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Perspectives of nutrient emission from fish aquaculture in coastal waters

Literature review with evaluated state of knowledge

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Lasse M. Olsen[1], Marianne Holmer[2], Yngvar Olsen[1,3]

[1]Trondhjem Biological Station, Department of Biology, Norwegian University of Science and Technology, Trondheim, Norway.
[2]Marianne Holmer, Institute of Biology, University of Southern Denmark, Odense, Denmark
[3]Corresponding author: yngvar.olsen@bio.ntnu.no

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Abstract

The worldwide fast growing aquaculture industry has caused a growing attention and concern on the more societal aspects of the industry, among them the environmental effects of aquaculture on coastal ecosystems. This has spawned new legislation, but also research activity to support legislation, management practices, and actions taken by the industry to reduce environmental impact. As a broad fundamental perspective for management, an emerging view is that fish farms should be managed as a part of the marine ecosystem according to a principle of "Ecosystem-based Approach to Aquaculture".

There is so far no general concept of understanding and sufficient knowledge as to how potential harmful impact on open water ecosystems can be assessed and managed. Benthic impacts are better understood and managed, but there is still a general lack of standardisation and agreement on monitoring practices. The main objective of the present report is to review and evaluate the state of knowledge from the recent accumulating international literature on environmental impact of aquaculture caused by nutrient emission to coastal waters.

Individual fish release nutrients as dissolved inorganic nutrients through excretion (NH_4 and PO_4), particulate organic nutrients (PON and POP) through defecation, and dissolved organic nutrients (DON and DOP) through resuspension from the particulate fractions. On the scale of a fish farm, there will additionally be a direct loss of Feed-N and Feed-P (uneaten feed). These different waste components will affect different parts of the marine ecosystem; feed losses and the larger faeces particles will sink and affect sediments and benthic communities whereas dissolved inorganic nutrients, dissolved organic nutrients, and small faeces particles affect the pelagic communities and state and quality of euphotic waters.

The release rate of nutrient wastes from intensive aquaculture can be estimated by mass balance based on statistical information on feed use and fish production, combined with information on feed losses, contents of N and P in the feed and the fish, and assimilation efficiencies of the dominant N and P components of the feed. Such estimates are particularly robust for N when feed losses are low, feed production in the system is low, and the statistical information is adequate for the purpose. Use of P-components from higher plants in the feed introduces a major uncertainty in the assimilation efficiency of P in the fish, making estimates more uncertain. An alternative approach for estimating waste emission rates from aquaculture, although not that robust, is to estimate waste losses by dynamic metabolically based modelling of fish feeding, growth, and metabolism.

An estimation of waste emission by mass balance made for salmon cage aquaculture with an overall feed conversion efficiency of 1.16 kg dry feed per kg fish produced shows that slightly less than 2/3 of N and P of the feed is released as wastes. NH_4 is the main waste component of N whereas particulate P is the main component of P. Feed-N, Feed-P, DON, and DOP are all minor waste components of salmon aquaculture. The estimations agree generally well with measured published values.

N and P are not toxins, but biogenic elements which are potential harmful in the marine environment only if their supply exceeds the assimilation capacity of the ecosystem. All ecosystems have an inherent capacity of persistence, and smaller changes in nutrient supply are mitigated through adaptive responses of the communities. The scientific understanding of these processes and impacts in benthic ecosystems is well developed, but this is not the case for pelagic ecosystems. We propose that the assimilation capacity of nutrients of the water column ecosystem is mediated by two main mechanisms: 1) Nutrient uptake by phytoplankton and further transfer to higher trophic levels and 2) Dilution of nutrients and organisms mediated by hydrodynamics at production sites and their surrounding, downstream

water masses. Both mechanisms will be important in a general scientific scheme, which can support management of the pelagic system. Hydrodynamics is most important; high energy is paramount for large intensive fish farms. Management of pelagic impacts require monitoring on a regional scale, because the potential environmental effects will normally become expressed on a regional scale of the fish farm. Satellite imaging and 3D modelling are suitable means for monitoring and managing these effects, in addition to more classical measurements.

Particulate nutrients affect the sediments and benthic ecosystems below fish farms and in the immediate surrounding area. It is quite well understood how these accumulations of nutrient wastes distribute in sediments as a consequence of fish production, bottom topography, water current velocity and water depth. Severe accumulations can cause major changes in the structure and function of benthic ecosystems locally, i.e., normally resulting in decreasing diversity of the benthic fauna and flora. The common result is highly reduced conditions due to sulphide accumulation with a shift in decomposition of organic matter from fauna mediated to microbial processes. Oxic microbial processes like nitrification are inhibited, and this may also limit denitrification due to low nitrate concentration and sulphide toxicity. The result is high ammonium and phosphate release from fish farm sediments. There are a number of methods and models to classify and assess trophic state of sediments, which can support a scientifically based monitoring and management of benthic impacts from aquaculture in coastal sites.

There are an increasing number of papers published on environmental interactions of aquaculture made available in ISI Web of Science databases over the last decades. The fraction of papers treating environmental aspects of aquaculture was 16% of the total numbers of aquaculture papers for the period 1990-1999 and 20% for 2000-2006. The corresponding fraction combining aquaculture and eutrophication for 1990-1999 and 2000-2006 were 1.4 (29 papers) and 2.4% (92 papers), respectively, corresponding to about 10% of the environmental papers related to aquaculture. There are as expected far more contributions for the benthic impacts of aquaculture than for the pelagic, which still lacks a general scientific concept that can support identification and management. The report summarise more recent, important contributions for eutrophication and aquaculture in general and for benthic and pelagic impacts. Finally, the report recommends R&D actions to fill gaps of knowledge needed to meet requirements of future, international legislation.

1. Introduction

The worldwide fast growing aquaculture industry has gone through its first immature stage in western countries, and there is now a growing attention and concern on the more societal aspects of the industry. These include for example animal of welfare and particularly environmental effects of aquaculture on coastal ecosystems (FAO 2006). The perception of environmental problems appears to be different in the different countries. Escapes and interactions with wild populations is a major issue in many salmon producing countries whereas other countries regard organic and inorganic waste emission to be more important (Muir 2005). The social perception of the importance of an environmental problem is affected by the physical and biological characteristics of the coastal system as well as interactions with other important industries or human activities in the coastal zone. The tourist industry is for example a major stakeholder and competitor for space in many Mediterranean countries, competing strongly with aquaculture. Associations and industries involved in river fisheries of salmon are important stakeholders opposing for example the Norwegian salmon industry (Porter 2003, McGinnity et al. 2003).

The growing concern on the environmental impacts of aquaculture has spawned new legislation, but also research activity to support legislation, management practices, and concrete action by the aquaculture industry to reduce environmental impact. It is quite clear that the early immature aquaculture industry has created major environmental damage, for example, the early shrimp industry (Alonso-Rodriguez and Paez-Osuna 2003). It is also clear that major improvements have been made through improved technology and management practice, and that the environmental impact of today's industry in many cases is less than believed by many citizens. It must also to greater extent than today be realised that aquaculture is the first to suffer from inadequate coastal water quality and environmental damage. The industry is thus very dependent on minimising its own potential pollution, but also the pollution originating from other human resources, be it industry or urban runoff. It is therefore more correct to consider the environmental interaction of aquaculture than the environmental impact of aquaculture.

The concept of an "Ecosystem-based Approach to Aquaculture" (EAA), now introduced by FAO representatives in cooperation with representatives of the scientific community, has been attractive for many scientists. The current EAA definition, established in a workshop on *"Building an Ecosystem Approach to Aquaculture: initial steps for guidelines"* took place in Palma de Mallorca, Spain, in early May 2007 is as follows (D. Soto, pers. comm.):

"An Ecosystem Approach for Aquaculture is a strategy for the integration of the activity within the wider ecosystem such that it promotes sustainable development, equity, and resilience of interlinked social-ecological systems"

According to the principles of this concept, the fish farms should be managed as a part of the marine ecosystem. It is our understanding that wastes generated by aquaculture activity should be evaluated holistically based on nature's inherent capacity to assimilate inorganic nutrients and organic matter and the potential danger of exceeding these limits.

The objectives of the present report are to review the state of knowledge from recent international literature on environmental impact of aquaculture. The environmental threats of other polluting industries for aquaculture, i.e., aquaculture interactions with the environment, are not covered. The scientific literature describing impacts on sediments and benthic ecosystems is very comprehensive, and there is a general scientific understanding on which we may base assessments of state and dynamics, management principles, and monitoring

measures. The potential impacts of wastes from aquaculture on water column ecosystems is far less studied, presumably because it has been difficult to identify and quantify such impacts. It is also apparent that there is no clear scientific concept derived for assessments of state and dynamics, for establishing management principles, and for undertaking monitoring measures. There is thus not sufficient knowledge as to how specific measured environmental variables can reflect potential harmful impact on open water ecosystems. Despite the extensive knowledge on benthic impacts, there is a general lack of standardisation and a lack of agreement on monitoring practices, primarily due to the lack of expertise and the high costs related to the proposed protocols. The present report describes a preliminary general concept applicable for assessing impacts in open water column ecosystems and presents the current state of knowledge on benthic impacts. Finally, the report lists the relevant recent literature on the issue and gives recommendations for R&D needed to meet future requirements of environmental legislation.

2. Food conversion and waste production by fish

2.1. Carbon, nitrogen and phosphorus in organisms

Nitrogen and phosphorus are the nutrients most likely to induce environmental impacts like eutrophication in the water column, whereas organic carbon deposition in addition can contribute to disturbance or severe impacts of benthic ecosystems. The following section is a brief summary of some aspects of the biogeochemical role of carbon, nitrogen and phosphorus relevant to our discussion of eutrophication effects of aquaculture.

Carbon is the backbone of all organic molecules, and therefore the ubiquitous element of all living organisms. The primary source for building organic carbon is inorganic carbon (CO_2) dissolved in the oceans. Photosynthesis[1] in aquatic and terrestrial plants converts CO_2 to organic carbon, thus making it available to the rest of the biosphere. All ecosystems are dissipative structures, in the sense that organic carbon is normally broken down to inorganic carbon by respiration[1] by the organisms. Approximately 50 % of the dry weight of any organism, including fish and its natural prey, is carbon (Sterner and Elser 2002).

Nitrogen is an essential element, with high contents in proteins and nucleic acids. Although 78% of the air is nitrogen in the form of N_2 gas, nitrogen is still often in limited supply for the organisms in the biosphere. Nitrogen fixing bacteria are the only organisms able to take up N_2 gas and convert it into biomass, whereas other organism use dissolved inorganic N (plants and bacteria) or organic forms of the element (e.g., animals) (Wallace et al. 1991). During digestion, proteins are broken down to amino acids, which can be used to build new proteins in the cells, or they may be converted to fatty acids or carbohydrate, or respired for maintenance metabolism. During deamination, the amine group, NH_2, is removed from the amino acid as ammonia (NH_3). Ammonia is toxic, but organisms living in water can usually exchange material easily and thus get rid of excess ammonia, which forms non-toxic ammonium (NH_4) in normal sea water (at pH≈8). Terrestrial animals, like e.g. mammals, convert ammonia to uric acid or urea, and excrete it in the urine. Some species of fish excrete mainly ammonia through the gills (Wilkie 2002), while other species excrete mainly urea, and undigested nitrogen compounds will leave the animal in faeces (Wallace et al. 1991).

Phosphorus in biological systems is always combined with oxygen as phosphate (PO_4). Phosphate may be free inorganic, or combined with organic molecules. The ultimate source of

[1] Element balance of photosynthesis, in general terms: $CO_2 + H_2O \rightarrow CH_2O + O_2$. Respiration is the inverse process of photosynthesis

phosphate is crystalline rocks, and erosion, weathering, and to some extent, the action of plant roots and lichens, which make it available to organisms (Clapham 1983). Phosphate is essential in the nucleic acids (DNA and RNA), in adenosine triphosphate (ATP) which is an important energy storage and transfer molecule in all organisms, in phospholipids in cell membranes, and in calcium phosphate forming teeth and bone (Wallace et al. 1991). In order to be taken up in the gastrointestinal tract in fish, P compounds must be broken down to inorganic phosphate. If the plasma concentration of PO_4 becomes too high, the fish excretes the excess in the urine, while undigested phosphorus is expelled from the fish in faeces (Bureau and Cho 1999, Roy and Lall 2004).

2.2. Composition of fish food

In nature, fishes eat other organisms with a biochemical composition quite similar to themselves. The crude biochemical composition of a fish is variable, but can typically be around 20 % protein, 1-2 % lipid (can be much higher), 2 % ash, 70 % water, 15 % carbon, 3 % nitrogen and 0.5 % phosphorus (Tantikitti et al. 2005). Formulated fish food is more or less tailor made for the cultured fish species and culture system in use, and has a distinctly different composition from natural prey. Table 1 compile some formulations of experimental pellet food for different fish species. The average pellet composition is 40 % protein, 17 % fat, 19 % carbohydrate, 14 % ash, 7 % water, 50 % carbon, 6 % nitrogen and 1 % phosphorus. The ranges for the components are, however, quiet broad. Commercial pellet food is likely to be of similar composition. Carbohydrates and lipids are included in formulated foods to increase the protein retention efficiency and reduce the nitrogen and phosphorus waste production (Adron et al. 1976, Ruhonen et al. 1999, Hillestad et al. 2001). Fish require essential fatty acids of the lipids, but fish do not require carbohydrate in the diet. If carbohydrates are excluded from the food, the fish catabolise proteins and lipids for energy and metabolic intermediates (Wilson 1994). Haddock, for example, seems to use preferentially protein as the source of energy (Tibbetts et al. 2005).

2.3. Food conversion ratio

The food conversion ratio (FCR) is most commonly defined as the amount of dry food consumed per wet fish biomass produced. The FCR will accordingly reflect the digestibility and quality of the feed. The feed quality is determined by the composition and the availability of the nutrients, i.e. the digestibility. Because parts of the food are not easily digested, the fish produce a certain amount of waste material (Figure 1). Because some of the ingested food is respired to sustain maintenance metabolism, the fish must eat more of some components than strictly needed to build biomass. Therefore, the energy availability also affects the FCR. Fernandez et al. (1998) studied the digestibility of pellet food in gilthead sea bream (*Sparus aurata*) and found the apparent digestibility of elements to be:

Nitrogen > Carbon > Dry Matter > Phosphorus.

N, C and Dry Matter digestibility was correlated, while P digestibility was not well correlated to the other nutrients or dry matter digestibility. In Table 2, we have compiled some FCR values for different experimental food-fish combinations. By fine tuning the composition of the food, and increasing the availability of both energy and nutrients, it is possible to reduce FCR to very low values, like for example 0.64 as reported for the Red Pacu (Table 2).

Table 1. Examples of the percent composition of main ingredients of fish food in use in different aquaculture systems.

Species	Food type	Protein	Fat	Carbohydrate	Ash	Moisture	N	P	References
Salmon		45.0	22.0	16.0	9.0	8.0	6.0	1.0	Cheshuk et al. (2003)
									Islam et al. (2005)
Salmon		43.6	34.1	12.7	7.8	7.3			Hevroy et al. (2004)
Salmon	Experimental pellet	34.0	29.3	17.8	6.3	9.8			Hillestad et al. (2001)
Salmon	Exp, low lipid	39.0	31.0	22.0	6.0				Hemre and Sandnes (1999)
Salmon	Exp, high lipid	40.0	47.0	7.0	6.0				Hemre and Sandnes (1999)
Milkfish	Natural food based	11.0	0.8		61.5		1.8	0.7	Sumagaysay-Chavoso (2003)
Milkfish	Formulated	33.5	8.8		9.7		5.4	1.3	Sumagaysay-Chavoso (2003)
Milkfish	commercial	29.6	7.6		9.5		4.7	1.1	Sumagaysay-Chavoso (2003)
Asian seabass	Exp1	40.8	11.5		17.6			2.6	Tantikitti et al. (2005)
Asian seabass	Exp2	42.3	11.5		15.3			2.0	Tantikitti et al. (2005)
Carp	exp	34.5	13.4		8.5	6.5		1.4	Jahan et al. (2003)
Carp	commercial	25.3	10.3		8.6	7.8		1.6	Jahan et al. (2003)
White sea bream	exp	64.1	18.5		16.2	11.4			Sá et al. (2007)
Gilthead sea bream	exp	39.9	10.4	35.5	10.9	10.1			Fountoulaki (2005)
Gilthead sea bream	Exp IND7	59.8	8.6	11.8	19.8	2.6			Fernández et al. (1998)
Gilthead sea bream	Exp NOR5	52.0	11.3	23.6	13.1	2.3			Fernández et al. (1998)
Haddock	exp	45.5	10.6	27.8	9.2	6.9			Tibbetts et al. (2005)

Table 2. FCR (wet weight fish produced per weight feed consumed) for individual fish reported for different aquaculture fish species.

Species	Country/region	Food type	FCR	Reference
Red Pacu	USA	Exp. pellet	0.64	Palacios et al. (2006)
Cuneate drum	China	Exp. pellet	1.05-1.4	Wang et al. (2006)
Japanese sea bass	China	Exp. pellet	0.99-1.08	Xue et al. (2006)
Red sea bream	Japan	Exp. pellet	1.14	Sarker et al. (2005)
Indian major carp	India		4.55-6.75	Singh and Balange (2005)
Nile Tilapia	Egypt	Exp. pellet	1.43-2.53	Sweilum et al. (2005)
Asian sea bass	Thailand	Exp pellet	1.03-1.33	Tantikitti et al. (2005)
Asian sea bass	Thailand	Trash fish	2.86	Tantikitti et al. (2005)
Salmon	Norway	Exp. pellet	0.78-0.86	Hevroy et al. (2004)

2.4. Estimation of nutrient release rate from fish

2.4.1. Mass balance in a Food-Fish-Waste system

The carbon (energy) mass balance for the flow of matter through a fish can be represented by the following simple mass balance equation:

(1) $I = A + F = G + R + F$

I is the food consumed, A is assimilated food, or uptake in tissues; F is defecation; R is respiration, and G is growth and reproduction (all in terms of carbon or energy). The corresponding nutrient balance is expressed using the analogue equation:

(2) $I_{NP} = A_{NP} + F_{NP} = G_{NP} + E_{NP} + F_{NP}$

where excretion of N and P (E_{NP}) replaces respiration. These general equations, together with knowledge on assimilation efficiencies of C, N, and P and the stoichiometric C:N:P composition of produced fish an feed, are fundamental for estimating nutrient and carbon intake, metabolism, and losses from individuals of cultured fish (i.e., Figure 1 and 2 below).

The processes of respiration and excretion release inorganic carbon and excess nutrients, respectively, from fish tissues (assimilated matter) to the water. Respiration is a loss of carbon dioxide (CO_2) reflecting the metabolic costs of growth and maintenance of the organisms. The excreted N and P species is mainly inorganic nutrients wastes, i.e. urine (urea-N, PO_4) and ammonia (NH_4). These losses of assimilated N and P are instrumental to maintain elemental homeostasis in fish tissues.

The assimilated food is the portion of the food that is digested by the fish and taken up in tissues, and the assimilation efficiency (AE) is defined as (similar for N and P):

(3) $AE = A/I$

The undigested food, termed faeces, passes through the fish gut undigested or partially digested. This fraction constitutes mainly particulate organic substances, including particulate forms of N and P, but some part is rapidly released in molecular dissolved forms in the water. The assimilated food supports growth and weight increment, and the growth efficiency (GE) is generally defined as (similar for N and P):

(4) $GE = G/I$

This term expresses the efficiency by which the food ingested is converted to new biomass, similar, although inverse, to the FCR which is an operational term established and used for aquaculture.

The total wastes of carbon (TL_C) and nutrients (TL_{NP}) generated by cultured fish is expressed as:

(5) $TL_C = I - G = R + F$

(6) $TL_{NP} = I_{NP} - G_{NP} = E_{NP} + F_{NP}$

Respiration results in a release of inorganic CO_2, the emission of organic carbon wastes (L_{OC}) is most easily estimates as:

(7) $L_{OC} = I - A = I (1 - AE)$

Values for AE of carbon or energy can be obtained from literature and in some cases from feed companies. For the dissolved components from faeces, there is no formal way to distinguish these dissolved organic components (DOC) from the particulate organic waste components (POC), but the particulate fraction is the most important (see below).

The corresponding estimate of organic nutrient wastes (L_{ONP}) from fish is:

(8) $L_{ONP} = I_{NP} - A_{NP} = I_{NP} (1-AE_{NP})$

I_{NP} can be estimated based on total feed intake times feed NP contents. The assimilation efficiency of N can be assumed to be equal to that of protein, and values are widely reported in literature and by feed companies. The assimilation efficiency of P is widely reported as well, but more uncertain because of the addition of indigestible P compounds from higher

plants in the feed (phytate P). As for carbon, there is no formal way to distinguish between dissolved organic nutrients (DON, DOP) and particulate organic nutrients waste components (PON, POP) originating from faeces, but the particulate nutrient fraction is the most important (see below).

The inorganic N and P release from the fish (L_{INP}) can be estimated as the difference between assimilation and production:

$$(9) \qquad L_{INP} = A_{NP} - G_{NP} = (I_{NP} * AE_{NP}) - G_{NP}$$

G_{NP} is N and P in produced fish, obtained as produced fish weight times N and P contents.

2.4.2. Input data and some mass balance exercises reported

The specific use of the above general mass balance model must be adapted to the data that are available for the actual species or aquaculture system. Some reports on these issues from the literature are as follows.

According to Cheshuk et al. (2003), the typical composition of extruded pellets for salmon or trout is 45 % protein, 22 % fat, 16 % carbohydrate, 9 % ash and 8 % moisture. These authors found the digestibility (AE) to be 0.83, i.e. the sum of the organic fractions of the pellet. Corner et al. (2006) studied the carbon assimilation efficiency for farmed salmon in their study of waste deposition. Approximately 50 % of the salmon feed pellet was organic carbon, and 14 % of this was allocated to growth, 60 % was respired by the fish and 26 % was excreted in faeces. According to these results, 74 % of the organic carbon in the pellet was digestible (Table 3).

The principal N source of salmonid feed pellets is the proteins that are vital for fish growth. Roughly 6 % of pellet dry weight is N, while approximately 3 % of the fish wet weight is N (Ackefors and Enell 1990, Davies and Slaski 2003, Islam 2005). According to Anderson et al. (1995) on average 80 % of the amino acids (i.e. proteins) in various fishmeals was available to Atlantic salmon. If the proteins are the main nitrogen source, this means that 20 % of ingested N is likely to be undigested and therefore lost in faeces, as particulate organic N. Fish and faecal N constitutes 64 % of ingested N. According to the budget above, 36 % of ingested N is excreted as dissolved inorganic N (NH_4). Alternatively, if 26 % of feed eaten ends up as faeces (Butz and Vens-Cappell 1982) and faeces contain 4 % N (Penczak et al. 1982), 17 % of total feed N is released in faeces, in agreement with the budget calculated from N in feed and fish (Figure 2, Table 3).

