeding devices with a hydroacoustic sensor (Juell, 1991).

Harvest pollutants by integrated culture

Marine macroalgae can take up N, whereas filter-feeders (e.g. bivalves) can remove particulate and phytoplankton from water at remarkable rates. Harvesting nutrients generated by marine fish-farming by macroalgae and filter-feeders would be an attractive option since this would alleviate nutrient pollution on one hand and increase productivity on the other.

Uptake of ammonia, tissue N content and yield of the green alga *Ulva lactuca* is linear with respect to ambient concentration (Ho, 1993). It has been shown that 1 kg of *U. lactuca* can remove 90% of ammoniacal nitrogen in effluents produced by 75 kg of fish and give a maximum yield of 55 g m^{-2} d^{-1} dry wt in a land-based fish farm (Cohen & Neori, 1991; Neori *et al.*, 1991). Similarly, culturing brown macroalgae (*Laminaria* and *Macrocystis*) near fish farms for nutrient removal was considered to be both economically and technically feasible by Petrell *et al.* (1993). An increase in fish yield (by 1.5%) and dissolved oxygen (by 9%) have been reported from a Japanese open-cage culture system consisting of red sea bream and sea lettuce (Anon., 1994).

The filtering rates of the oyster *Crassostrea gigas* and the scallop *Patipectin yessoensis* may be up to 101 and 95 l d^{-1} individual^{-1}, respectively (Inui *et al.*, 1991). In-situ clearance rates of mussel beds ranged from 0.4 to 2.7 m^3 m^{-2} h^{-1} (Prins *et al.*, 1994). Small & Prins (1993) estimated that the uptake rates of suspended particulate matter (SPM), particulate organic carbon (POC) and particulate organic nitrogen (PON) by natural oyster/mussel beds were very high at 81–2800 g SPM, 4.6–186.5 g POC and 0.6–33.6 g PON m^{-2} d^{-1}, respectively. The use of mussel/oyster beds to control phytoplankton growth and eutrophication has been suggested by Cloern (1982) and Loo & Rosenberg (1989). Recently, Shpigel *et al.* (1993) demonstrated that some 63% of N in the feed can be harvested in an integrated land-based culture system composed of bivalves (*Crassostrea gigas* and *Tapes semidecussatus*) and seaweed (*Ulva lactuca*) (26% as fish yield, 14.5% as bivalve yield and 22.4% as seaweed yield). Compared with a 'monoculture' fish farm in which 80% of N input will be lost (Gowen & Bradbury, 1987), only 37% of N input was lost into the environment in this integrated culture system. Culturing shellfish to remove N and plankton production derived from fish-farm pollution appears to be a viable and practical option and should be adapted to open-cage culture systems. However, caution must be exercised to avoid bacterial contamination of shellfish from fish farms.

Mud consumption of many bottom deposit-feeders is high. For example, mud consumption by the sea cucumber ranged from 441 g to 3.5 kg individual^{-1} yr^{-1} (Ong Che, 1990; Inui *et al.*, 1991). The possibility of culturing bottom deposit-feeders (e.g. polychaetes, holothurians) to remove organic deposits should be explored. However, since bioturbation of sediment may lead to the release of organic matter and nutrients in the sediment, this option should be treated with caution.

Environmental impact assessment, monitoring and control

While environmental impact assessment (EIA) tends to become a standard requirement for major developmental projects nowadays, the application to marine fish-farming is much less common. Indeed, EIA helps to prevent conflict between coastal users, protects sensitive habitats and is important for sustainable development of the mariculture industry. Since nitrogen and organic wastes are major concerns, the susceptibility of the site to dissolved oxygen changes and nitrogen pollution should be given special attention in any EIA of marine fish-farming.

Effluent standards for land-based fish farms have been well established (e.g. Alabaster, 1982). It would, however, be much more difficult to set effluent standards and control effluent quality in open-water cage farming systems. For example, insufficient information is available on effluent toxicity in seawater to allow environmental standards for N to be set (Handy & Poxton, 1993). It appears that the most practical way to control pollution from open-cage farming would be to control the stocking density in relation to the carrying capacity of the receiving water. Regular monitoring of water and sediment quality at fish culture sites would also be required, and indeed, becomes mandatory in some countries (Hensey, 1992).

In summary, marine fish-farming can be a sustainable development, and may serve as an environmental probe to detect coastal pollution and protect marine life and non-culture species, if the industry is properly managed.

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