The assimilation efficiency for Feed-P in fish is typically found to be 40-50% (Sugiura et al. 2006). Animal ingredients, e.g. fishmeal, contains bone-P, (mainly hydroxyapatite, i.e. acid extractable P), while plant ingredients contains phytate-P, which is almost indigestible to fish because they lack the phytase enzyme. Both animal and plant ingredients contain other organic P components that will easily be hydrolyzed by phosphatases. These components are therefore assumed to be digestible for fish (Hua and Bureau 2006). As an example, one brand of commercial trout feed contained 0.9 % acid extractable P and 0.05 % phytate (Sugiura et al. 2006). When fed a herring fish meal diet containing 53 % bone-P, 16% phytate-P and 31 % organic P, rainbow trout released 32, 100 and 16 % of the three ingredients, respectively, through faeces (Hua and Bureau 2006). Fecal P from fish is mostly undigested P and represents solid P waste. If the plasma concentration of phosphate (PO_4) becomes too high,

the fish excretes the excess through urine (Bureau and Cho 1999, Roy and Lall 2004). In the example above, the fish released 38 % of ingested P through faeces. If the assimilation efficiency of P was 50 %, the fish must have released (approximately 10% of the ingested P through urine. The P in urine is digested and assimilated, i.e. free phosphate, which is dissolved and readily available to phyto- and bacterioplankton (Table 3).

Sugiura et al. (2006) established a P-effluent profile for rainbow trout fed a diet with 1.4 % P. They found that 42 % of the P consumed was recovered in the effluent, the same amount was retained in the fish, and 16% of total P fed was un-recovered in the experiment. 23 % of the total P fed was recovered in faeces, 13 % in small particles and 6 % in the dissolved fraction. In Table 3, we have compiled values for assimilation efficiency, faeces production and excretion of the elements C, N and P extracted from the literature.

2.5. Leaching of dissolved organic matter from waste particles

Chen et al (2003) found that about 15 % of the C and N in faeces leached after a few minutes in water. This will add to the dissolved organic carbon and nitrogen (DOC) pools of the water column. Phillips et al. (1993) studied P-leaching from salmon diets and faeces. The leaching from pellets was linear up to 10 min, in which 1.7 mg P was lost per g dry weight of the pellets (ca 15 % of total P). The leaching from faeces was non-linear and most of it took place during the first 2.5 minutes in which 2.5 mg P was lost per g dry weight of faeces. Sugiura et al. (2006) found that approximately 15 % of faecal P was soluble in during minutes to hours.

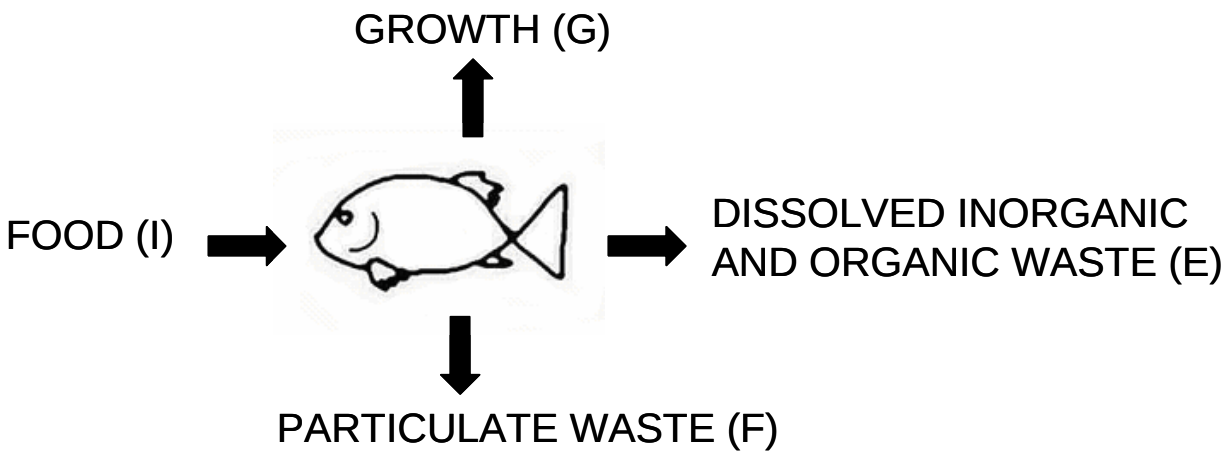

Figure 1. Concept model for Food-Fish-Waste system

Table 3. Assimilation efficiency, faeces production and excretion as weight fraction of ingested C, N, and P in fish reported in recent publications.

Element: **Carbon**	Fraction of ingested C	Reference
Assimilation efficiency	0.74-0.83	Corner et al. (2006) Cheshuk et al. (2003)
Faeces production	0.26	Corner et al. (2006)
Excretion (respiration)	0.48	Corner et al. (2006)

Element: **Nitrogen**	Fraction of ingested N	Reference
Assimilation efficiency	0.80	Corner et al. (2006)
Faeces production	0.17-0.20	Butz and Vens-Cappel (1982)
		Penczak et al. (1982)
		Anderson et al. (1995)
Excretion	0.36	Butz and Vens-Cappel (1982) Penczak et al. (1982)

Element: **Phosphorus**	Fraction of ingested P	Reference
Assimilation efficiency	0.40-0.50	Sugiura et al. (2006)
Faeces production	0.38	Sugiura et al. (2006)
Excretion	0.10-0.20	Sugiura et al. (2006)
		Roy and Lall (2004)

Table 4. Release of dissolved organic carbon (DOC), nitrogen (DON) and phosphorus (DOP) from particulate aquaculture waste as percentage of total weight (see elemental flows of C, N, and P of salmon farming in Figure 2).

Particle type	Release rate	Substance released	Reference
Salmon faeces:	10-15 % loss of C and N in 2.5 minutes, then stops (on short timescale)	DOC, DON	Anderson et al. (1995) Chen et al. (2003)
Faeces	ca 15 % of P released in minutes to hours	DOP	Sugiura et al. (2006)
Pellet	ca 15 % of P released in minutes to hours	DOP	Phillips et al. (1993)

Table 5: Chosen values for the parameters in the mass balance budgets for carbon, nitrogen and phosphorus in the food-fish-waste system. See text for references. Assimilation is the weight fraction of ingested element taken up by the fish.

	Carbon	Nitrogen	Phosphorus
Assimilation efficiency (A)	0.8	0.8	0.5
% of food dry weight	0.5	6.0	1.0
% of fish dry weight	0.5	8.0	0.5

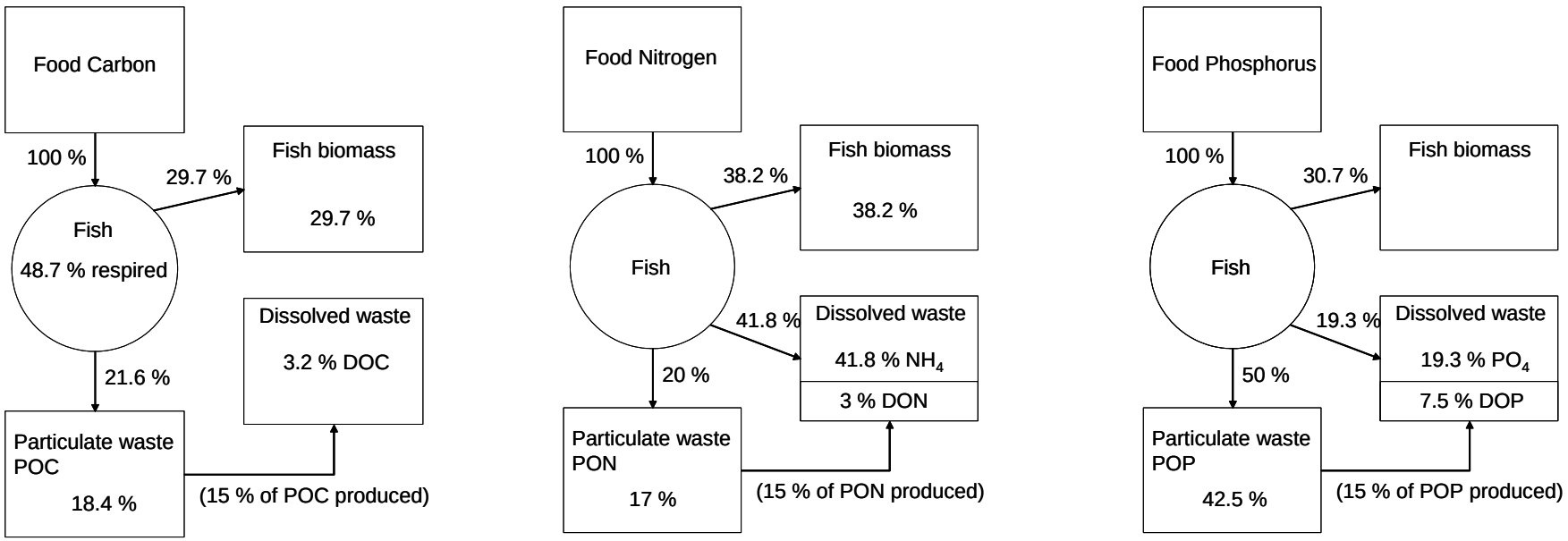

Figure 2. The flow of C, N, and P based on mass balance budgets (Equation 1-9) and literature data for key parameters (Table 3 and 5). The resuspension rate for dissolved organic matter was set to 0.15 according to values compiled from recent publications (Table 4).

3. Food conversion and waste production by aquaculture facilities

The waste production budget for an aquaculture facility differs from the fish budgets, as there is an additional waste component in the uneaten feed that may affect the environment. If mortality is significant, it should be included as well as a loss of nutrients. Dead fish are usually collected for land deposition, and are as such not lost directly to the environment. To what extent a fish farm and the produced wastes affect the surrounding environment, depends on the size and type of the fish farm, the structure and functioning of the pelagic and benthic surrounding ecosystems, the overall management practise of the aquaculture facility, and perhaps most important; the hydrodynamic energy and physical conditions of the location.

The nutrient release rate of a fish farm can, like for individual fish, be estimated using the general metabolic balance of animals (model and data in Chapter 2). This method is most applicable for situations when the cultured organisms feed primarily on the added feed, and not on feed which are produced in the system, like for example in pond aquaculture systems. The method is robust and most accurate when there is statistical information available for use of feed and fish production with time. If such statistical information are more fragmented, the alternative to a metabolic balance model (Chapter 2) is a more general dynamic metabolic model describing fish metabolism, growth, and waste production (e.g. Fernandes et al. 2007). Such models are not that robust, but can be useful when the input data needed for a more robust mass balance exercise are lacking.

3.1 Closed or semi-closed systems

Various closed or semi-closed aquaculture systems are in use today, from pond or raceway systems where water quality is controlled by water dilution or discharged to the environment, to recirculation aquaculture systems (RAS), in tanks or ponds, where water quality is controlled and managed by microbial reactions (Brune et al. 2003). In a typical tank or pond system without sufficient water exchange or aeration, the oxygen concentration will fall to critical levels within minutes due to fish respiration. Even if pure oxygen is supplied, respiratory carbon dioxide will skew the carbonate equilibrium chemistry and suppress the pH to toxic levels. Aeration with air meets the needs for both O_2 supply and CO_2 degassing, but within a few hours, the free ammonia concentration will reach toxic levels. The need for ammonia removal can be mediated with water exchange, microbial treatment using biofilters, or physical/chemical removal techniques (Brune et al. 2003). Three important processes affecting the water quality in pond systems are: 1) photosynthesis (algae), 2) heterotrophic bacterial biosynthesis, and 3) nitrification (bacteria). All these processes remove ammonia, either through uptake and assimilation in the organisms, or through conversion of ammonia to nitrate. Potential problems with algal growth in the ponds are fluctuations in oxygen concentration, pH, and blooms of toxic algae (Brune et al. 2003). These problems can be mitigated by confining the algae in separate algal treatment ponds (Porello et al. 2003).

Treatment of waste effluents is crucial for maintenance of closed and semi-closed systems with more or less recirculation of water. Such treatments includes mechanical filtration, chemical precipitation using e.g. alum and polymers (Yang et al. 2006, Ebeling et al. 2006, Rishel and Ebeling 2006), biofiltration with heterotrophic bacteria (Schneider et al. 2007, Schneider et al. 2006), cyanobacteria (Kamilya et al. 2006), macroalgae (Henry-Silva and Camargo 2006, Schuenhoff et al. 2006), microalgae (Metaxa et al. 2006), aquatic and terrestrial plants (Castro et al. 2006), mangrove, and oysters (Shimoda et al. 2006).

Pond aquaculture systems without efficient recirculation of water frequently direct the effluent through a treatment pond or lagoon before discharging their wastewater into ambient

waters. In a land-based sea bass farm on the French Atlantic coast, the daily water exchange rate varied between 30 % of pond volume in the winter, to 400 % in the summer. The effluent was drained through a lagoon, retained there for at least 24 hours, and finally discharged into ambient waters (Brune et al. 2003).

In tropical and subtropical areas in Asia, Mexico and Latin America, shrimp farming has developed immensely during the last 15 years. Shrimps are most commonly reared in pond cultures, often in connection with coastal lagoons (Alonso-Rodriguez and Paez-Osuna 2003). Coastal water is a common water source for the shrimp ponds, either used directly or pumped via coastal lagoons. The algal community of the supplied water is being modified in shrimp ponds. In early developmental stages, the shrimp larvae feed on microalga, copepods, detritus, and mollusc larvae. In later stages, the phytoplankton contributes mostly through the trophic chain, because the shrimp feed on macro and meio-fauna. The special conditions in shrimp ponds, characterised by elevated nutrient concentration and altered salinity due to mixing with freshwater, can promote blooms of algae, sometimes affecting the shrimp production negatively. Alonso-Rodriguez and Paez-Osuna (2003) compiled publications describing events of harmful algal blooms in shrimp ponds between 1993 and 2003. They found that anoxia negatively affected shrimp production in five cases, and ammonia and mucus production in some other cases. Blooms of toxic algae were observed in four cases (Alonso-Rodriguez and Paez-Osuna 2003).

Water and effluents from shrimp ponds are enriched with organic matter, nutrients and solids, including algal cells. The intensity (stocking density, water use, food and fertilizers) determines the loading rate of wastes (Jones et al. 2001). Concerns about the environmental impact of shrimp farm effluents have prompted a comprehensive search for efficient methods for effluent treatment. Jones et al. (2001) tested a three-stage effluent treatment system. In the first stage, the particle concentration was reduced by natural sedimentation. In the second stage, filtration by oysters further reduced the concentration of particles like detritus, phytoplankton, and bacteria. In the third and final stage, macroalgae absorbed the dissolved nutrients. Ammonium concentration was reduced by 76 %, nitrate by 30 %, phosphate by 35 %, bacteria by 30 %, and chlorophyll *a*, i.e. algal biomass, was not affected (Jones et al. 2001).

3.2 Open system: cage fish farm in coastal waters

The cage aquaculture system (CAS) is perhaps the most abundant system for producing seafood worldwide. The principal method for estimation of the nutrient loading rates is the same as described above, with losses of feed (and optionally fish mortality) included. The loading rate of carbon, nitrogen and phosphorus wastes to the sediment under the fish cages constitutes mainly uneaten feed and the faeces production from the farming unit. The C, N and P composition of feed pellets and faeces particles, water depth, water current velocity, sinking speed of particles, disaggregations, and the leaching from the sinking particle will modify the environmental impact (Table 6, Chapter 4).

Table 7 compile some food conversion ratios for different aquaculture facilities (FCR_{Farm}). These conversion ratios are in general higher than the FCR values for individual fish presented in Table 2. The reason is that direct food losses are included in the FCR_{Farm}. In addition, other factors like increased respiration due to stress, e.g. temperature or feeding activity, may also have contributed to elevate the feed conversion ratio (Islam 2005, Mente et al. 2006).

Table 6. Factors influencing the waste loading from cage aquaculture in open waters.

Factor	Quantity	Reference
Food waste	5-40% of weight, depending on food type and culture system	Islam (2005), Mente et al. 2006), Wu (1995)
Faeces production	15-26 % of feed quantity used	Cho and Bureau (2001) Butz and Vens-Cappel (1982)
Range for current velocity in coastal seawater	0-25 cm s^{-1}	Carroll et al. (2003)
Sinking speed		
Feed pellets:	6-14 cm s^{-1} (2-10 mm sizes)	Chen et al. (1999)
Faeces:	3.7-9.2 cm s^{-1}	Chen et al. (2003)
Resuspension of sediment	depends on sediment type and current speed close to sediment	Cromey et al. (2002)
Disaggregation of particles from fish cages		
Feed pellets	20-30 % (6,5-12 mm size range) mass loss after 120 h	Stewart and Grant (2002)
Faeces	mean particle size for faeces was 0.71 mm for sea bream and sea bass	Magill et al. (2006)

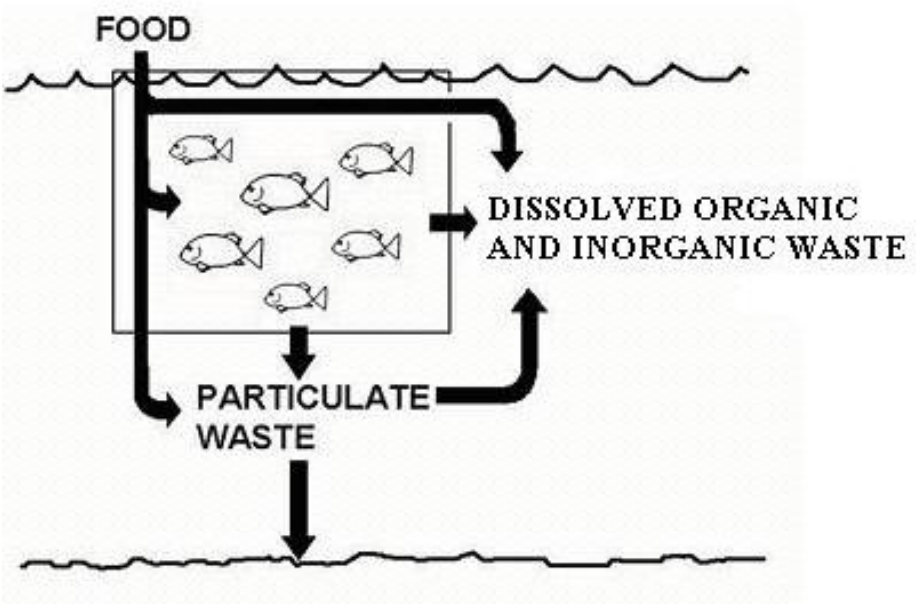

Figure 3. Principal fluxes of energy and nutrients in a cage fish farm suspended in open waters

Table 7. FCR_{Farm} for different combinations of fish species and food types (kg dry food supplied per kg wet weight fish produced).

Species	**Country/ region**	**Food type**	**FCR_{farm}**	**Reference**
Cod	USA	Commercial pellet	1.4-1.5	Chambers and Howell (2005)
Haddock	USA	Commercial pellet	1.3-2.4	Chambers and Howell (2005)
Barramundi snapper	Australia	Commercial food	1.23	Islam (2005)
Rainbow trout		Commercial pellet	1.14-1.28	Bureau et al. (2003)
Mandarin, bream, catfish		Forage fish, formulated diet	2.29	Guo and Li (2003)
Salmon	Europe	Pellet	1.17	Gillibrandt et al. (2002)
Halibut	Europe	Pellet	1.3	Gillibrandt et al. (2002)
Turbot	Europe	Pellet	1.3	Gillibrandt et al. (2002)
Cod	Europe	Pellet	1.5	Gillibrandt et al. (2002)
Haddock	Europe	Pellet	1.5	Gillibrandt et al. (2002)
Silver perch	Australia	Exp. and commercial pellet	1.9-2.2	Allan et al. (2000)
Silver perch		Commercial food	2.45	Gooley et al. (2000)
Areolated grouper		Trash feed	6.52[2]	Leung et al. (1999)
Gilthead seabream		Commercial food	1.79	Lupatsch and Kissil (1998)
Gilthead seabream and sea bass	Europe	Pellet	1.60-2.39	Holmer and Frederiksen (2007)

Feed losses is claimed to be low in modern CAS using dry feed (<5 %, Islam 2005, Mente et al. 2006). In fish farms using trash fish or other types of food, e.g. moist or wet pellets, feed losses may be much higher (Islam 2005). Leung et al. (1999) reported direct feed losses of 37 % in a cage farm using trash fish for food. The deposition of organic material below fish farms is complex; sinking particles will disaggregate in water, which will affect the sinking speed of the material. Magill et al. (2006) found that the mean particle size for faecal material was 0.71 mm, and that re-suspension rate of particles into dissolved components was species specific.

CAS in open waters are normally exposed to high rate of water exchange, and the fish do seldom experience direct harmful exposures to the wastes produced in the farm itself. This implies that specific management treatments, else than a major concern on where to locate the farm, has not been necessary to maintain high and cost efficient productivity. An increasing number of authors and other stakeholders have suggested to move CAS to more open sea (not

[2] FCR in terms of wet weight feed used per wet weight fish produced

protected by islands in all directions) or offshore locations (outside economic zone), which are normally deeper and have a higher rate of water exchange (e.g. Maldonado et l. 2005, Feng et al. 2004). This is particularly important for sensitive benthic ecosystems such as sea grasses (Holmer et al. 2005) or maerl communities (Wilson et al. 2004). Others have suggested to integrate fish culture with seaweeds, which may accumulate excess nutrients (integrated multi-trophic aquaculture, e.g. Yang et al. 2006, Carmona et al. 2006, Zhou et al. 2006b), and filter-feeders like mussels and scallops for remediation (e.g. Zhou et al. 2006c, Gifford et al. 2005, Newell 2004).

3.3 Emission flows of N, P and C from hypothetical CAS

The present section describes an annual mass balance for the main C, N, and P flows in a hypothetical salmon CAS producing 1000 metric tonnes fish per year (mt, wet weight). Assumptions made are according to our present state of knowledge and the mass balance model and data presented above (Chapters 2 and 3). Assumptions made are: the farm area is 51 000 m^2, the volume of the fish farm is 770 000 m^3, the FCR_{farm} is 1.16 kg dry feed used per kg fish produced (Gillibrandt et al. 2002), feed input is 1160 mt $year^{-1}$, direct feed loss is 5 % (Mente et al. (2006), feed P content is 1 % (Table 1), feed N content is 6 % (Table 1), and feed organic C content is 50 % (Table 1).

Tables 8 and 9 summarize the annual loading rates of organic C and nutrient components to benthic and pelagic ecosystems from this fish farm. To put the numbers into a perspective, the total nutrient waste generation of the fish farm corresponds to the emissions from a community of 7,500 to 10,000 people (2 g P per person and day, 13 g N per person and day, http://www.miljostatus.no/). The majority of the N wastes are released to open waters (68% of total) whereas the majority of the P is will accumulate in sediments (63%). We regard the P emission rates to be more uncertain than those for N. This is mainly because of the uncertainties in assimilation efficiency of P as compared to N (Table 3).

It is noteworthy that the N:P ratio of the wastes to open waters is well above the Redfield ratio (7.2, by weight); N:P for the total and inorganic nutrient emission was found to be 10 and 13, respectively (N supplied in excess). Contrary to this, the N:P ratio of the particulate wastes was 2.7 and therefore far lower than Redfield (P supplied in excess).

For the defined Norwegian salmon farm, there is a pronounced annual variation in fish and waste production over the year (mean for all farms), among others because of variable fish biomass and temperatures over time. The annual variation of N-and P-allocation flows into biomass and different waste components from the salmon farm are at their maximum in August in an average Norwegian salmon farm, with release rates less than half the maximum in the winter months (Figure 4). As a general trend, a lower proportion of Feed-P than of Feed-N is excreted as phosphate and ammonia, respectively. Contrary, the fraction of Particulate P, mainly released through defecation, is throughout higher than the fraction of Particulate N. The excretion of phosphate is highly dependent of the Feed-P-content, and the calculated P flows are relatively uncertain because of the problems to define accurate P contents of the feed and its overall digestibility (Tables 3-5). The release rate values for N are more robust, because the protein content of the feed is better defined, less variable, and its digestibility is higher and better known. It is also noteworthy that the emission rates of dissolved organic nutrients (resuspended DON), which is biologically relatively inert, and feed losses are relatively low as compared to other waste emission flows.

Table 8: Estimated loading rates of organic C and specified nutrient components from a hypothetical salmon CAS producing 1000 tonne fish per year.

Pelagic loading rates	tonnes farm^{-1} year^{-1}	g m^{-3} year^{-1}	mg m^{-3} day^{-1} (June-Sept)
OC-loading	20	26	100
NH_4-loading	28	36	140
PO_4-loading	2.1	2.7	11
Total N-loading	30	39	150
Total P-loading	3.0	3.9	15
DON + PON loading	17	22	86
DOP + POP loading	6.1	7.9	31

Table 9: Estimated loading rates of particulate organic carbon, nitrogen and phosphorus to the sediment below a hypothetical CAS producing 1000 tonnes of fish per year.

Benthic loading rates	tonnes farm^{-1} year^{-1}	g m^{-2} year^{-1}	mg (g sediment)$^{-1}$ year^{-1}
OC-loading	116	2300	92
PON-loading	14	270	11
POP-loading	5.2	100	4

The nutrient loading from a cage fish farm (Table 8 and 9, Figure 4) represents a point source of nutrients to the surrounding water. It is the hydrodynamic energy and the depth of the site that to great extent determine the water volume and area of sediments that receives the daily nutrient emission of wastes. Due to coastal and tidal currents, coastal seawater is rarely or never stagnant. The current velocity usually varies below 25 cm s^{-1} (Carroll et al. 2003), but metrological forcing can sometimes be paramount (Goyondet et al. 2005). The currents will contribute to water exchange in the cages, dilute the nutrient waste flows around and downstream of the farm, and broaden their deposition area considerably.

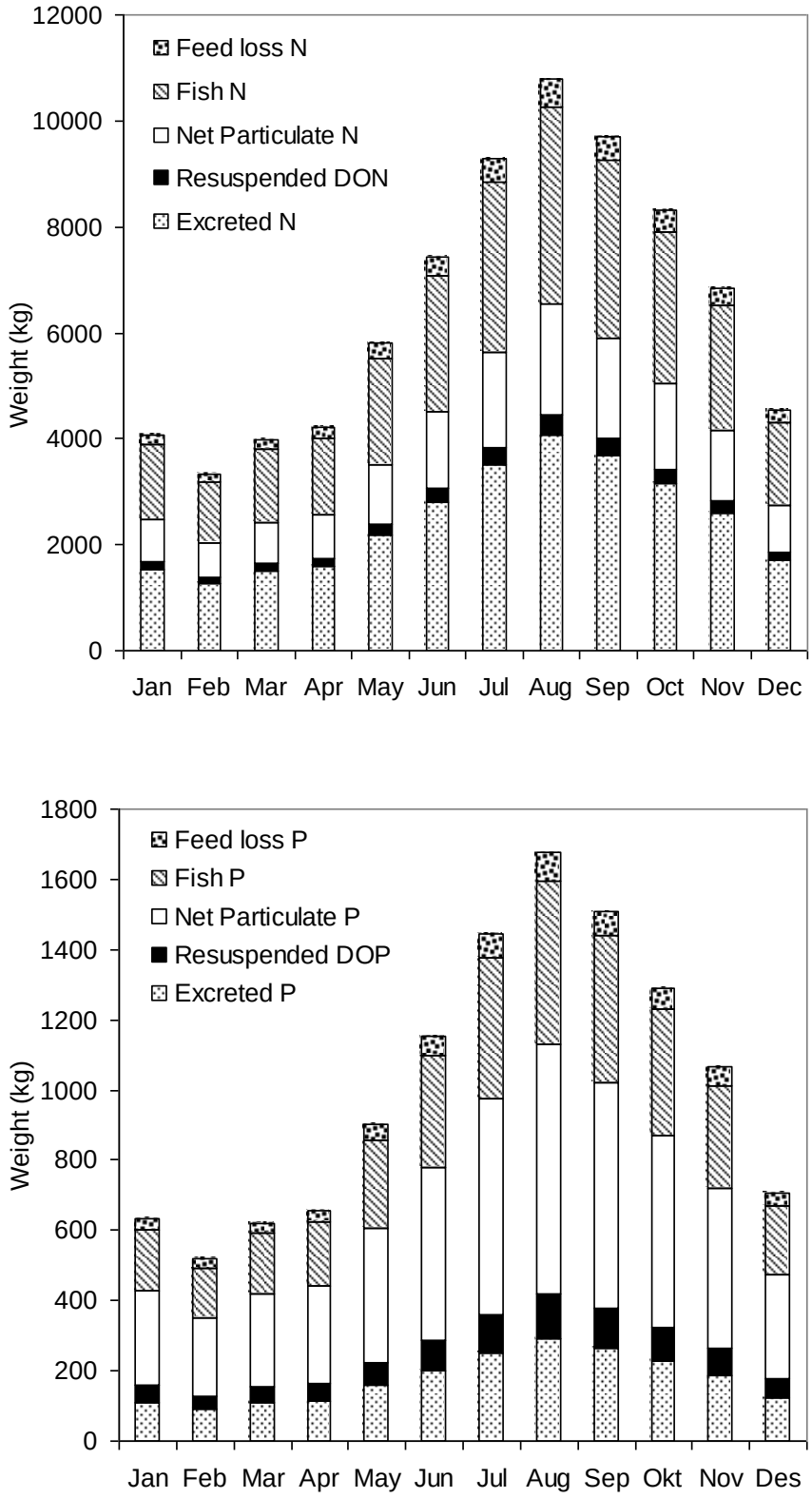

Figure 4. Annual variation in nitrogen (upper panel) and phosphorus (lower panel) in fish biomass and waste components for a hypothetical CAS producing 1000 tonnes salmon per year. The sum of the fractions equals the food nitrogen supplied.

3.4 Total waste production: a comparison with literature data

The production of one tonne of fish resulted in a release of 44 kg N, 8 kg P and 136 kg organic carbon (OC) according to the established mass balance of our hypothetical salmonid fish farm (Figure 4 and Table 7). An FCR_{Farm} of 1.17 kg dry feed used per kg wet weight fish produced is the typical current value for Norwegian salmon farms (FHL-statistics, Mente et al. 2006). For comparison and independent support, 35-45 kg N was released per tonne salmon produced in Scotland according to Davies (2000).

Different types of fish food have slightly different nitrogen contents (Table 1). Gillibrandt et al. (2002) compared total N discharge for salmon, halibut, turbot, cod, and haddock. Food N content varied between 6.25 and 9.3 %, the value of FCR_{Farm} was in the range 1.1-1.3, and total discharge was from 48 to 87 kg N per tonne fish produced. Islam (2005) constructed a conceptual model adapted to Asian conditions with FCR_{farm} equal to 2.5, with assumption of 1.4 % P and 6.5 % N in the food and 1 % P and 3 % N in the fish. According to this model, the waste release was 133 kg N and 25 kg P for each tonne of fish produced. Mediterranean species show yet relatively poorer FCR_{farm}; values around 2 (1.6 – 2.5) are commonly reported for dry feed by the industry. This generally implies far higher waste emissions to the environment for sea bass and sea bream aquaculture as compared to salmon.

The frequent use of moist or wet feed or whole fish as food source in many Asian countries results in higher FCR_{Farm} values, primarily because of increased food losses. Leung et al. (1999) studied the nitrogen budget in a cage farm with grouper and found that 87.7 % of the total N input to the farm was lost to the surrounding environment, corresponding to 321 kg N per tonne fish produced. Food losses were 37.7 %, and only 12.3 % of N input was accordingly harvested as fish biomass. The corresponding FCR_{Farm} was 6.52.

The assimilation efficiency may in some cases be lower when using trash fish as feed than for formulated pellets. Xu et al. (2007) measured total nitrogen and phosphorus budgets in CAS with trash fish as food for various fish species. The major waste fractions consisted of bones, scales and soft tissue, and 8 % of the P was recovered in bones and 6 % in scales of uneaten food. For N, 3 % was recovered in soft tissues, and 4 % in bones of uneaten food. .

Table 10. Summary of the C, N and P fractions (in metric tonnes) from the annual budgets of a hypothetical cage fish farm producing 1000 mt fish per year (Figure 5).

Element	Feed input	Fish harvest	Particulate Waste	Total Dissolved Waste	Total Waste	% Waste
Carbon	568	169	116	20	136	24
Nitrogen	70.2	25.7	14.3	30.1	44.4	63
Phosphorus	11.7	3.4	5.2	3.0	8.2	70
C:N:P (weight)	49:6:1	50:8:1	22:3:1	7:10:1	17:5:1	

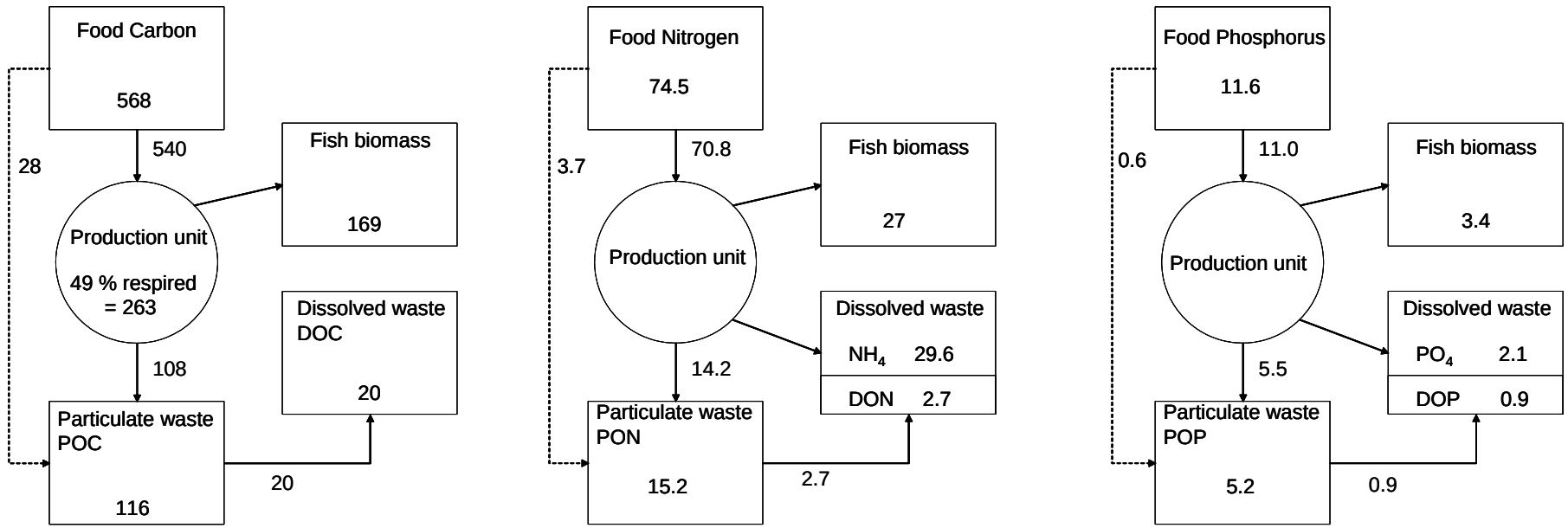

Figure 5. Annual nutrient loading to pelagic water and sediment (metric tonnes) in a hypothetical cage fish farm producing 1000 metric tonnes wet weight of fish per year. The budgets were constructed by using the fluxes from the fish mass balance budgets in Chapter 2 plus 5 % feed loss. The feed demand was calculated by using a FCR_{Farm} (weight feed supplied per wet weight fish produced) value of 1.16.

4. Scientific understanding of the environmental impact of wastes from aquaculture

The above sections have treated the quantification of nutrient release rates from individual fish and fish farms, which represent point sources of nutrients with variable nutrient and organic waste emission rates in space and time. The point source may be in open waters, like for CAS, or through drainage from land like for RAS and pond aquaculture systems. It was concluded above that it is the standing stock of fish, the feed conversion ratio of the farm, and the feed composition which primary determine the quantitative emission of nutrients from aquaculture systems. CAS normally represents larger point sources of nutrients to coastal waters than RAS (and pond systems). This is because of the larger standing stocks of fish in such systems as for example compared to land based RAS. Emissions from CAS are also often simpler to predict because land based systems may involve various post-treatments of the wastewater to remove particles and inorganic nutrients (Neori et al. 2004). We will accordingly make our outlines and evaluations below based on the characteristics of CAS, but the same thinking is valid for RAS and other systems as well.

4.1. Wasted nutrient components and impacts

Table 8 and 9 above show the annual load of different nutrient components released from fish farms, and Figure 4 shows the average variability of nitrogen waste components over an annual cycle in a salmon farm producing 1000 tonnes of fish per year. The waste estimates can be assumed to be proportional to the annual production, meaning that a fish farm producing 10.000 tonnes per year will have ten times higher waste generation if the feed conversion efficiency remains constant. Table 11 below summarise the characteristics and ecological fate of the nutrient components released from CAS. Fish residing in the water column will be affected by the particulate wastes (i.e., feed losses), but the ecological interactions of fish aggregation is not covered in this report. The general impacts of the nutrient components on pelagic and benthic coastal ecosystems can then be summarised as follows:

- **Benthic ecosystem, sediments:** Particulate nutrients that sink rapidly to deepwater will affect sediments and the ecosystem of the sea floor. Organic and inorganic dissolved nutrients have a minor direct ecological impact, only indirect through increased primary production of phytoplankton resulting in enhanced sedimentation of organic matter.
- **Pelagic ecosystem, water column:** Inorganic dissolved nutrients released can affect phytoplankton in euphotic waters quite strongly; the upper mixed, illuminated layer of the water column where photosynthesis takes place. Organic dissolved nutrients are too biologically inert to affect phytoplankton essentially. Particulate nutrients do not affect the phytoplankton in the mixed layer, but zooplankton are moderately affected all through the water column

There is a general scientific understanding of how particulate wastes from aquaculture accumulate in sediments below fish farms, and there is an understanding of how modifying factors like water current velocity, depth, and type of bottom substrate affect the accumulation patterns of wastes. There is also a general understanding of how benthic ecosystems react upon increased exposure of nutrient waste accumulation. This scientific concept of understanding is the basis for monitoring and managing CAS in coastal waters and for the

Table 11. Characteristics and fate of nutrient components released from aquaculture systems.

Nutrient component	**Acronym**	**Characteristics and fate**
Particulate nutrients	PON (particulate organic nitrogen) POP (particulate organic phosphorous)	Whole feed pellets, small to very small particles originating from the feed and fish faeces, and other particles generated in fish farms (e.g., fouling). Pellets and larger particles sinks rapidly to the seafloor, consumed immediately by fish or other benthic organisms, or accumulated/decomposed in sediments. Small particles are suspended in the water column, consumed within days by filter feeders (mussels, zooplankton) and bacteria. Particles are not available for phytoplankton and macro algae.
Dissolved organic nutrients	DON (dissolved organic nitrogen) DOP (dissolved organic phosphorous	Molecular nutrient components (organic), mostly complex chemical compounds, released from faeces particles and feed, and other biological activity. Stable N and P components, available for phytoplankton on very long time scale, some components are very stable (>100 year lifetime). To some extent consumed by bacteria-microbial food web, can aggregate and sink (marine snow), relatively slow process
Dissolved inorganic nutrients	DIN, dissolved inorganic nitrogen (ammonium, NH_4) DIP, dissolved inorganic phosphorous (phosphate PO_4)	Inorganic nutrients, i.e., ammonia (urea dissolves into ammonia) and phosphate Immediately taken up by phytoplankton, macro algae, and also by bacteria, used for growth, can in the worst case generate algal blooms.

management models derived to support that, e.g., MOM (Ervik et al. 1997) and DEPOMOD (Cromey et al. (2002).

There is no similar scientific concept for understanding how nutrient wastes from CAS distribute and accumulate in water column ecosystems, and there is a poor understanding of how these nutrients affect the structure and function of the pelagic ecosystem. For sure, water current velocity or hydrodynamic energy, and mixing depth modifies the environmental impact, but what is the assimilation capacity and the critical nutrient input rates of the pelagic ecosystem? This lack of quantitative knowledge means that there is no scientific base for monitoring and managing environmental effects of CAS on open waters, implying that environmental restrictions and management must comply with the principle of precautionary approach.

In the below sections, we emphasize particularly to establish a general understanding of pelagic impacts of CAS, including where and how they potentially will appear in space and time. We particularly point to how these scientific understanding should be made operational through further research and developments. We also describe the current scientific concept for managing benthic impacts of marine aquaculture.

4.2. The pelagic ecosystem

The studies made to assess pelagic ecosystem impacts of nutrient wastes released from fish farms, as well as from other anthropogenic sources, have generally revealed a lack of ecological responses in the plankton community. This is among others because there is no clear understanding of how potential effects will become manifested in pelagic ecosystems, there is in fact no general scientific concept established for how such potential effects must be traced and assessed (e.g., Cloern 2001). Because N and P are biogenic elements forming parts of all biomass, and not environmental toxins harmful for plant and animal health, these nutrients are rapidly assimilated in the microorganisms present in the water column during the summer to autumn period (e.g., Thingstad et al 1993). This is why for example ammonia, which is both the principal excretion product from all aquatic animals and the most common limiting nutrient for the phytoplankton, is not easily found in high concentrations in water masses around and downstream of cages. In addition to this rapid assimilation, the nutrients are, whether they are in their inorganic form or organically bound in biomass, dispersed downstream of the cage farms by hydrodynamics of variable patterns and strengths.

The inorganic nutrients released from CAS (i.e., NH_4 and PO_4) are the main driver for changing structure and function of the pelagic ecosystem, and thereby affecting the state and quality of coastal waters (Table 11). The nutrients associated to organic molecules, dissolved organic N and P, will as mentioned above have little impact because these components are biologically relatively inert, with very long lifetimes. Particulate nutrients will not affect phytoplankton, but small particles can be eaten by zooplankton or decomposed by bacteria and thereby have an indirect minor effect.

4.2.1. The pelagic ecosystem - assimilation and response to nutrients

The organisms forming the lower planktonic food web are the key actors in the assimilation of nutrients in the water column (e.g., Olsen et al 2007). The main group of organisms are:

- Phytoplankton - microscopic photosynthetic algae and cyanobacteria acting as primary assimilators of nutrients. The main taxonomic groups are picocyanobacteria, diatoms, dinoflagellates and a diverse group of small eukaryotic flagellates. Sizes typically varies between 1 to 100 µm, colonies of cells may be bigger
- Heterotrophic bacteria (and Archaea) - small organisms (<1 µm) that are accumulating dissolved organic compounds, but also inorganic nutrients like the phytoplankton
- Heterotrophic nanoflagellates (HNF) - small (4 – 8 µm) protozoan grazers that feed on picocyanobacteria and bacteria
- Microzooplankton - often dominated by ciliates in NE Atlantic waters (5 – 50 µm). Ciliates are protozoan grazers that feed on small phytoplankton mainly, and less important on HNF, the smallest ciliate individuals (5-10 µm) may feed on bacteria.
- Mesozooplankton - larger individuals (>100 µm, younger stages may be smaller) often dominated by crustacean zooplankton like copepods in NE Atlantic waters.

Enhanced nutrient supply results in a stepwise process where first step includes an increased inorganic nutrient uptake in phytoplankton (and bacteria), resulting in an increased nutrient content in the cells, and in turn an increased growth rate (i.e., primary production). If the zooplankton grazing pressure of phytoplankton is not too high, the phytoplankton biomass may accumulate and increase, leading to higher food concentration for all groups of

heterotrophs, and in turn to successive responses in their feeding activities and growth responses (bacteria take up dissolved carbon compounds from the algae and zooplankton defecation).

The typical responses in primary production and accumulated phytoplankton biomass in stagnant NE Atlantic coastal waters are illustrated in Figure 6. There is a close to linear response below N loading rates of 11-18 mg N m^{-3} day^{-1}. Open dynamic systems will respond differently, advection and vertical mixing will reduce the quantitative response (slope of curve) quite pronouncedly for biomass, dependent of the mixing rates (see arrows). The quantitative response in primary production is less sensitive than biomass for physical mixing processes and grazing (Olsen et al 2006).

The responses and trophic interactions of the heterotrophic planktonic food web following enhanced nutrient input are complex, but the bottom line is that the food web acts like a buffer which, within certain limits, may mitigate negative ecological effects of enhanced nutrient input, for example any extensive blooms of phytoplankton. Both structure and function of the planktonic food web will respond in a predictable way to increased nutrient supply, as illustrated in Figure 6, although not on species level.

In coastal areas, the primary production is usually limited by the availability of inorganic nitrogen or phosphorus (brackish waters) during the vegetation period. Algae and bacteria have the ability to store excess nutrients in the cells. Nutrient uptake may therefore take place well before the nutrients are incorporated in new biomass, and the time scale of biomass response following enhanced uptake of nutrients is some 3-5 days. The nutrient to biomass (e.g., carbon) ratio in phytoplankton and bacteria is a dynamic variable, which can vary widely depending on the nutrient availability in the water column (Sakshaug et al. 1983, Vadstein 2000).

The increased primary production represents an increased food availability for the heterotrophic plankton groups (i.e., zooplankton and bacteria). The main take home message of Figure 7, summarising changes in the main functional heterotrophic groups, is that enhanced nutrient supply affects the carbon flows to and in between the heterotrophic components more strongly than the biomass of these functional groups. The phytoplankton biomass (indirectly shown in Figure 6) is responding more strongly (Olsen et al 2006), as indicated by the increased availability as food for the zooplankton in the figure (input arrows).

Another apparent pattern of Figure 7 is that the bacterial part of the planktonic food web (the microbial food web, prokaryotes-HNF) is responding very little to nutrient addition. It is the larger groups of phytoplankton and grazers that primarily respond.

The most important message of Figure 7 is the very pronounced increase in sedimentation rate that follows enhanced nutrient input. This flow of dead organic matter to deepwater and sediments represent an organic loading that becomes important for the oxygen requirements and concentration in aphotic waters and sea floor communities. A non-linear, accelerating increase in sedimentation per primary production reflects the fact that the zooplankton grazers, at some point of nutrient input, are not any more able to consume, and efficiently remove, the enhanced primary production. A high sedimentation fraction of primary production is an indication that the planktonic ecosystem is no longer able to assimilate the enhanced nutrient input very efficiently (see below).

4.2.2. Critical loading rate of plankton ecosystem

All ecosystems have an inherent capacity of persistence, and smaller environmental changes are mitigated through adaptive responses of the organisms. Major changes in ecosystem structure and function, be it reversible of irreversible changes, will only take place if the environmental signal, or the environmental interaction, is sustained and too strong. For the planktonic ecosystem of the water column, we may deduce that nutrients are efficiently assimilated without any harm as long as the input rate remains below a critical upper level. The assimilation capacity of the water column ecosystem is mediated by two main mechanisms:

- Nutrient uptake and assimilation by phytoplankton, with successive trophic transfers of energy and materials (e.g., nutrients) in the planktonic food web to the higher trophic levels
- Dilution of nutrients and organisms mediated by hydrodynamics at production sites and their surrounding, downstream water masses

The dilution mechanism is independent of the organisms of the water column ecosystem; major physical forcing processes drive hydrodynamics. The assimilation capacity of the planktonic community are strongly dependent of hydrodynamics, because dilution leads to a reduction of nutrient concentrations and biomass, and therefore also to an increase in capacity of nutrient assimilation in the pelagic community. Both mechanisms are accordingly working in concert. Nutrient uptake and allocation in planktonic food webs and hydrodynamics are the fundamental processes determining the assimilation capacity of the water column of open and more protected coastal waters.

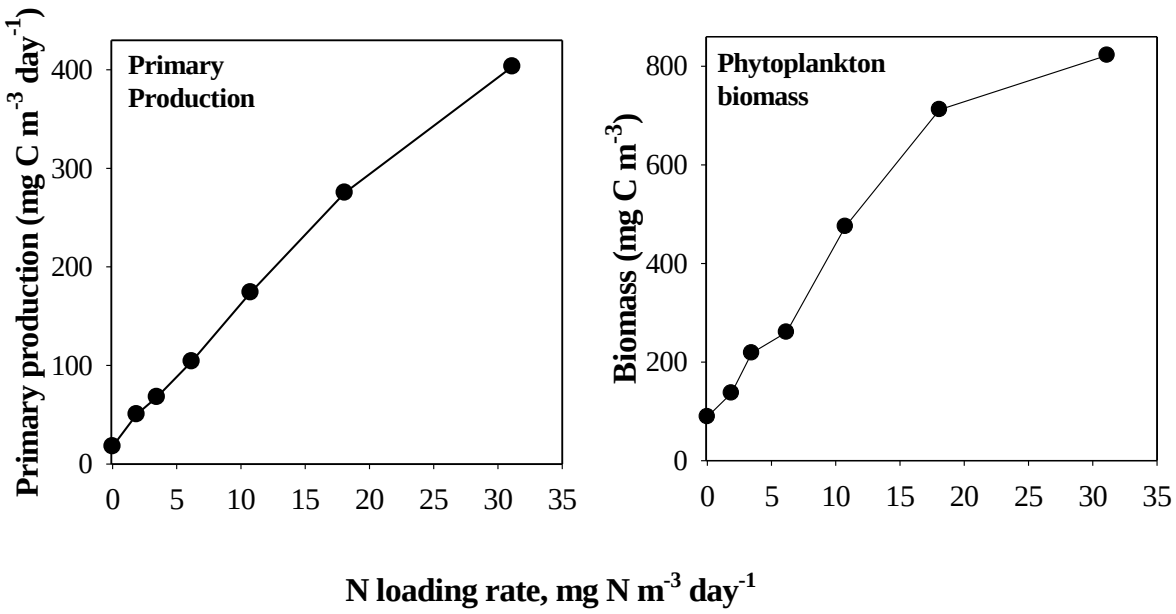

Figure 6. Primary production (left) and phytoplankton biomass (right) as functions of N loading rate in stagnant NE Atlantic plankton ecosystems. Arrows illustrates the sensitivity to rates of water advection and mixing (Norwegian data taken from Olsen et al 2006).

Looking more closely at the biological mediated assimilation mechanism, which is perhaps more cryptic than the physical, the upper panel of Figure 7 is representative for the normal, undisturbed situation in NE Atlantic coastal water. The food web organisms are capable to assimilate efficiently the nutrients input without major exports to sediments. On the other hand, the lower panel, reflecting a situation with a nutrient supply 4-5 times above the natural level, describes a situation where the nutrient loading rate exceeds the rate that can be efficiently assimilated in the planktonic ecosystem without major export to sediments. Somewhere in between the specified loading rates in Figure 7 (3-20 mg N m^{-3} day^{-1}), there must be a critical nutrient loading rate (CNLR) which cannot be exceeded without loss of ecosystem integrity.

There is so far no generally accepted method to determine a CNLR for coastal waters experimentally or empirically, and there is no published value, as far as we understand. There is, however, evidence showing that the primary production and the zooplankton feeding rates in stagnant systems both levels off for volumetric loading rates above around 1 mmol N m^{-3} day^{-1} (14 mg N m^{-3} day^{-1}) in NE Atlantic coastal waters (see Figure 6 above and Olsen et al 2006). It is also apparent that the percent sedimented C of total primary production reaches values above 30% in mesocosm experiments above this point (37% for situation in lower panel of Figure 7, 3-week experiment).

4.2.3. Modification by hydrodynamic forces

The volumetric loading rate of inorganic nutrients is defined as mass of nutrients released per volume of water and day. The pattern of water currents in coastal waters is complex and cannot easily be measured, even after extensive field surveys. As a first approximation for estimating the volume that nutrients from CAS are drained into in the course of one day, we may assume that water are drained through the cage farm in a plug flow pattern with no further mixing downstream of the farm site and no tidal oscillations. There is no standardized size of a CAS, and our further exercise assumes a farm of 160 x 320 m^2 with depth 15m (area and volume of 51,000 m^2 and 768,000 m^3, respectively). If water enters the cage area directly from the length side, and there is no major resistance in the cages, Figure 8 shows the number of water exchanges and the resulting total volume passing the cage farm as a function of the average daily water current velocity. Even relatively slow water currents will result in a high water renewal rate and exchanged water volume. This is for sure an underestimate of the real receiving volume, because the nutrients will be continuously mixed with other water masses downstream of the fish farm. The exercise illustrates how important hydrodynamics are for the nutrient loading rate and the nutrient dispersion patterns, illustrating that hydrodynamics are instrumental to mitigate negative environmental effects of nutrients released from fish farms, as for any other point sources of nutrients.

This above exercise demonstrates that advanced 3D hydrodynamic modelling is needed to estimate drainage volume, spreading pattern of nutrients originating from point sources, and the volumetric loading rates in dynamic coastal waters (see below).

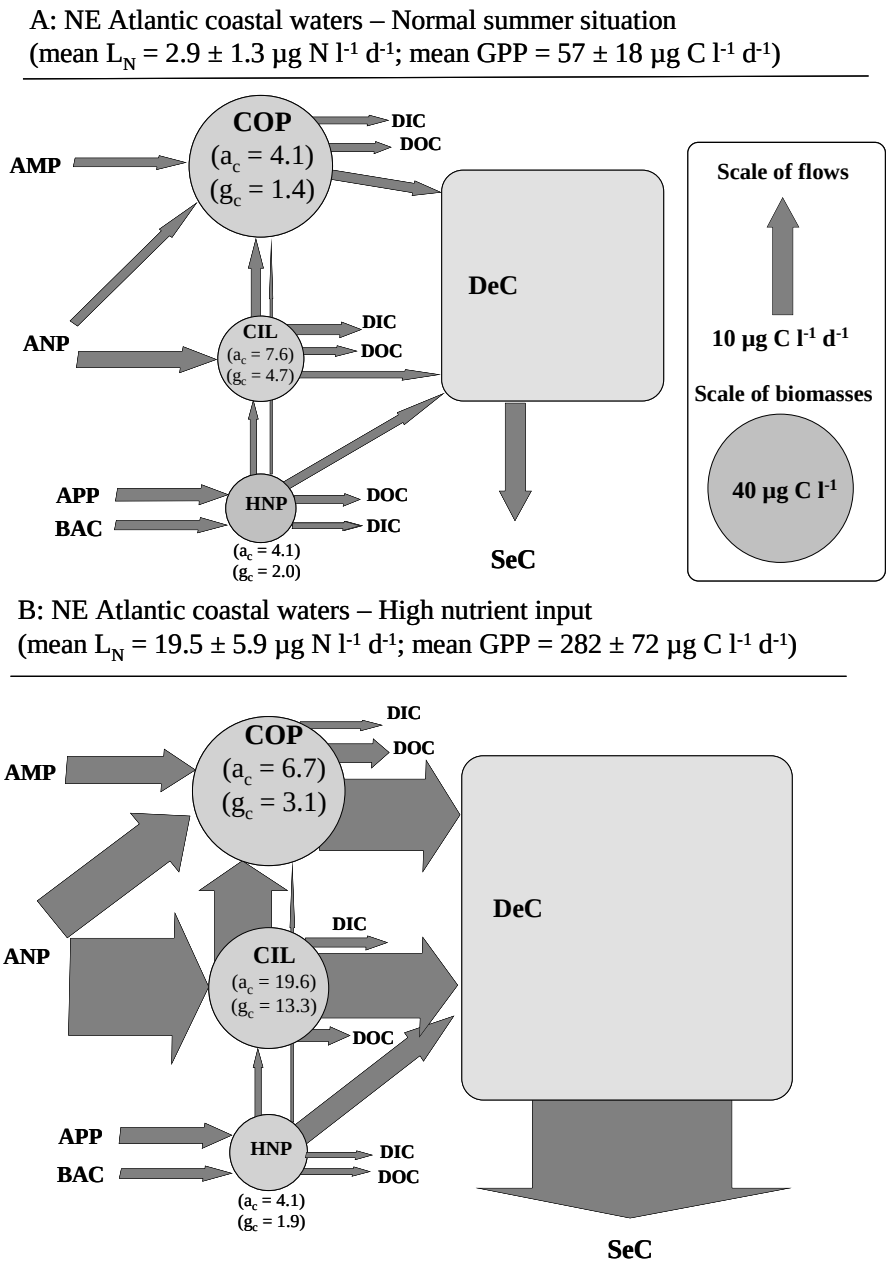

Figure 7. Schematic view of carbon flow networks during (A): a normal summer situation in NE Atlantic coastal waters and (B): conditions of high nutrient input. Arrows shows flows, boxes show biomasses (and their allocation of energy). AMP: Feeding of large sized phytoplankton (20-200µm), ANP: Feeding of medium sized phytoplankton (2-20µm), APP: Feeding of small sized phytoplankton (<2 µm); HNP: Heterotrophic nanoplankton (<20µm); CIL: ciliates, main constituent of microplankton; COP: Copepods, main constituent of meso-zooplankton; DIC: CO2 release (respiration); DOC: released of dissolved organic components; DeC: release of particulate organic components; SeC: sedimented carbon; a_c: assimilation rate; g_c: growth and reproduction rate. All concentrations are expressed in terms of µg C l^{-1} and rates as µg C l^{-1} d^{-1} (from Olsen et al 2007).

4.2.4. Trophic state assessment and time lags of responses

A major experienced problem of diagnosing trophic state in pelagic ecosystems is the very fast accumulation of the inorganic nutrients into phytoplankton biomass (see above) and the dispersion of released nutrients and phytoplankton biomass caused by hydrodynamic forces. Different from the benthic ecosystem, which accumulates nutrient wastes in a particulate form, both inorganic nutrients and the planktonic organisms becomes dispersed in a relatively unpredictable way. The consequence is in most events that any biological response of an increased nutrient supply will become realized downstream of, and sometimes far downstream of, the nutrient source. At that point, nutrients are normally diluted and, dependent of the dynamic state, often no longer detectable above the background level.

Inorganic N and P are taken up in phytoplankton within minutes or up to one day, dependent of the extent of nutrient limitation in the phytoplankton. Assessment trials must consider all the time lags of the response chain between the enhanced nutrient supply and the realization of the nutrients in *de novo* phytoplankton biomass. These steps involve:

- Phytoplankton uptake of NH_4^+ and PO_4^{3-}, both macronutrients which normally are limiting for growth, is immediate although dependent of the nutritional state of the phytoplankton and its biomass (minutes to day). Uptake causes an immediate increased endogenous nutrient concentration in the phytoplankton cells (increased nutrient per biomass). Inorganic nutrients in the water may accumulate temporarily only if the nutrient input rate is high and sustained (e.g., in stagnant cages, Olsen et al 2007).
- The primary production of the phytoplankton can respond and increase 2-5 days after an increased supply of nutrients. The time lag reflects the physiological adaptation of the cells and is dependent of the initial nutritional state of the phytoplankton.

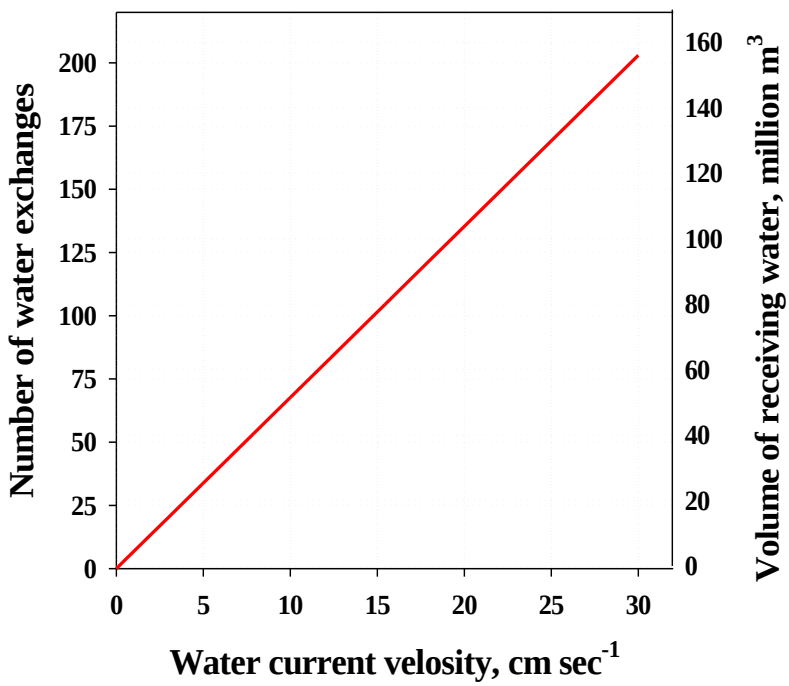

Figure 8. Water exchange and volume of receiving water of hypothetical CAS as a function of the water current velocity.

- Measurable accumulation of phytoplankton biomass can be detected 1-2 days after the increase in primary production.

An increased phytoplankton biomass, following enhanced nutrient supply from an upstream point source, will accordingly become expressed as an increased biomass as late as 3-7 days after the phytoplankton have assimilated the nutrients. In the case of strong hydrodynamics, the potential response in phytoplankton biomass will then normally take place 3-7 days downstream from the emission point, the fish farm. The excess nutrients from the fish farm (inorganic + accumulated in phytoplankton) will also be strongly diluted at that point. This is important, because efficiently diluted nutrients will have neutral or positive effects on the pelagic ecosystem. These nutrients will enter the major bio-geochemical cycles of the sea and in general terms stimulate natural production and fisheries.

4.2.5. Monitoring techniques

Monitoring techniques, which allow assessment of pelagic waters over a wider geographic scale, are paramount for detecting potential impacts from nutrient sources like CAS in coastal waters. The most apparent options are regional scale satellite imaging and 3D hydrodynamic or ecological modelling (Figure 9). Satellite images can provide real situations for phytoplankton blooms in surface waters at any given time, but it cannot easily distinguish between the production driven by the highly variable natural nutrient supply and the optional anthropogenic signal. 3D hydrodynamic-ecosystem models produce a virtual world, not a real one, but it can cover the entire water column continuously with time over any geographic region. Moreover, modelling allows us to distinguish between natural and anthropogenic signals, and it can potentially predict phytoplankton production and effects also on higher trophic levels. Models can be run with and without nutrient emission from fish farms included, and the difference, termed the “excess” nutrient concentration, phytoplankton production, or phytoplankton biomass can be estimated right away. Such models are also well suited to assess the integrated effects of all CAS and other nutrient sources located in a region, and accordingly instrumental tools for a future ecosystem based management of aquaculture. Classical measurements must be used to validate the major trends found by satellite images and modelling at specific locations.

4.2.6. Preliminary operational scheme for management

Advance 3D modelling is not always practical for daily environmental management of CAS, but such modelling can generate the more simplified means needed for management and monitoring exercises, and particularly for localisation of CAS in coastal water.

As discussed above, the dilution of a nutrient point source by hydrodynamic forces will continuously reduce the nutrient concentrations, and therefore the volumetric loading rate of the water masses downstream of the farm. Assimilation by organisms and hydrodynamics will therefore work in concert and together determine the overall assimilation capacity of nutrients in every water packages of coastal waters. Yokoyhama et al (2004) have suggested a proxy (ISL) expressing the assimilation capacity represented by physical forces:

(10) $ISL = D*V^2$

where D is the depth and V is the water current velocity. The proxy was primarily tested for benthic communities, but it can be useful also for pelagic, with depth interpreted as the depth of the mixed water layer.

Figure 10 illustrates a scheme in which experimental and empirical results, including results obtained through 3D modelling, can be aggregated to form an operational, graphical tool for risk management of the water column ecosystem. In the "*water current velocity – nitrogen loading rate"*- space described as Area I, water dynamics are strong enough to maintain nutrient loading below the limits of the assimilation capacity of the water column ecosystems. In Area III, the nutrient loading rate exceeds the limits of the assimilation capacity of the pelagic ecosystem, the loading rate is then above the critical level (CNLR), and the water column ecosystem can loose its integrity. Sedimentation rates are strongly enhanced, which may cause harmful coastal eutrophication. In the space described as Area II, the loading rate is coming close to the CNLR where nutrient loading exceeds the assimilation capacity. These situations represent increased risks and calls for special attention and a precautionary approach. The solid curve expresses that the CNLR will increase with strong hydrodynamics.

The concept illustrated in Figure 10 is preliminary, slopes and exact x-axis intersections of the indicated lines are not adequately known. Moreover, the axis legends are so far accidentally chosen. Moreover, the proxy ISL, the water dilution rate, or any measurable variable expressing the energy of hydrodynamics can replace water current velocity on the y-axis. The volumetric loading rate can be translated into a standing stock of fish in a given CAS or region. An ultimate R&D challenge is to examine different response variables for hydrodynamic energy and to quantify the boarders of the areas in Figure 10.

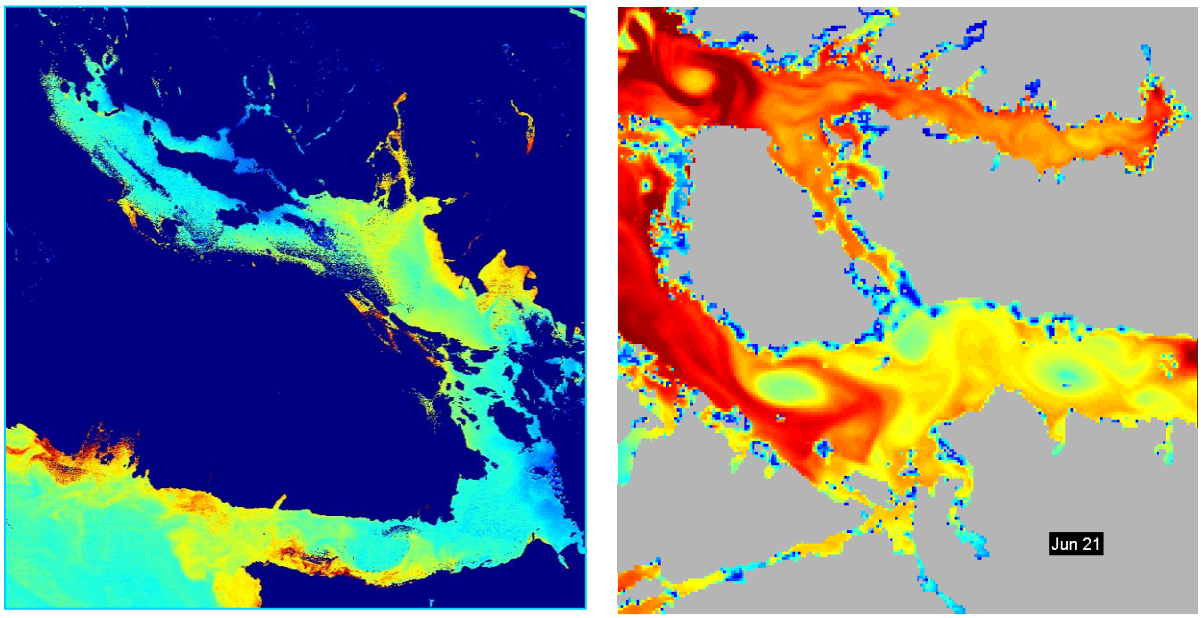

Figure 9. Examples of a satellite image and 3D modelling of coastal waters. The satellite image (left) from Southern Vancouver Island, British Columbia shows phytoplankton fluorescence from MERIS orbital platform. The 3D modelling is made for a coastal region in Vesterålen, Northern Norway, red colour reflects high, yellow intermediate and blue low phytoplankton production per m^2 per day (Satellite image provided by Stephen F. Cross, model data by D. Slagstad).

4.3. The benthic ecosystem

The benthic ecosystem is affected by particles generated by aquaculture systems (Table 6). These particles include uneaten wastes of feed, which sink rapidly to the bottom, and larger faecal particles generated by the fish as well as sinking of material from pelagic production and debris from fouling of the net cage structures. A fraction of the waste particles will be small in size, and will become decomposed by zooplankton and bacteria in the water column and not accumulate in sediments.

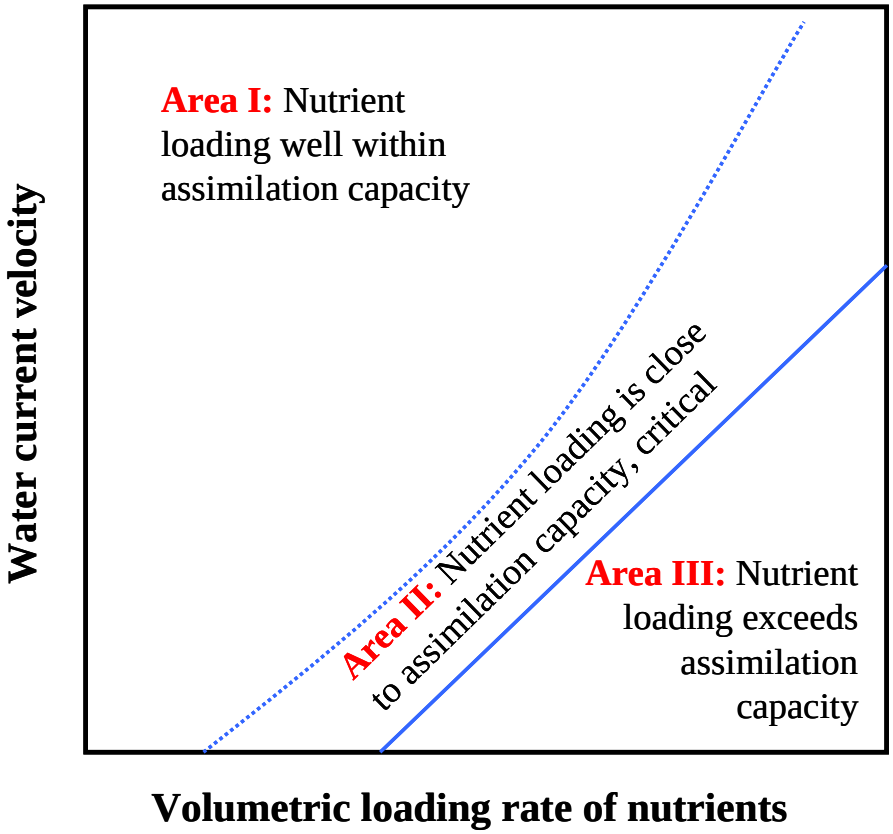

Figure 10. Conceptual relationship describing the ability of the water column ecosystem to assimilate nutrient input as a function of the volumetric loading rate of inorganic nutrients and the water current velocity. Area I: Water dynamics are strong enough to maintain nutrient loading within the limits of the assimilation capacity of the water column ecosystems; Area II: The critical zone where loading rate is coming close to the critical nutrient loading that exceeds assimilation capacity. Situations represent increased risks and calls for special attention and a precautionary approach; Area III: Nutrient loading rate exceeds the limits of the assimilation capacity; the water column ecosystem can loose its integrity, which may cause harmful coastal eutrophication. The figure is preliminary, slopes and exact x-axis intersections of the indicated lines are unknown.

4.3.1. Food web structure and function response to enhanced loading

In coastal environments, benthic processes mediate up to 90 % of the degradation of organic carbon (Jørgensen and Richardson 1996). The thickness of the oxic surface layer of the seabed varies from a few millimetres in near shore and sheltered coastal areas, to 1 meter, or more in offshore sediments (Murray et al. 1980, Revsbech et al. 1980, Jørgensen 1982). The organic carbon supply to the sediment originates from the autotrophs in sediments and material settling out of the water column above. The latter process will dominate because light is limited at depth. Photosynthetic organisms can exist only at the sediment surface and a few mm down in crevices (Megonigal et al. 2004).

In the oxic layer, aerobic heterotrophs, including bacteria, protozoa and metazoans, are the main decomposers of organic material (Megonigal et al. 2004). The term infauna, meaning the animals living inside the sediment, not just on the surface, is often used when referring to the communities in the oxic layer. The infauna is divided into microfauna; protozoans and small metazoans, meiofauna; small invertebrates that can pass through a 1 mm mesh, but is retained on a 45 µm mesh, and macrofauna; organisms ≥ 1 mm (www. Wikipedia.org 2007).

In general, infaunal animals become rarer with increasing water depth and distance from the shore due to decreasing supply of organic matter to the sediments. Bacterial numbers are more or less constant, at approximately 10^9 cells per ml interstitial seawater (www. Wikipedia.org 2007). All animals, except some protozoan species, are aerobic, i.e. they rely on aerobic respiration with oxygen as the terminal electron acceptor ($CH_2O + O_2 \rightarrow H_2O + CO_2$). This means that the whole community, or the food web of micro, meio and macro-fauna, can only survive in the oxic sediment layer. In an intertidal bay, the contribution to total benthic community respiration in the upper five centimetres was as follows:

bacteria > macrofauna > meiofauna

and bacterial respiration represented up to 88 % of total respiration (Hubas et al. 2006). Nevertheless, the fauna is important for sediment metabolism. Predation keeps bacterial productivity high through biomass removal and regeneration of nutrients, and macrobenthos affect the re-mineralization of organic matter through burrowing and ventilation activity (Aller and Aller 1998). Burrows of shrimps and polychaetes irrigate the sediment (Webb and Eyre 2004, Kristensen et al. 1985) and increase the penetration of oxygen, thereby stimulating aerobic respiration and coupled redox reactions like nitrification-denitrification (Aller and Aller 1998) and reoxidation of reduced compounds (Banta et al. 1999).

There are constant anoxic conditions deeper into the sediment. In this layer, mainly sulphate-reducing bacteria ($SO_4^{2-} + CH_2O \rightarrow HS^- + 2HCO_3$), and deep in the sediment (>2m) methanogens ($CH_2O \rightarrow CH_4 + CO_2$), perform the breakdown of organic material. Sulphate reducers are usually the dominating group (Martens and Val Klump 1984). Sulphur bacteria and reoxidation processes in the oxic/anoxic boundary layer consume most of the hydrogen sulfide produced, which is usually not released directly to the water column (Megonigal et al. 2004). Also the anoxic metabolism is affected by the burrowing action of the fauna. Irrigation increases the transport of solutes to and from the anoxic layers, resulting in increased metabolic activity and increased regeneration of e.g. phosphate and ammonium (Rysgaard et al. 2000, Aller and Aller 1998).

When excessive amounts of organic material and nutrients accumulate in the top layer, the aerobic heterotrophs deplete the oxygen. This in turn results in a reduced activity of these organisms. In this situation, the sulphate reducers and the methanogens can totally dominate the microbial community also in the top layer, and this may lead to out gassing of hydrogen

sulphide (H_2S) and methane (CH_4) from the sediments (Megonigal et al. 2004). These gases are harmful to fish at high concentration.

Holmer et al. (2003) have summed up the main effects of excess aquaculture waste loading in the sediments below fish farm cages as follows: "*When electron acceptors like oxygen, manganese and iron are depleted the sediments become highly reduced due to sulphide accumulation. Decreasing diversity of the benthic fauna and flora reduce the chemical and physical modifications of the surface sediments, because the decomposition of organic matter is shifted to microbial processes. Oxic processes like nitrification are inhibited, and this may also limit denitrification due to low nitrate concentration. The result is high ammonium release from fish farm sediments*." Similarly high phosphorus release can be expected due to the exhaustion of the P-binding capacity through oxidized iron. Enrichment of sediments with organic matter thus enhances the benthic-pelagic coupling of inorganic nutrients and potentially stimulates phytoplankton growth as discussed above.

In areas with large clarity of the water and long growth seasons, such as in the Mediterranean and in oligotrophic trophic coastal zones, it is important to consider the habitat structure in more detail. Under high light conditions, benthic vegetation dominate the benthic habitat, and seagrass and coralline red algae communities are very sensitive to deposition of waste products from fish farms (Wilson et al. 2004, Marbá et al. 2006, Frederiksen et al. 2007). Similarly to the benthic infauna, the vegetation suffer from changes in the sediment biogeochemical conditions through anoxia and sulphide, but nutrient enrichment may also affect the plants by increasing epiphytic growth and overtaking by faster growing primary producers such as macroalgae (Holmer et al. 2003b).

4.3.2. Classification of farming sites

Different classification systems of the benthic impact of CAS have been worked out and tested. Carroll et al. (2003) performed an environmental survey of the sediments under 80 sites of salmon cage farms in different parts of Norway during the years 1996-98. They classified the site sensitivity according to average water current velocity (Velvin 1999; Table 12) and the environmental impact based on organic carbon (TOC) concentration in the sediments (Table 13), according to the environmental quality standards (EQS) of the Norwegian Pollution Control Authorities (Molvær et al. 1997). In their survey, 27 % of the salmon farms had a feed consumption exceeding 1000 metric tons per year, whereas 54 % less than 500 mt per year and can be considered as small farms under present conditions[3]. Only 10% of the farm sites had average water current velocities in the least sensitive class, while 75 % had speed in the moderately and very sensitive classes, and 59 % of the farmers used fallowing (abandoning) to let the sites recover before restocking.

Several other countries have developed similar classification systems, which also are used for initial site selection for the new farms. Characteristic for these systems are consideration of hydrodynamic regime, habitat complexity and sedimentation patterns. Whereas shelter was one of the most important factors in the initial stages of CAS farming, sites with weak currents and high natural sedimentation are now avoided promoting off-coast and exposed locations (Holmer et al. 2007).

[3] Production is far higher now. The largest salmon farms produce almost 10,000 tonnes per year, this can be produced in some 10 large cages (1000 tonnes per cage and year). The production per far is still highly variable, but very seldomly as low as 1000 tonnes.

Table 12. Sensitivity classification based on water current velocity, from Velvin 1999.

Water current velocity (cm s^{-1})	**Class**	**Classification**
< 3	4	Very sensitive
4-6	3	Moderately sensitive
7-10	2	Slightly sensitive
10-25	1	Not sensitive

Table 13. Sensitivity classification based on TOC of sediments, from Molvær et al. (1997).

[TOC] (mg/g sediment)	Classification	% of sites
< 20	Excellent	16
20-27	Good	34
27-34	Intermediate	18
34-41	Poor	12
> 41	Very poor	20

4.3.3. Assessment methods for trophic state of sediments

There are several methods developed for assessment of the trophic state of sediments below aquaculture sites. The MOM (Modelling – Ongrowing fish farms – Monitoring) is an operational management method developed in Norway (Ervik et al. 1997). The MOM system regulates the local environmental impact of marine CAS relative to the holding capacity of the sites. The aim of the system is to integrate environmental impact assessment, monitoring and environmental quality standards (EQS). The degree of monitoring of a site is dependent on the degree of exploitation, i.e. how much a site is utilized (Ervik et al. 1997).

The MOM monitoring system includes three types of investigation of increasing accuracy. The A investigation measures sedimentation rate under the fish farm net cages. The B investigation provides a trend monitoring of the sediment condition, and the C investigation is a comprehensive study of the benthic macrofaunal community structure (Hansen et al. 2001).

DEPOMOD is a computer particle-tracking model developed by Cromey et al. (2002) to enable prediction of the impact from marine fish farms on the benthos. The input is loss rate of uneaten food, faeces production, and hydrodynamics. The model predicts the initial deposition of particles on the seabed, and an added re-suspension model redistributes particles according to near bed current flow. The model has been validated by using sediment trap studies. Observed Infaunal Trophic Index (ITI) and total individual abundance have been used to assess the impact of the predicted accumulation of organic waste on the fauna (Cromey et al. 2002), and if the sampling and the modelling deviate from each other, field conditions are

up for a more detailed examination. As for the MOM system, monitoring operates within different zones around the farms, where sites at increasing distance from the farms are allowed different degrees of benthic impact (Cromey and Black 2005).

Despite the extensive knowledge on the benthic impacts, there is major confusion as to which monitoring strategies are the best, and the above-mentioned are just two examples, but many more can be added from other countries. The current research efforts are within an optimisation of sensitive and cost-efficient indicators. The EU funded ECASA project (www.ecasa.org.uk) has assessed several benthic and sediment indicators, and conclude that no single "magic-bullet" indicator exists. Rather a suite of indicators should be evaluated in order to correctly interpret the sediment state, if an inappropriate indicator set is chosen then it is quite possible to draw misleading conclusions (Mulsow et al. 2006). Although this is unfortunate in that it often does not allow direct numerical comparison between countries, in general, similar qualitative information on sediment state (i.e. position on the Pearson – Rosenberg continuum) can be derived if a sufficiently broad range of indicators have been evaluated. Previously, faunal indices were strongly supported, but due to the rarity of taxonomic experts and the cost related to these analyses, other strategies are now under investigation (Kalantzi and Karakassis 2006). This could be simpler analysis based on indicator species (Kalantzi and Karakassis 2006), sensitive sediment methods such as SPI (Karakassis et al. 2002) or simple biogeochemical measures (Brooks and Mahnken 2003, Holmer et al. 2005, Hyland et al. 2005, Aguado-Gimenéz et al. 2007, Hargrave et al. 2008).

5. Research activity on eutrophication related to aquaculture: a literature survey

A survey of the international peer reviewed literature from the years 2000-2006 derived using the ISI Web of Science databases yielded 3914 references on the search word "aquaculture" (Table 14). The number of citations has increased almost linearly during the last two decades, with 40 references found for 1990 and 838 for 2006 (Figure 11). The fact that most scientific publications were not digital before the late 1990's may to some extent skew the impression, but it is anyway clear that the research activity related to aquaculture has increased. This increase is certainly related to the steady increase in aquaculture production on a world basis (Figure 12).

The newest 25 references on "aquaculture" cover a diverse set of topics like diseases and use of antibiotics, larval development, growth, morphology, genetics, molecular biology, effect of turbulence and salinity, ecology, economy, production of lipids and fatty acids, and waste treatment. The last topic leads towards the issue of this review; a synopsis of information on eutrophication effects related to marine aquaculture. The main focus is on the release of nutrient from aquaculture and its potential effects on the pelagic and benthic ecosystems of coastal waters and the effect that these nutrients indirectly may have on marine aquaculture (i.e., to be understood as aquaculture in brackish and seawater undertaken in coastal waters or in more open ocean locations).

A search on the key word "eutrophication" yielded 3705 references, while "coastal AND eutrophication" and "marine AND eutrophication" yielded 903 and 621 references, respectively, in the ISI databases from the years 2000-2006 (Table 14). In contrast, only 45 publications were found when combining the search words "coastal AND eutrophication AND aquaculture" for the same years, while "aquaculture AND eutrophication" yielded 92 references (Table 14). Thus, research on coastal eutrophication is a considerable part of the total eutrophication research, while eutrophication in relation to aquaculture has been studied

in 10% of these published papers. Based on the quantitative international research output, eutrophication related to aquaculture on a global basis is not a big issue compared with other aspects of eutrophication.

An ISI search on the phrase ”aquaculture AND waste” resulted in 227 references for the years 2000-2006. Of the newest 100 cited papers, 36 were from ponds, tanks, or semi-closed systems. Of these papers, 25 treated waste management and 7 effects or accumulation of waste. A number of 32 of the retrieved articles treated cage cultures or other open production systems. Of these 32 papers, 21 covered effects on the surrounding environment or on the culture environment itself, while 6 publications were about waste treatment. The phrase ”aquaculture AND effluent” yielded 174 references (Table 14). Of the 100 published latest, 57 treated aspects of pond/recirculation culture systems. Of these 42 were about effluent treatment, 10 about effects of the effluent on the culture system or the environment. A number of 18 citations were about cage or open production, off which 14 covered effects or impacts of effluent and 4 optional treatments.

Of the 92 references covering “aquaculture AND eutrophication”, 51 treated open marine aquaculture sites, i.e. fish cages or mussel farms. Of these papers, 40 were on pelagic environmental effects, 22 treated benthic impacts, while 11 covered both pelagic and benthic eutrophication effects.15 references treated ponds and tank systems, and 26 covered other miscellaneous topics (Reference 1 to 92 in aquaculture and eutrophication list). A search on “pond AND aquaculture” and “cage AND aquaculture” yielded 193 and 65 references, respectively.

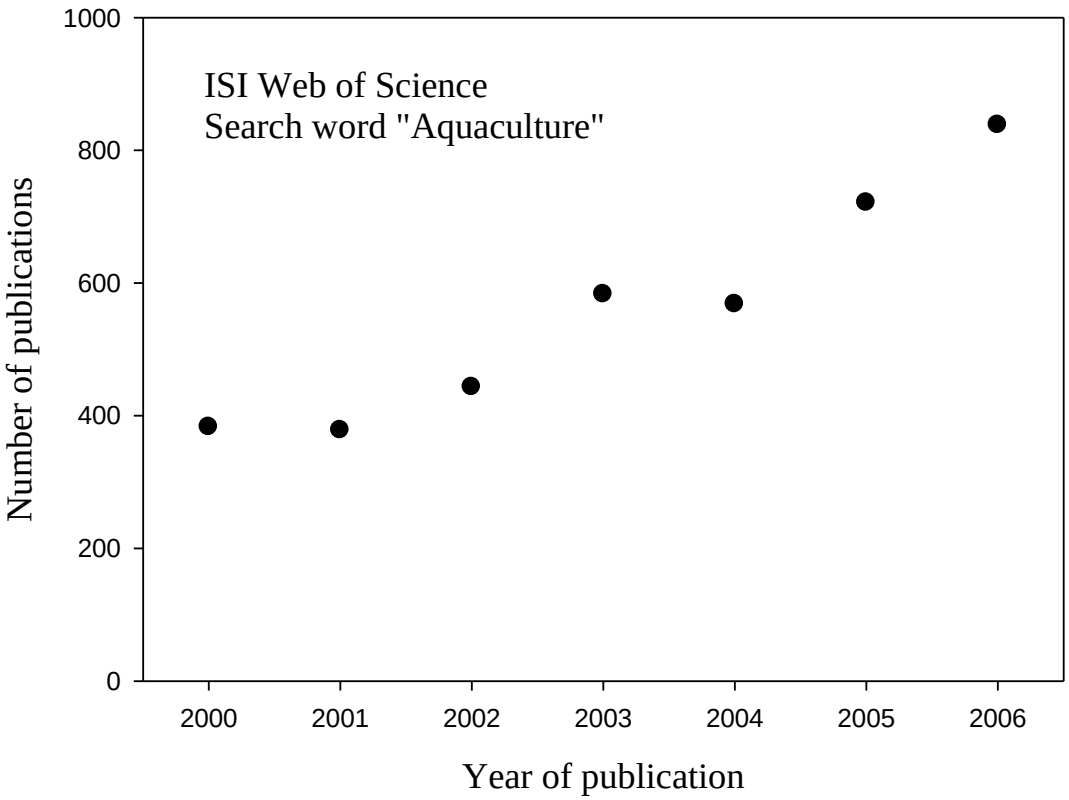

Figure 11: The number of references retrieved from the ISI Web of Science databases using the search word ”aquaculture” for the years 1990-2006.

From this survey of the recent peer reviewed international literature, it can be conclude that environmental interactions of closed or semi-closed aquaculture systems are more thoroughly studied than open systems. This is not very surprising, considering that 43 % of total global aquaculture production is freshwater fish, mostly reared in tanks or pond systems (Primavera 2006). Marine diadromous fish and crustaceans reared in open pelagic waters constitute 11 % of the global production. Molluscs and other animals constitute 24 % of the total production.

The research approach in closed or semi-closed aquaculture systems is mainly from an engineering point of view. It aims at solving the problems related to waste accumulation in the tanks or ponds, in order to keep productivity stable and high, but also to avoid serious nutrient discharge to the outside environment. In a sense, the approach is comparable to sewage treatment. For open systems, like CAS and mussel farming using long-line cultures, the focus has been on quantification of waste discharge, and the effects of waste effluents on the aquaculture system and the surrounding environment. The research efforts, however, have been very limited up to now.

6. Eutrophication due to aquaculture: synthesis of observations

This section summarise some specific issues, studies and literature related to measured or modelled environmental impact of aquaculture on the coastal environment. Appendix Tables 1 -3 below summarise important references obtained for specific search phrases in the ISI Web of Science databases for the years 2000-2006 (see legends of table), and is relevant for different sections below.

6.1. Aquaculture systems release nutrients

In closed or semi-closed aquaculture facilities, loading rates of nutrients can easily exceed the critical levels, and the aquaculture facilities must be managed holistically on the premises of the ecosystem to prevent negative impacts on the cultured organism, be it toxicity of too high ammonia concentration, anoxia, or blooms of toxic algae. Harmful impacts inside the fish farm can generally be alleviate through increased water exchange with the outside environment or alternatively, through various methods of effluent water treatment. The first method may induce eutrophication effects in the ecosystem receiving the wastewater, while the latter method may alleviate such eutrophication impacts on the outside environment if particles or nutrients are removed from the effluent (Gelfand et al. 2003). On the Italian west coast, for example, intensive land based fish farms rearing European sea bass and gilthead sea bream released 265 kg of nitrogen and 13 kg of phosphorus per day to the nearby lagoon. This resulted in extensive growth of opportunistic green seaweeds (Porello et al. 2003). By redirecting the effluent through algal treatment ponds for up to 8 hours prior to discharge, the PO_4 concentration was reduced by 15 %, while inorganic nitrogen was reduced by only 5 %, although very active nitrification converted much of it to nitrate (Porello et al. 2003).

Table 14. Number of retrieved references for different search phrases applied in the ISI Web of Science databases for the years 1990-2006.

Search phrase	2000	2001	2002	2003	2004	2005	2006	2000 - 2006	1990 - 2000
Aquaculture	383	378	443	583	568	721	838	3914	2138
aquaculture&environment &impact	9	10	11	23	11	20	17	101	44
aquaculture&pelagic& impact	1	1	3	2	2	3	1	13	5
aquaculture&benthic& impact	4	10	8	9	6	9	16	62	18
aquaculture&waste	15	16	24	40	28	48	56	227	125
aquaculture&effluent	20	16	7	35	25	36	35	174	78
aquaculture& eutrophication	6	6	9	20	15	14	22	92	29
pond&aquaculture	36	28	36	54	48	39	56	297	193
cage&aquaculture	11	12	23	26	19	35	41	167	65
eutrophication&algal& bloom	10	9	11	16	16	16	28	106	38
aquaculture&algal&bloom	0	4	1	3	4	6	6	24	10

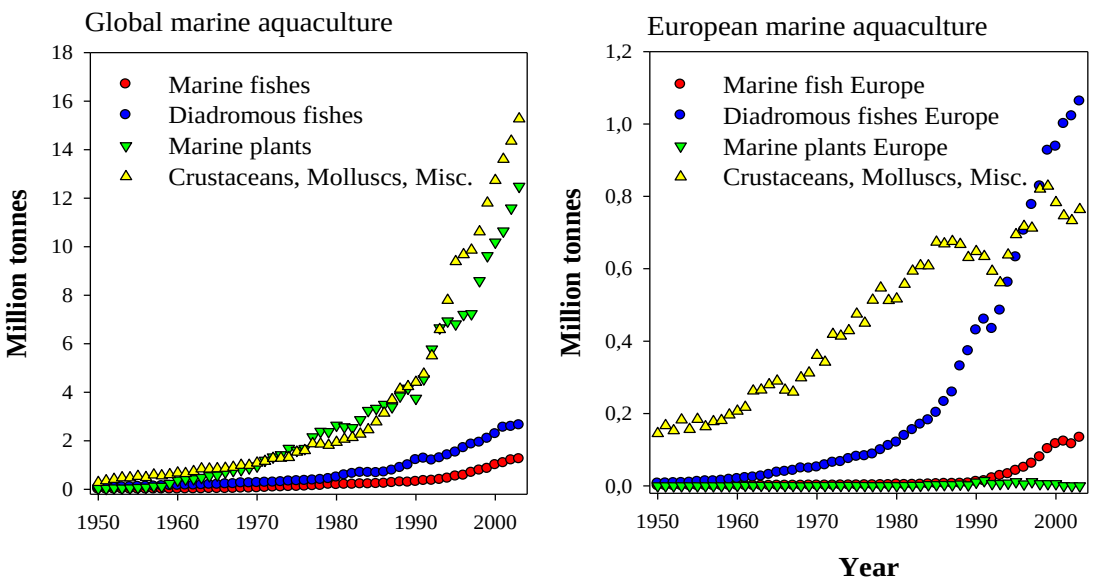

Figure 12. Historical development of some main groups cultured in the sea. A: Global production values and B: European production values (FIGIS, FAO statistics).

It is obvious that CAS represent point sources of nutrients and organic wastes affecting benthic and pelagic ecosystems, but this is also the case for semi-closed systems like recycling fish farms (RAS) and ponds, as described above. The main difference is that wastes from for example RAS is controllable, at least in theory. To what degree nutrient emissions from fish farms may result in harmful eutrophication depends on production scale, production management, including feed composition and conversion efficiency, hydrodynamics of the site, and sensitivity of surrounding habitats. It is important that the natural marine bio-geochemical fluxes of nutrients on a regional or global scale are much bigger than the nutrient discharges from fish farms. This is, however, not a protection against local harmful environmental impacts at or downstream of a production site.

6.2. Regional seas

Marginal seas and regions characterised by high population density are often considered to be more sensitive to harmful eutrophication than the major oceans. Gyllenhammar and Hakanson (2005) categorized the impacts of aquaculture nutrient release in the Baltic Sea according to size scales. The smallest scale, <1 ha (10,000 m^2) was defined as the "footprint" area of the fish farm. This area, which is slightly larger than the area of the fish farm because of water currents, suffers normally from benthic impacts due to deposition of waste material. The "local scale" corresponds to a size between 1 ha to 100 km^2. On this scale, the authors defined the boundaries of coastal areas using GIS (Geographical Information System) and constructed a mass balance model for water exchange in surface and deep water. A simple loading diagram was established from regressions relating response measured as nutrient concentration, chlorophyll a concentration or Secchi depth, to the nutrient loading from the fish farms. Sensitivity of the coastal area was defined as the ratio between the coastal volume and the theoretical surface water retention time. This enabled an estimate of maximum allowable fish production in the area. The regional scale was defined to be 100-10,000 km^2. The authors argued that it is possible to obtain net removal of nutrients by using more than 1.1 units of regionally caught wild fish to rear 1 unit cultivated fish on this scale. Anyway, in agreement with our general discussion of pelagic impact in coastal waters in Chapter 4, the effects of increased primary production are most likely to appear on local or regional scales, due to differences in the water exchange by coastal currents depending on local topology, depth and other variables. At geographic scales >10,000 km^2, the contribution from fish farms to the overall nutrient fluxes are small in the Baltic, according to the authors.

Increased chlorophyll *a* concentrations has been linked to fish farming activities in the Archipelago Sea in SW Finland and in particular in non-stratified shallow waters, where regeneration of nutrients in the sediments contributed to nutrient loading of the water column (Honkanen and Helminen 2000).

In the Mediterranean Sea, which is considered to be an oligotrophic ocean (Krom et al. 1991), the recent relocation of many coastal fish farms to deeper, more exposed offshore localities, has reduced the impact on sediment and water quality (Maldonado et al. 2005, Pitta et al. 2006). Karakassis et al. (2005) estimated that total nutrient discharge from aquaculture is less than 5 % of total nutrient runoff from human sources in the Mediterranean. Belias et al. (2003) concluded that no eutrophication incidents due to aquaculture have been identified in the eastern Mediterranean. However, Machias et al. (2006) found an increase in fisheries landings in the same area and found that it was strongly correlated to fish farming activities, suggesting a fast incorporation of the nutrient released from farms to higher trophic levels.

China has the largest aquaculture production in the world, with a mariculture production (marine plus brackish waters) of 23.5 million metric tonnes in 2005, off which marine plants

(10.8 million metric tonnes) and molluscs (10.7 million metric tonnes) are the main components (FAO-statistics). There is an increasing concern about the environmental impact of aquaculture in China, including the environmental interaction with aquaculture by other polluting industries and aspects of food security. According to Dong et al. (2006), "eutrophication due to aquaculture and other sources jeopardize aquaculture". Xie et al. (2004) have reported significant pollution based on water quality criteria of the local legislation. Inorganic nitrogen was found to be the most important pollutant. This concern on harmful eutrophication has prompted many suggestions for treatments that may alleviate environmental impacts. These include co-culture with macroalgae, which remove excess nutrients through uptake and growth (Yang et al. 2006, Zhou et al. 2006b, Fei 2004) or co-culture with filter feeders, like scallops, which enhance the pelagic-benthic coupling, thus removing particles, including phytoplankton and wasted food from the water column (Zhou et al. 2006a, Zhou et al. 2006c). Other authors have suggested to move the aquaculture facilities to more exposed and open localities (Feng et al. 2004, Liu et al. 2004).

6.3. Pelagic nutrient loading and impact

There are very few studies published for the pelagic effects of cage aquaculture on the pelagic environment, and there is, as discussed above, no established scientific concept for understanding, monitoring, and managing such potential impacts. The literature search on ISI Web of Science revealed for example 258 papers for key words "aquaculture AND ammonia", but only one for "salmon aquaculture AND ammonia". "Aquaculture AND nutrients" gave 174 papers, but few were relevant for the impact on the pelagic ecosystem. "Aquaculture AND chlorophyll" gave 99 mostly irrelevant paper. The limited efforts made is most likely a result of the problems to detect clear environmental signals of wastes from fish farms in the water masses and the fact that there is no general applicable scientific concept established for assessing and judging impacts of nutrients released from fish farms in water column ecosystems. Appendix Table 2 presents a brief review of the seven main published studies on the pelagic impacts of aquaculture.

Nordvarg and Johansson (2002) provide a comprehensive list of references for earlier papers on pelagic effects of fish farming. The volumetric loading rate, as defined here (i.e., Vollenweider 1980), is hardly covered in any paper, but some studies report the concentrations of nutrient components, particularly ammonia (NH_4) and phosphate (PO_4), around and downstream of cages farms, including some few salmonid farms. Merceron et al. (2002) studied pelagic waters around a brown trout farm located in the Northern coast of France. The site was well flushed; the water current velocity was 2 - 25 cm sec^{-1} in surface waters. The authors found an excess concentration of NH_4 within and shortly downstream of the site, but the nutrient faded away already close to the farm. The concentrations of PO_4 and chlorophyll *a* remained at background levels all through. Soto and Norambuena (2004) examined the pelagic effects of 29 salmon farms grouped in 9 locations in southern Chile and found no effect on water column variables like ammonia, phosphate, and chlorophyll *a*, but they found significant effects for benthic variables. Nordvarg and Johansson (2002) found a positive pelagic effect of some studied fish farms on total P and periphytic growth in the Baltic Sea. Some other farms had no measurable impacts. Secchi depths and oxygen saturation was not affected in any farm. Maldonado et al (2005) studied a number of chemical and biological variables in five Mediterranean fish farms exploiting semi-offshore conditions but found no substantial differences between farm and control sites.

Table 15. Hydrodynamic characteristics and mean excess nitrogen concentrations of the 3 virtual salmon farms studied. Values express the concentration of excess N in the water occupied by the fish cages (hot-spot, one model grid of 160 x 320 m^2). The situations are representative for farms producing 1000 tonnes per year, which is well below today's production (close to 10,000 tonnes per year), but the results nevertheless demonstrates the options of the method. PON concentration in undisturbed coastal waters is set to 60 mg N l^{-1}, which is representative for the region (from Olsen et al 2005, model data are provided by D. Slagstad, SINTEF).

Fish farm number and location	Location conditions	Excess N in farm hotspot, mg N m^{-3} (mmol N m^{-3})	% Excess N of natural PON in farm hotspot, %
1 - Langøya, outer exposed area	Strongly exposed, water is efficiently mixed with the open ocean	0.6 (0.04)	<1
2 - Langøysundet, a straight between islands	Tidal driven water exchange, efficiently mixed	6.4 (0.46)	11
3 - Eidsfjorden, a fjord bottom	Unidirectional, steady water currents, relatively stagnant	17.9 (1.3)	30

An experimental approach has recently been undertaken to measure nutrient availability and incorporation into primary producers by the use of bioassays where growth of phytoplankton and macroalgae is monitored. Dalsgaard and Krause-Jensen (2006) detected pelagic impacts of 4 fish farms across a transect from Eastern to Western Mediterranean. They found, despite of no measurable change in nutrient concentrations in the water column, positive growth responses above the background level within a distance <150 m downstream of the cages, but no effects at greater distances. An additional experiment, where grazers were included in the phytoplankton bioassays, showed no change in chlorophyll a concentrations, suggesting a rapid transfer of nutrients to higher trophic levels in the oligotrophic ocean (Karakassis pers. comm.).

A modelling study examining loading and spreading of inorganic nitrogen from 3 hypothetical fish farms in Norway can serve to illustrate how a 3D simulation hydrodynamic modelling can be used to assess concentrations and distribution of any component, including released inorganic nitrogen and phosphorous, from fish farms (Olsen et al 2005). The virtual farms of the study have been located to Northern Norway (Figure 13), and are placed at sites characterized by variable hydrodynamic conditions (Table 15).

The excess N concentration (N released from the fish farm, whether it is dissolved or taken up by organisms) for a 30 days simulation period showed regular oscillations with tides, but the values remained stable for all sites through the illustrated simulation period (Figure 14). The mean excess N concentration at the farm hot-spot (the model grid where the farm were placed) in the outer exposed site at Langøya showed very low N values, hardly measurable using analytical techniques (Table 15). Nitrogen was immediately dispersed, and neither enhanced primary production nor enhanced phytoplankton biomass (not shown) could be traced downstream of the farm in detectable amounts.

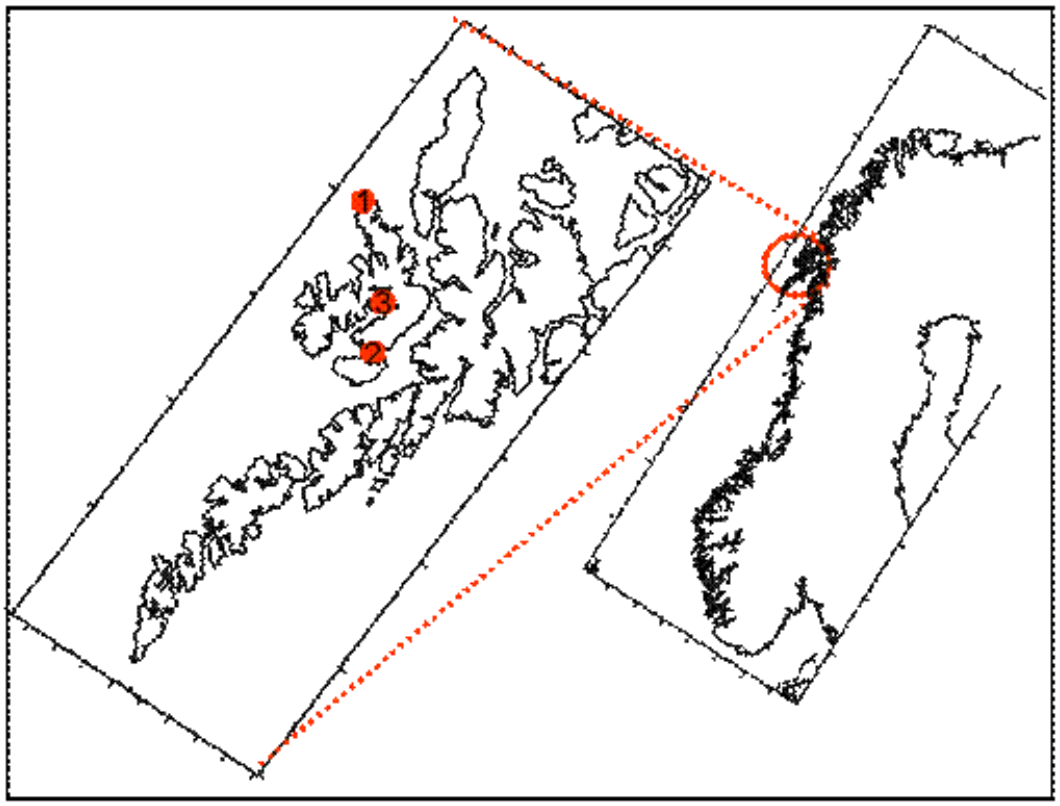

Figure 13. Geographical position of the virtual salmon farms studied (from Olsen et al 2005).

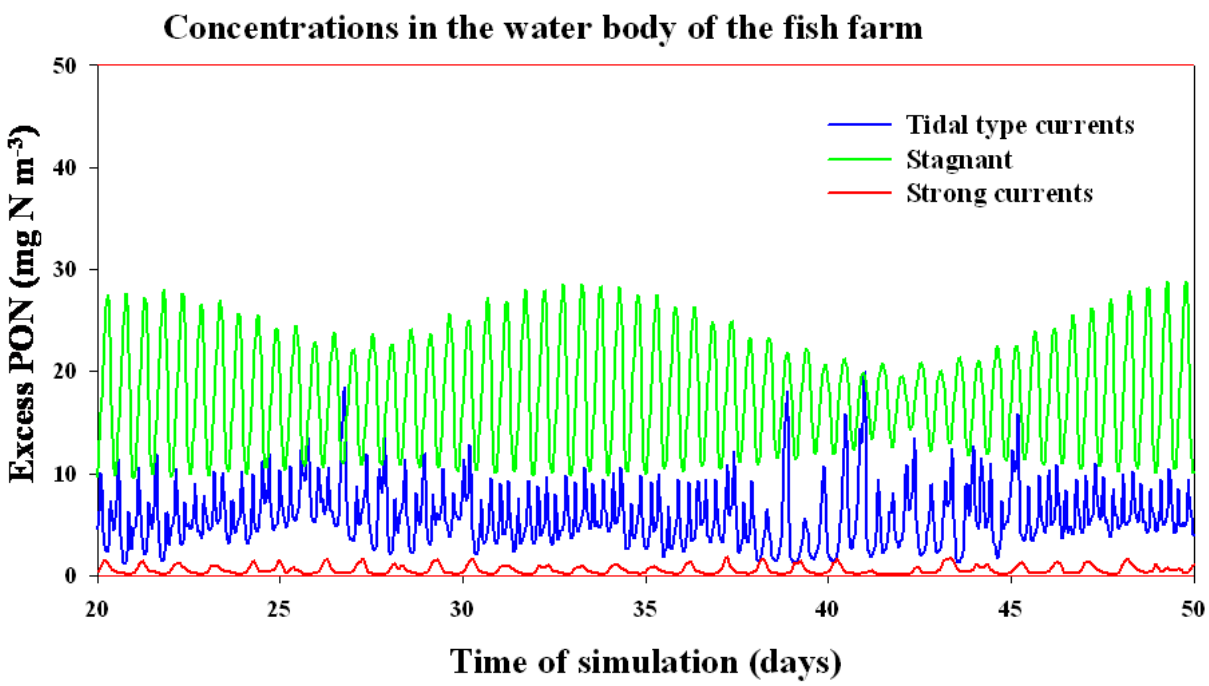

Figure 14. Simulated concentrations of excess N in the three farm hot-spot water masses (the model grid 160 x 320 m^2) illustrated for one month of the summer steady-state situation (from Olsen et al 2005, model data provided by D. Slagstad, SINTEF).

The excess N concentration inside the farm situated in Langøysundet, a straight between islands, was higher (Table 13), corresponding to 11% of the natural PON concentration at the site. Tides moved the water back and forth in the straight, and the surrounding water masses on both sides were to some extent affected locally (<1 km). The hydrodynamic forces were still strong, and the nutrients were widely spread. The residence time of the water was too short to allow a significant enhanced primary production and phytoplankton biomass (not shown) around the fish farm.

The third site, Eidsfjorden, which is a relatively stagnant fjord bottom, was characterised by regular anti-clockwise water currents with a main pattern affected by the tidal cycle (Figure 14). The concentration oscillated quite pronouncedly, meaning that water current velocity varied with tides. Water entered the fjord bottom along the south coast and left along the north. The farm affected the water masses downstream of the site, and the mean hot-spot concentration of excess N was around 30% of the natural PON concentration (Table 15).

Areas characterised by sustained enhanced primary production and phytoplankton biomass were identified downstream the fish farm, in relatively stagnant water bodies along the southern coast of Eidsfjorden (Figure 15). Excess biomass accumulated further downstream than the area of sustained enhanced excess primary production, in agreement with the general considerations of time lags between nutrient pulsing, increasing primary production and increasing phytoplankton biomass. The excess biomass concentration of the affected area was generally <25% of the natural biomass level of phytoplankton.

Excess N, primary production, and biomass did not significantly affect the outer water masses of the fjord (Figure 15). Hydrodynamics were surprisingly strong downstream of the farm along the north coast of the fjord, and the resources were rapidly spread to large water masses.

The virtual fish farm site in Eidsfjorden will probably not affect the water masses in an unacceptable way, but there is probably not space for a much larger salmon farm on this site, a farm producing 4-5000 metric tonnes would probably be too large. The site could, on the other hand, be suitable for integrated aquaculture of mussels, because excess phytoplankton biomass tended to accumulate in downstream locations (Figure 15). These types of locations are not often used for salmon production in Norway any more. The farms have grown much bigger and have tended to move out to more open, although still protected, coastal waters.

6.4. Benthic nutrient loading impact

A search in the ISI databases on “aquaculture AND benthic AND impact” returned 80 references for the years 1990 to 2006 (Appendix Table 3). Accordingly, benthic impact is a much more intensively studied issues than pelagic impact. In a recent synthesis of literature, Mente et al (2006) listed 36 papers published between 1984 and 2003, describing effects on the benthos under salmon cage farms in Europe. All papers report some effects from waste deposition directly under the cages and close to the cage margins. In most cases, the effects seemed to be confined to the immediate vicinity of the fish farm.

Kalantzi and Karakassis (2006) published a meta-analysis of 41 papers on benthic effects of fish farming published between 1988 and 2003, covering mostly salmonid farms, but also bream and bass farms in different regions of the world. They concluded from their regressions that a combination of distance from the farm and bottom depth and/or latitude explained most of the biological and geochemical variation in their material. They also found that the size of the area affected by wastes from farms varied substantially between different sediment types, but the overall conclusion was similar to Mente et al. (2006), with a general local impact found on the benthic compartment.

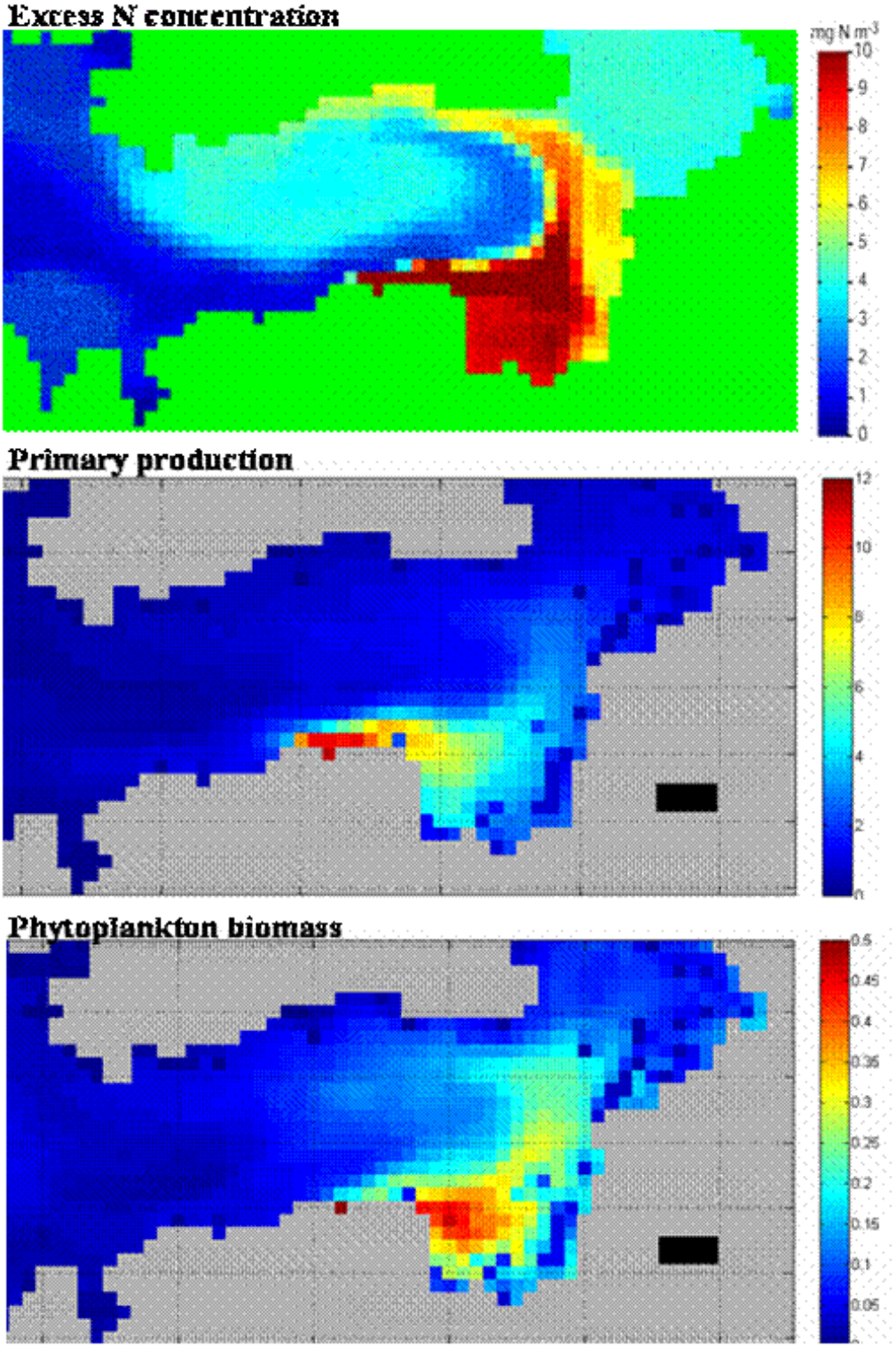

Figure 15. Modelling results for the relatively stagnant virtual salmon farm located in Eidsfjorden. Upper panel: Excess N concentration (mg N m^{-3}); Middle panel: Accumulated excess primary production during 40 days of modelling (gC m^{-2} 40 $days^{-1}$); Lower panel: Mean excess phytoplankton biomass during the 40 days period (mg Chl *a* m^{-2}). Model data are provided by D. Slagstad, SINTEF.

Hyland et al. (2005) found the pools of organic matter (TOC) to be a reliable measure of benthic impacts, whereas others have shown that this indicator is unable to predict the recovery process (Pereira et al. 2004). A large seasonal variation in TOC related to the productivity in the farm also weakens the indicator. Brooks and Mahnken (2003) instead used the sedimentation of organic matter as indicator for benthic impacts at Canadian fish farms, and found quite strong correlations. Similarly, sedimentation rate was used as an indicator of benthic changes in Mediterranean fish farms surrounded by seagrass meadows (Frederiksen et al. 2007). Brooks and Mahnken (2003) found that sedimentation drives changes in sediment biogeochemical conditions, which was also found by Holmer et al (2003a) at fish farms in the Philippines. However, this will occur only up to a certain threshold, where after the sediment metabolism declined, probably due to exhaustion of the sulphate pools in the sediments. Similarly, a lower threshold value with no impact on the benthic environment has been found, indicating that the benthic impacts are confined to a certain range of loading rates (Holmer et al. 2005).

Sedimentation is generally considered as an important driver of changes in benthic communities, e.g. affecting fauna biomass and microbial activity (Jørgensen and Richardson 1996), and has the potential as a strong indicator of the benthic impacts at CAS sites. This concept is used in several of the established environmental quality standards and monitoring programs. Although sedimentation is not measured directly, both the MOM concept and the DEPOMOD consider the sedimentation through the division into different zones of impact, and DEPOMOD includes a direct modelling of the affected areas based on feed inputs, hydrodynamic regimes and farm characteristics (Cromey and Black 2002).

There are many other recent examples of biogeochemical measures in the literature in addition to the major reviews mentioned above. For instance, Yokoyama et al. (2006) established a relationship between N concentration and dissolved oxygen and acid volatile sulphide concentration in the sediment below fish farm cages. According to their regressions, an N-concentration >0.3% (3 mg N/g sediment) may induce serious anoxia with high sulphide concentration in Japanese coastal waters.

McGhie et al. (2000) challenge this relationship in their study of the degradation of wastes under trout cages at two different sites in a Tasmanian estuary. In their reference sites, both unaffected by fish farms, one site with coarse sand showed an N-concentration of 0.05 %. At the other reference site, with fine mud, the N-concentration was 0.49 % and accordingly higher than the limit for anoxia suggested by Yokoyama et al. (2006) for Japanese coastal waters. However, the surface sediment was still oxic at both Tasmanian reference sites (McGhie et al. 2000). Under the fish cages, the maximal N-concentrations were 0.65 and 1.42 % for the sand and mud sites, respectively. Anoxic conditions were found at both fish farm sites (McGhie et al. 2000).

McGhie et al. (2000) used fatty acids as a tracer for fish food wastes. After 12 months of fallowing fish food (i.e. temporary abandonment of the site), fatty acid nitrogen was reduced from 125 to 50 µg N per g sediment directly under the cages at the sand bottom site, while a reduction from 2300 to 100 µg/g sediment was observed at the muddy site (McGhie et al. 2000). This clearly reflects a difference in the degradation rate of wastes, and thus a different capacity for waste assimilation in different sediment types. It took approximately 12 months for the sediment to return to normal oxic condition at both Tasmanian sites, while at this time there was still remnants of aquaculture derived organic matter at both sites (4-40 % of aquaculture-derived fatty acids). Other authors have reported much faster remediation of fish farm sediments during fallowing, in the order of months (e.g. Brooks et al. 2002).

Because of the great variation in the metabolic activity and robustness of different sediment types towards organic loading, it is important to understand the baseline conditions at the aquaculture site in order to be able to predict impact and recovery (Macleod et al. 2004, 2006). Despite the large number of studies on this issue, there is still important knowledge missing such as the resistance and resilience of the sediments towards organic enrichments, and it is not possible to provide confident predictions of many important benthic responses, e.g. the precise determination of the accumulation rate that causes anoxia in the sediments. There is a need for much better understanding the relationships between organic accumulation, sediment geochemical response, consequences for the faunal community, and the role of bioturbation and bioirrigation in carbon degradation by microbial processes. This requires a combined experimental, observational and modelling approach, with a focus on sediment biogeochemistry. Ideally, such understanding would lead to simple chemical proxies (indicators) of sediment state from which faunal community state could be inferred.

6.5. Benthic-pelagic coupling

Pelagic and benthic processes are clearly coupled in shallow waters, and eutrophication of coastal waters affect loading and metabolism of sediments, which in turn can affect pelagic processes through enhanced release and up-welling of nutrients (e.g., Figure 7). The results of fundamental eutrophication studies of coastal planktonic food webs (e.g., Olsen et al. 2006, 2007) indicate that functional groups of the planktonic food web respond in a predictable way to increased nutrient supply (Figure 6 and 7). The carbon flows between heterotrophic compartments are more strongly affected than the biomass of the compartments. An important message from these studies is that increased nutrient input results in increased sedimentation rate of organic and nutrient wastes to deep water and sediments. This means that harmful eutrophication in the water masses will affect the seafloor ecosystem when the zooplankton can no longer effectively remove the increase in primary production and sedimentation accelerates nonlinearly (Olsen et al. 2007). This effect is not restricted to the area around the production site.

The increased nitrogen loading rate due to runoff from human activities during the last 50 years in Chesapeake Bay has been accompanied by an increase in hypoxia and anoxia in sediments and bottom waters. This is but one indication that sedimentation from the pelagic zone to deepwater and seafloor has increased. There is also evidence for that the benthic macro-infauna has degraded, or even vanished, due to low oxygen concentration (Kemp et al. 2005). In the southern Baltic Sea, and increasing eutrophication resulted in reduced penetration depth of oxygen in the sediments, a domination of sulphate respiration over oxygen respiration, and a domination of photolytic enzymes over carbohydrate decomposing enzymes (Meyer-Reil and Köster 2000).

Development of anoxia in the sediments will stimulate the release of ammonia (NH_4) and phosphate (PO_4) to the bottom waters. Such secondary effects of increased sedimentation of organic wastes have, according to Ma et al. (2006), induced blooms of dinoflagellates when the oxic-anoxic interface moved toward the surface in shallow lagoons in Delaware Bay. Carstensen et al. (2007) concluded that the bloom frequency of phytoplankton was directly linked to the total nitrogen concentration in shallow Danish estuaries. They identified four mechanisms linked to nutrient enrichment as sources of summer blooms: 1) advection from biomass rich inner estuary, 2) resuspension of nutrients and algae from sediments, 3) nutrient release from sediments during hypoxic conditions, and 4) decoupling of benthic grazers.

An additional aspect to consider is the large changes in phytoplankton nutrient limitation over the growth season. Nutrients are limiting for most of the summer situation, and benthic-

pelagic coupling is most intense during this period due to higher temperatures and higher production in the farms, increasing the risk of summer blooms (Holmer and Kristensen 1996).

6.6. Importance of future trends in changing technology - concluding remarks

Environmental concern and competition for space in the coastal zone are believed to be main drivers of the development of aquaculture technology in the western countries, which develop along two main lines:

- Sea cage systems (CAS) – floating structures suspending nets, maintained in open waters, used primarily for fish and long line structures for mussels and seaweed.
- Land- or seashore based systems with water flow through or water recycling (RAS; Recycling Aquaculture Systems) - allows control with waste emissions, mainly used for high valued fish, younger stages.

CAS will in the nearest future remain the economically most important system for aquaculture production with respect to volume. This also implies that the majority of wastes from aquaculture will still originate from CAS. Integration of production to include organisms on other trophic level using wastes or biological/chemical waste water treatments are the obvious options to better control the waste emission from RAS and other land based aquaculture systems. The moving out to more open or even offshore location with CAS will to great extent affect the environmental challenges of aquaculture.

Hydrodynamic energy and the planktonic capacity of assimilating nutrients without negative ecological effects (e.g., increased sedimentation) determine the nutrient assimilation capacity of open and offshore locations, like for inshore coastal aquaculture locations, although the benthic-pelagic coupling part plays a more important role in shallow waters. Water current velocities, wind velocity, tides, and depth determine the hydrodynamic regime. Open ocean and offshore sites do not always exhibit stronger hydrodynamic forces than protected coastal sites, but there is a general relationship between degree of openness and hydrodynamic energy. As clearly documented by 3D hydrodynamic modelling (e.g., Figure 14, Table 15), hydrodynamics of an open location suitable for aquaculture, probably characterised by intermediate hydrodynamic forces, will very efficiently spread and dilute nutrient wastes from open sea/offshore fish farms to concentrations below the detectable level. The increased biomass of phytoplankton will be spread as well, and will be found downstream of the farm (3-7 days of transport, see above). The diluted and spread nutrients will enter the planktonic and nektonic food web and stimulate natural fisheries, in accordance with the state of knowledge in biological oceanography and fisheries biology.

The benthic impact of open sites characterised by high hydrodynamic energy and depth will also become less extensive, but there will most likely always be detectable environmental impacts on the benthic ecosystem, impacts that can be predicted by current models. As for the pelagic impacts, the sedimentation will be spread over larger areas and decrease to below detection limit. The incorporation of waste products into biomass of benthic animals will be spread around and down-stream of the farms.

All human activities creates wastes, it is at the end a question about what society will view as an acceptable environmental impact of industrial activity.

7. Recommendations for further R&D

Upcoming environmental legislation will generally require that industry will have to document that their activity is not harming nature in an unacceptable way. The knowledge available to support such considerations is partly there, but there are gaps of knowledge that needs to be filled. Our recommended R&D needed to achieve the knowledge needed to meet the requirements of future environmental legislation is as follows.

1. Obtain a more complete scientific understanding of how marine benthic and pelagic ecosystems assimilate nutrient wastes from aquaculture, in particular for pelagic ecosystems, the scientific understanding of these processes are inadequate. This challenge involves to:
   1.1. Verify and further elaborate on existing nutrient dose – ecological response functions which quantify how nutrients affects structure and function of planktonic and benthic ecosystems, and particularly provide more quantitative documentation for a critical nutrient loading rate and its value.
   1.2. Increase the understanding of how hydrodynamic energy mitigates negative effects of nutrient loading in benthic and pelagic ecosystems and how these hydrodynamic mixing and dilution processes can be adequately modelled.
2. For pelagic ecosystems, quantify coefficients, validity and robustness of the proposed simplified management scheme for nutrient management (Figure 10), using combined experimental and modelling exercises.
3. Evaluate by modelling combined with satellite monitoring the assessment methods for local *versus* regional situations, for regions with variable farm densities and hydrodynamic conditions
4. For benthic ecosystem, further develop and support implementation of cost-effective tools for monitoring and management, using existing knowledge on impacts combined with impact modelling.
5. Explore the potential added value of the production and the positive environmental aspects of using wastes from CAS and RAS as a resource for growing benthic and pelagic organisms on other trophic levels (IMTA, Integrated multi-trophic aquaculture).

The perspective of the R&D should all through comply with the concept of ecosystem based management of nutrient wastes from aquaculture, which is a sound approach for future management of aquaculture in coastal waters.

8. References

Ackefors H, Enell M (1994) The release of nutrients and organic matter from aquaculture systems in Nordic countries. Journal of Applied Ichthyology 10 (4): 225–241.

Ackefors H, Enell M (1990) Discharge of nutrients from Swedish fish farming to adjacent sea areas. Ambio 19 (1): 28-35.

Adron JW, Blair A, Cowey CB, Shanks AM (1976) Effects of dietary energy level and dietary energy source on growth, feed conversion and body composition of turbot (*Scophthalmus maximus* L.) Aquaculture 7:125-132.

Marin A, Montoya S, Aguado-Gimenez F, Piedecausa A, Garcia-Garcia B(2007) Comparison between some procedures for monitoring offshore cage culture in western Mediterranean Sea: Sampling methods and impact indicators in soft substrata Aquaculture 271:357-370

Allan GL, Rowland SJ, Mifsud C, Glendenning D, Stone DAJ, Ford A (2000) Replacement of fish meal in diets for Australian silver perch, *Bidyanus bidyanus* V. Least-cost formulation of practical diets. Aquaculture 186: 327-340.

Aller RC, Aller JY (1998) The effect of biogenic irrigation intensity and solute exchange on diagenetic reaction rates in marine sediments. Journal of Marine Research 56 (4): 905-936.

Alongi DM, Chong VC, Dixon P, Sasekumar A, Tirendi F (2003) The influence of fish cage aquaculture on pelagic carbon flow and water chemistry in tidally dominated mangrove estuaries of peninsular Malaysia. Marine Environmental Research 55:313-333.

Alonso-Rodriguez R, Paez-Osuna F (2003) Nutrients, phytoplankton and harmful algal blooms in shrimp ponds: a review with special reference to the situation in the Gulf of California. Aquaculture 219: 317-336.

Anderson DM, Glibert PM, Burkholder JM (2002) Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences. Estuaries 25:704-726.

Anderson JS, Lall SP, Anderson DM, McNiven MA (1995) Availability of amino acids from various fish meals fed to Atlantic salmon (*Salmo salar*). Aquaculture 138: 291-301.

Aure J, Stigebrandt A (1990) Quantitative estimates of the eutrophication effects of fish farming on fjords. Aquaculture 90: 135-156.

Azanza RV, Fukuyo Y, Yap LG, Takayama H (2005) *Prorocentrum minimum* bloom and its possible link to a massive fish kill in Bolinao, Pangasinan, Northern Phillippines. Harmful Algae 4: 519-524.

Banta GT, Holmer M, Jensen MH, Kristensen E (1999). The effects of two polychaete worms, *Nereis diversicolor* and *Arenicola marina*, on aerobic and anaerobic decomposition in an organic-poor and an organic-enriched marine sediment. *Aquatic Microbial Ecology*19:189-204.

Bartoli M, Nizzoli D, Naldi M, Vezzulli L, Porrello S, Lenzi M, Viaroli P (2005) Inorganic nitrogen control in wastewater treatment ponds from a fish farm (Orbetello, Italy): Denitrification versus Ulva uptake. Marine Pollution Bulletin 50:1386-1397.

Belias CV, Bikas VG, Dassenakis MJ, Scoullos MJ (2003) Environmental impacts of coastal Aquaculture in eastern Mediterranean bays the case of Astakos Gulf, Greece. Environmental Science and Pollution Research 10:287-295.

Brett MM (2003) Food poisoning associated with biotoxins in fish and shellfish. Current Opinion in Infectious Diseases 16:461-465.

Brooks KM, Stierns AR, Mahnken CVW, Blackburn DB (2002) Chemical and biological remediation of the benthos near Atlantic salmon farms. Aquaculture 219: 355-377.

Brooks KM, Aguado-Gimenez F (2003) Interactions of Atlantic salmon in the Pacific northwest environment II. Organic wastes. Fisheries Research 62:255-293.

Brune DE, Schwartz G, Eversole AG, Collier JA, Schweder TE (2003) Intensification of pond aquaculture and high rate photosynthetic systems. Aquacultural Engineering 28: 65-86.

Bureau DP, Cho CY (1999) Phosphorus utilization by rainbow trout (*Oncorhynchus mykiss*): estimation of dissolved phosphorus waste output. Aquaculture 179: 127-140.

Bureau DP, Gunther SJ, Cho CY (2003) Chemical composition and preliminary theoretical estimates of waste outputs of rainbow trout reared in commercial cage culture operations in Ontario. North American Journal of Aquaculture 65 (1): 33-38.

Burford MA, Longmore AR (2001) High ammonium production from sediments in hypereutrophic shrimp ponds. Marine Ecology-Progress Series 224:187-195.

Buschmann AH, Riquelme VA, Hernandez-Gonzalez MC, Varela D, Jimenez JE, Henriquez LA, Vergara PA, Guinez R, Filun L (2006) A review of the impacts of salmonid farming on marine coastal ecosystems in the southeast Pacific. Ices Journal of Marine Science 63:1338-1345.

Butz I, Vens-Cappell B (1982) In: Alabaster, J.S. (Ed.), Report of the FIFAC Workshop on Fish-Farm Effluents. Denmark, May 1981. FIFAC Technical Paper 41, 113–121.

Carmona R, Kraemer GP, Yarish C (2006) Exploring Northeast American and Asian species of Porphyra for use in an integrated finfish-algal aquaculture system. Aquaculture 252:54-65.

Carroll ML, Cochrane S, Fieler R, Velvin R, White P (2003) Organic enrichment of sediments from salmon farming in Norway: environmental factors, management practices, and monitoring techniques. Aquaculture 226:165-180.

Carstensen J, Henriksen P, Heiskanen AS (2007) Summer algal blooms in shallow estuaries: Definition, mechanisms, and link to eutrophication. Limnol. Oceanogr. 52: 370-384.

Castro RS, Azevedo CMSB, Bezerra-Neto F (2006) Increasing cherry tomato yield using fish effluent as irrigation water in Northeast Brazil. Scientia Horticulturae 110:44-50.

Chamberlain J, Fernandes TF, Read P, Nickell TD, Davies IM (2001) Impacts of biodeposits from suspended mussel (Mytilus edulis L.) culture on the surrounding surficial sediments. Ices Journal of Marine Science 58:411-416.

Chambers MD, Howell WH (2006) Preliminary information on cod and haddock production in submerged cages off the coast of New Hampshire, USA. ICES Journal of Marine Science 63: 385-392.

Chen YS, Beveridge MCM, Telfer TC (1999) Physical characteristics of commercial pelleted Atlantic salmon feeds and consideration of implications for modelling of waste dispersion through sedimentation. Aquaculture International 7: 89-100.

Chen YS, Beveridge MCM, Telfer TC, Roy WJ (2003) Nutrient leaching and settling rate characteristics of the faeces of Atlantic salmon (*Salmo salar* L.) and the implications for modelling of solid waste dispersion. J. Appl. Ichtyol. 19: 114-117.

Cheshuk BW, Purser GJ, Quintana R (2003) Integrated open-water mussel (*Mytulis planulatus*) and Atlantic salmon (*Salmo salar*) culture in Tasmania, Australia. Aquaculture 218: 357-378.

Cho CY, Bureau DP (2001) A review of diet formulation strategies and feeding systems to reduce excretory and feed wastes in aquaculture. Aquaculture Research 32: 349-360.

Christensen PB, Rysgaard S, Sloth NP, Dalsgaard T, Schwaerter S (2000) Sediment mineralization, nutrient fluxes, denitrification and dissimilatory nitrate reduction to ammonium in an estuarine fjord with sea cage trout farms. Aquatic Microbial Ecology 21:73-84.

Clapham WB (1983) Natural ecosystems. 2nd ed. Macmillan Publishing Co Inc. New York.

Cloern JE (2001) Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology-Progress Series 210:223-253.

Corner RA, Brooker AJ, Telfer TC, Ross LG (2006) A fully integrated GIS-based model of particulate waste distribution from marine fish-cage sites. Aquaculture 258: 299-311.

Crawford C (2003) Environmental management of marine aquaculture in Tasmania, Australia. Aquaculture 226:129-138.

Crawford CM, Macleod CKA, Mitchell IM (2003)a Effects of shellfish farming on the benthic environment. Aquaculture 224:117-140.

Cromey CJ, Nickell TD, Black KD (2002a) DEPOMOD – modelling the deposition and biological effects of waste solids from marine cage farms. Aquaculture 214: 211-239.

Cromey CJ, Nickell TD, Black KD, Provost PG, Griffiths CR (2002)b Validation of a fish farm waste resuspension model by use of a particulate tracer discharged from a point source in a coastal environment. Estuaries 25(5): 916-929.

Cromey CJ, Black KD (2005) Modelling impacts of finfish aquaculture. In Hargrave BT (ed) The Handbook of Environmental Chemistry Volume 5: Water Pollution Environmental effects of marine finfish aquaculture. Springer Verlag. Berlin Heidelberg New York, p. 129-156.

Da Costa KG, Nalesso RC (2006) Effects of mussel fanning on macrobenthic community structure in Southeastern Brazil. Aquaculture 258:655-663.

Dalsgaard T, Krause-Jensen D (2006) Monitoring nutrient release from fish farms with macroalgal and phytoplankton bioassays. Aquaculture 256: 302-310.

Danovaro R, Gambi C, Luna GM, Mirto S (2004) Sustainable impact of mussel farming in the Adriatic Sea (Mediterranean Sea): evidence from biochemical, microbial and meiofaunal indicators. Marine Pollution Bulletin 49:325-333.

Das B, Khan YSA, Das P (2004) Environmental impact of aquaculture-sedimentation and nutrient loadings from shrimp culture of the southeast coastal region of the Bay of Bengal. Journal of Environmental Sciences-China 16:466-470.

Davies IM (2000) Waste production by farmed Atlantic salmon (*Salmo Salar*) in Scotland. ICES, CM 2000/0:01.

Davies IM, Slaski RJ (2003) Waste production by farmed Atlantic halibut (*Hippoglossus hippoglossus* L.). Aquaculture 219: 495-502.

Dempster T, Fernandez-Jover D, Sanchez-Jerez P, Tuya F, Bayle-Sempere J, Boyra A, Haroun RJ (2005) Vertical variability of wild fish assemblages around sea-cage fish farms: implications for management. Marine Ecology-Progress Series 304:15-29.

Dong QX, Lin XP, He X, Kelso WE, Huang CJ (2006) Taxonomic composition, abundance and biomass of mesozooplankton in the Zhelin Bay - an estuary with intensive aquaculture. Acta Oceanologica Sinica 25:92-111.

Ebeling JM, Welsh CF, Rishel KL (2006) Performance evaluation of an inclined belt filter using coagulation/flocculation aids for the removal of suspended solids and phosphorus from microscreen backwash effluent. Aquacultural Engineering 35:61-77.

Eden N, Katz T, Angel DL (2003) Dynamic response of a mud snail Nassarius sinusigerus to changes in sediment biogeochemistry. Marine Ecology-Progress Series 263:139-147.

Edwards M, Johns DG, Leterme SC, Svendsen E, Richardson AJ (2006) Regional climate change and harmful algal blooms in the northeast Atlantic. Limnol. Oceanogr. 51: 820-829.

Elser JJ, Sterner RW, Gorokhova E, Fagan WF, Markow TA, Cotner JB, Harrison JF, Hobbie SE, Odell GM, Weider LJ (2000) Biological stoichiometry from genes to ecosystems. Ecol. Lett. 3: 540-550.

Enell M (1995) Environmental impact of nutrients from nordic fish farming. Water Science and Technology 31 (10): 61-71.

Ervik A, Hansen PK, Aure J, Stigebrandt A, Johannesen P, Jahnsen T (1997) Regulating the local environmental impact of intensive marine fish farming. I. The concept of the MOM system (Modelling-Ongrowing fish farms-Monitoring). Aquaculture 158: 85-94.

FAO (2006) The state of world fisheries and aquaculture. FAO Fisheries and Aquaculture Department, Food and Agriculture Organization of the United Nations. Rome,. ISSN 1020-5489.

Fei XG (2004) Solving the coastal eutrophication problem by large scale seaweed cultivation. Hydrobiologia 512:145-151.

Felsing B, Glencross B, Telfer T (2005) Preliminary study on the effects of exclusion of wild fauna from aquaculture cages in a shallow marine environment. Aquaculture 243:159-174.

Fenchel T (1988) Marine plankton food chains. Ann. Rev. Ecol. Syst. 19: 19-38.

Feng YY, Hou LC, Ping NX, Ling TD, Kyo CI (2004) Development of mariculture and its impacts in Chinese coastal waters. Reviews in Fish Biology and Fisheries 14:1-10.

Fernandes M, Leuer P, Cheshire A, Angove M (2007) Preliminary model of nitrogen loads from southern bluefin tuna aquaculture. Marine Pollution Bulletin 54: 1321-1332.

Fernandez F, Miquel AG, Guinea J, Martinez R (1998) Digestion and digestibility in gilthead sea bream (*Sparus aurata*): the effect of diet composition and ration size. Aquaculture 166: 67-84.Fountoulaki E, Alexis MN, Nengas I, Venou B (2005) Effect of diet composition on nutrient digestibility and digestive enzyme levels of gilthead sea bream (Sparus aurata L.). Aquaculture Research 36: 1243-1251.

Frederiksen MS, Holmer M, Diaz-Almela E, Marbà N, Duarte CM (2007). Sulfide invasion in the seagrass Posidonia oceanica at Mediterranean fish farms: assessment by stable sulphur isotopes. Mar. Ecol. Prog. Ser. 345:93-104.

Gao QF, Cheung KL, Cheung SG, Shin PKS (2005) Effects of nutrient enrichment derived from fish farming activities on macroinvertebrate assemblages in a subtropical region of Hong Kong. Marine Pollution Bulletin 51:994-1002.

Gelfand Y, Barak Z, Even-Chen E, Cytryn M, Krom MD, Neori A and J. van Rijn (2003) A novel zero-discharge intensive seawater recirculating system for culture of marine fish, J. World Aquac. Soc. 34, pp. 344–358.

Gennaro P, Guidotti M, Funari E, Porrello S, Lenzi M (2006) Reduction of land based fish-fanning impact by phytotreatment pond system in a marginal lagoon area. Aquaculture 256:246-254.

Gifford S, Dunstan RH, O'Connor W, Roberts T, Toia R (2004) Pearl aquaculture - profitable environmental remediation? Science of the Total Environment 319:27-37.

Gifford S, Dunstan H, O'Connor W, Macfarlane GR (2005) Quantification of in situ nutrient and heavy metal remediation by a small pearl oyster (Pinctada imbricata) farm at Port Stephens, Australia. Marine Pollution Bulletin 50:417-422.

Gillibrandt PA, Gubbins MJ, Greathead C, Davies IM (2002) Scottish executive locational guidelines for fish farming: predicted levels of nutrient enhancement and benthic impact. Scottish Fisheries Research Report 63/2002. Fisheries Research Services, Aberdeen.

Gooley GJ, DeSilva SS, Hone PW, McKinnon LJ, Ingram BA, (2000) Cage aquaculture in Australia: a developed country perspective with reference to integrated aquaculture development within inland waters. In: Liao, I.C., Lin, C.K. (Eds.), Cage Aquaculture in Asia: Proceedings of the First International Symposium on Cage Aquaculture in Asia, pp. 21–37. Asian Fisheries Society, Manila, and World Aquaculture Society—Southeast Asian Chapter, Bangkok.

Goudey CA, Loverich G, Kite-Powell H, Costa-Pierce BA (2001) Mitigating the environmental effects of mariculture through single-point moorings (SPMs) and drifting cages. Ices Journal of Marine Science 58:497-503.

Goyondet T, Koutitonsky VG, Roy S (2005) Effects of water renewal estimates on the oyster aquaculture potential of an inshore area. Journal of Marine Systems 58: 35-51.

Guo LG, Li ZJ (2003) Effects of nitrogen and phosphorus from fish cage-culture on the communities of a shallow lake in middle Yangtze River basin of China. Aquaculture 226: 201-212.

Grall J, Chauvaud L (2002) Marine eutrophication and benthos: the need for new approaches and concepts. Global Change Biology 8:813-830.

Gyllenhammar A, Hakanson L (2005) Environmental consequence analyses of fish farm emissions related to different scales and exemplified by data from the Baltic - a review. Marine Environmental Research 60:211-243.

Hall POJ, Anderson LG, Holby O, Kollberg S, Samuelsson MO (1990) Chemical fluxes and mass balances in a marine fish cage farm. I. Carbon. Mar. Ecol. Prog. Ser. 61: 61-73.

Hall POJ, Anderson LG, Holby O, Kollberg S, Samuelsson MO (1990) Chemical fluxes and mass balances in a marine fish cage farm. IV. Nitrogen. Mar. Ecol. Prog. Ser. 89: 81-91.

Hall SJ (2002) The continental shelf benthic ecosystem: current status, agents for change and future prospects. Environmental Conservation 29:350-374.

Hansen PK, Ervik A, Schaanning M, Johannessen P, Aure J, Jahnsen T, Stigebrandt A (2001) Regulating the local environmental impact of intensive, marine fish farming II. The monitoring programme of the MOM system (Modelling-Ongrowing fish farms-Monitoring). Aquaculture 194:75-92.

Hargrave B (2005) (Ed.). Environmental effects of marine finfish aquaculture. The Handbook of Environmental Chemistry 5M. Springer-Verlag, Berlin, Heidelberg, New York.

Hargrave BT, Holmer M, Newcombe C (2008) Towards a classification of organic enrichment in marine sediments by the use of (bio)geochemical indicators Mar. Pollut. In press.

Hartstein ND (2005) Acoustical and sedimentological characterization of substrates in and around sheltered and open-ocean mussel aquaculture sites and its bearing on the dispersal of mussel debris. Ieee Journal of Oceanic Engineering 30:85-94.

Hemre GI, Bjornevik M, Beattie C, Bjornson BT, Hansen T (2002) Growth and salt-water tolerance of juvenile Atlantic salmon, *Salmo salar*, reared under different combinations of dietary carbohydrate and photoperiod regime. Aquaculture Nutrition 8: 23-32.

Henry-Silva GG, Camargo AFM (2006) Efficiency of aquatic macrophytes to treat Nile tilapia pond effluents. Scientia Agricola 63:433-438.

Herrera-Silvera JA, Comin FA, randa-Cirerol N, Troccoli L, Capurro L (2004) Coastal water quality assessment in the Yucatan Peninsula: management implications. Ocean & Coastal Management 47:625-639.

Hevroy EM, Sandnes K, Hemre GI (2004) Growth, feed utilization, appetite and health in Atlantic salmon (*Salmo salar* L.) fed a new type of high lipid fishmeal, Sea Grain, processed from various pelagic marine fish species. Aquaculture 235: 371-392.

Hillestad M, Johnsen F, Åsgård T (2001) Protein to carbohydrate ratio in high-energy diets for Atlantic salmon (*Salmo salar* L.) Aquaculture Research 32: 517-529.

Holmer M, Kristensen E (1994) Organic matter mineralization in an organic-rich sediment: Experimental stimulation of sulphate reduction by fish food pellets. FEMS Microbiology Ecology 14: 33-44.

Holmer M, Kristensen E (1996). The seasonality of sulfate reduction and major pore water constituents in a marine fish farm sediment: The importance of sedimentary carbon. *Biogeochemistry* 32(1):15-39.

Holmer M, Duarte CM, Heilskov A, Olesen B, Terrados J (2003a) Biogeochemical conditions in sediments enriched by organic matter from net-pen fish farms in the Bolinao area, Philippines. Marine Pollution Bulletin 46: 1470-1479.Holmer M, Pérez M, Duarte CM (2003b). Benthic primary producers – a neglected environmental problem in Mediterranean maricultures? *Mar. Poll. Bull*. 46:1372-1376

Holmer M, Wildish D, Hargrave B (2005). Organic enrichment from marine finfish aquaculture and effects on sediment processes. In B.T. Hargrave (ed) *The Handbook of Environmental Chemistry* Volume 5: Water Pollution: Environmental effects of marine finfish aquaculture. Springer Verlag. 181-206.

Holmer M, Frederiksen MS (2007) Stimulation of sulphate reduction rates in Mediterranean fish farm sediments inhabited by the seagrass *Posidonia oceanica*.

Holmer M, Hansen PK, Karakassis I, Borg JA, Schembri PJ (*in press*). Monitoring of environmental impacts of marine aquaculture. In Aquaculture in the Ecosystem, Springer, Netherlands.

Honkanen T, Helminen H (2000) Impacts of fish farming on eutrophication: Comparisons among different characteristics of ecosystem. International Revue der gesamten Hydrobiologie 85: 673-686

Hua K, Bureau DP (2006) Modelling digestible phosphorus content of salmonid fish feeds. Aquaculture 254: 455-465.

Hubas C, Davoult D, Cariou T, Artigas LF (2006) Factors controlling benthic metabolism during low tide along a granulometric gradient in an intertidal bay (Roscoff Aber Bay, France). Mar Ecol Prog Ser 316: 53-68.

Hyland J, Balthis L, Karakassis I, Magni P, Petrov A, Shine J, Vestergaard O, Warwick R (2005) Organic carbon content of sediments as an indicator of stress in the marine benthos. Marine Ecology Progress Series 295:91–103

Islam MS (2005) Nitrogen and Phosphorus budget in coastal and marine cage aquaculture and impacts of effluent loading on ecosystem: review and analysis towards model development. Marine Pollution Bulletin 50: 48-61.

Jahan P, Watanabe T, Kiron V, Satoh S (2003) Phosphorus and nitrogen excretion during growth span of carp kept under two rearing systems. Fisheries Science 69 (3), 431–437.

Janowicz M, Ross J (2001) Monitoring for benthic impacts in the southwest New Brunswick salmon aquaculture industry. Ices Journal of Marine Science 58:453-459.

Jones AB, Dennison WC, Preston NP (2001) Integrated treatment of shrimp effluent by sedimentation, oyster filtration and macroalgal absorption: a laboratory scale study. Aquaculture 193: 155-178.

Jørgensen BB (1982) Mineralization of organic matter in the sea bed: the role of sulphate reduction. Nature 296: 643-645.

Jørgensen BB, Richardson K. (eds.). Eutrophication in coastal marine ecosystems. Coastal and estuarine studies 52. American Geophysical Union.

Kalantzi L, Karakassis I (2006) Benthic impacts of fish farming: Meta-analysis of community and geochemical data. Marine Pollution Bulletin 52:484-493.

Kamilya D, Sarkar S, Maiti TK, Bandyopadhyay S, Mal BC (2006) Growth and nutrient removal rates of Spirulina platensis and Nostoc muscorum in fish culture effluent: a laboratory-scale study. Aquaculture Research 37:1594-1597.

Karim MR, Sekine M, Ukita M (2003) A model of fish preference and mortality under hypoxic water in the coastal environment. Marine Pollution Bulletin 47:25-29.

Karakassis I, Pitta P, Krom MD (2005) Contribution of fish farming to the nutrient loading of the Mediterranean. Scientia Marina 69 (2): 313-321.

Karakassis I, Tsapakis M, Hatziyanni E, Papadopoulou KN, Plaiti W (2000) Impact of cage farming of fish on the seabed in three Mediterranean coastal areas. Ices Journal of Marine Science 57:1462-1471.

Karakassis I, Tsapakis M, Smith CJ, Rumohr H (2002) Fish farming impacts in the Mediterranean studied through sediment profiling imagery. Marine Ecology Progress Series 227:125–133.

Kemp et al. (2005) Eutrophication of Chesapeake Bay: Historical trends and ecological interactions. Mar. Ecol. Prog. Ser. 303:1-29.

Klaoudatos SD, Klaoudatos DS, Smith J, Bogdanos K, Papageorgiou E (2006) Assessment of site specific benthic impact of floating cage fanning in the eastern Hios island, Eastern Aegean Sea, Greece. Journal of Experimental Marine Biology and Ecology 338:96-111.

Kristensen E (1985) Oxygen and inorganic nitrogen exchange in a *nereis virens* (polychaeta) bioturbated sediment-water system. Journal of Coastal Research 1(2): 109-116.

Krom MD, Brenner S, Kress N, and Gordon LI (1991) Phosphorous limitation of primary productivity in the Eastern Mediterranean Sea. Limnol Oceanogr 36: 424–432.

La Rosa T, Mirto S, Marino A, Alonzo V, Maugeri TL, Mazzola A (2001) Heterotrophic bacteria community and pollution indicators of mussel - farm impact in the Gulf of Gaeta (Tyrrhenian Sea). Marine Environmental Research 52:301-321

Laurent C, Tett P, Fernandes T, Gilpin L, Jones K (2006) A dynamic CSTT model for the effects of added nutrients in Loch Creran, a shallow fjord. Journal of Marine Systems 61:149-164

Lee HW, Bailey-Brock JH, Mcgurr MM (2006) Temporal changes in the polychaete infaunal community surrounding a Hawaiian mariculture operation. Marine Ecology-Progress Series 307:175-185.

Lefebvre S, Bacher C, Meuret A, Hussenot J (2001) Modelling approach of nitrogen and phosphorus exchanges at the sediment-water interface of an intensive fishpond system. Aquaculture 195: 279-297.

Lenzi M, Persia E, Tomassetti P, Finoia MG (2003)b Reduction of aquaculture wastewater eutrophication by phytotreatment ponds system I. Dissolved and particulate nitrogen and phosphorus. Aquaculture 219:515-529.

Leung KMY, Chu JCW, Wu RSS (1999) Nitrogen budgets for the areolated grouper Epinephelus areolatus cultured under laboratory conditions and in open-sea cages. Mar Ecol Prog Ser 186: 271-281.

Liu H, Fang JG, Zhu JX, Dong SL, Wang F, Liang XM, Zhang JH, Lian Y, Wang LC, Jiang WW (2004) Study on limiting nutrients and phytoplankton at long-line-culture areas in Laizhou Bay and Sanggou Bay, northeastern China. Aquatic Conservation-Marine and Freshwater Ecosystems 14:551-574.

Luning K, Pang SJ (2003) Mass cultivation of seaweeds: current aspects and approaches. Journal of Applied Phycology 15:115-119.

Lupatsch I, Kissil GW (1998) Predicting aquaculture waste from gilthead seabream (Sparus aurata) culture using a nutritional approach. Aquatic Living Resources 11: 265-268.

Ma SF, Whereat EB, Luther GW (2006) Shift of algal community structure in dead end lagoons of the Delaware Inland Bays during seasonal anoxia. Aquat. Microb. Ecol. 44: 279-290.

Martens CS, Val Klump J (1984) Biogeochemical cycling in an organic-rich coastal marine basin 4. An organic carbon budget for sediments dominated by sulphate reduction and methanogenesis. Geochimica and Cosmochimica Acta 48: 1987-2004.

McIntosh D, Fitzsimmons K (2003) Characterization of effluent from an inland, low-salinity shrimp farm: what contribution could this water make if used for irrigation. Aquacultural Engineering 27:147-156.

Machias A, Giannoulaki M, Somarakis S, Maravelias CD, Neofitou C, Koutsoubas D, Papadopoulou KN, Karakassia I. (2006). Fish farming effects ond local fisheries landings in oligotrophic seas. Aquaculture 261: 809-816.

Macleod CK, Moltschaniwskyi NA, Crawford CM (2006) Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages. Marine Pollution Bulletin 52: 1458-1466.

Macleod CK, Crawford CM, Moltschaniwskyj NA (2004) Assessment of long term change in sediment condition after organic enrichment: defining recovery. Marine Pollution Bulletin 49:79-88.

Magill SH, Thetmeyer H, Cromey CJ (2006) Settling velocity of faecal pellets of gilthead sea bream (Sparus aurata L.) and sea bass (Dicentrarchus labrax L.) and sensitivity analysis using measured data in a deposition model. Aquaculture 251:295-305.

Maldonado M, Carmona MC, Echeverria Y, Riesgo A (2005) The environmental impact of Mediterranean cage fish farms at semi-exposed locations: does it need a re-assessment? Helgoland Marine Research 59:121-135.

Mallet AL, Carver CE, Landry T (2006) Impact of suspended and off-bottom Eastern oyster culture on the benthic environment in eastern Canada. Aquaculture 255:362-373.

Marbà N, Santiago R, Díaz-Almela E, Àlvarez E, Duarte CM (2006). Seagrass (*Posidonia oceanica*) vertical growth as an early indicator of fish farm-derived stress. Estuarine, Coastal and Shelf Science 67:475-483.

Maroni K (2000) Monitoring and regulation of marine aquaculture in Norway. J. Appl. Ichtyol. 16: 192-195.

McGhie TK, Crawford CM, Mitchell IM, O'Brien D (2000) The degradation of fish-cage waste in sediments during fallowing. Aquaculture 187: 351-366.

McGinnity P, Prodohl P, Ferguson A, Hynes R, Maoileidigh NO, Baker N, Cotter D, O'Hea B, Cooke D, Rogan G, Taggart J, Cross T (2003) Fitness reduction and potential extinction of wild populations of Atlantic salmon, *Salmo salar*, as a result of interactions with escaped farm salmon. Proc. R. Soc. Lond. B. 270: 2443-2450.

Megonigal JP, Hines ME, Visscher PT (2004) Anaerobic Metabolism: Linkages to trace gases and aerobic processes. Pages 317-424 in Schlesinger WH (Ed.) Biogeochemistry. Elsevier-Pergamon, Oxford, UK.

Mente E, Pierce GJ, Santos MB, Neofitou C (2006) Effect of feed and feeding in the culture of salmonids on the marine aquatic environment: a synthesis for European aquaculture. Aquaculture Int. 14: 499-522.

Merceron M, Kempf M, Bentley D, Gaffet JD, Le Grand J, Lamort-Datin L (2002) Environmental impact of a salmonid farm on a well flushed marine site: I. Current and water quality. J. Appl. Ichthyol. 18: 40-50.

Metaxa E, Deviller G, Pagand P, Alliaume C, Casellas C, Blancheton JP (2006) High rate algal pond treatment for water reuse in a marine fish recirculation system: Water purification and fish health. Aquaculture 252:92-101.

Meyer-Reil LA, Köster M (2000) Eutrophication of marine waters: Effects on benthic microbial communities. Marine Pollution Bulletin 41: 255-263.

Miron G, Landry T, Archambault P, Frenette B (2005) Effects of mussel culture husbandry practices on various benthic characteristics. Aquaculture 250:138-154.

Mitra A, Flynn KJ (2006) Promotion of harmful algal blooms by zooplankton predatory activity. Biology Letters 2: 194-197.

Molvær J, Knutzen J, Magnusson J, Rygg B, Skei J, Sørensen J (1997) Classification of environmental quality in fjords and coastal waters: a guide. Norwegian State Pollution Control Authority (SFT) publication 97:03. Oslo, Norway. 36 pp. in Norwegian.

Morrisey DJ, Gibbs MM, Pickmere SE, Cole RG (2000) Predicting impacts and recovery of marine-farm sites in Stewart Island, New Zealand, from the Findlay-Watling model. Aquaculture 185:257-271.

Muir J (2005) Managing to harvest? Perspectives on the potential of aquaculture. Phil. Trans. R. Soc. B 360: 191-218.

Mulsow, S., Krieger, Y. and Kennedy, R. (2006). Sediment profile imaging (SPI) and micro-electrode technologies in impact assessment studies: Example from two fjords in Southern Chile used for fish farming. Journal of Marine Systems 62**:** 152-163.

Murray JW, Grundmanis V (1980) Oxygen-consumption in pelagic marine-sediments. Science 209: 1527-1530.

Neori A, Chopin T, Troell M, Buschmann AH, Kraemer GP, Halling C, Shpigel M, Yarish C (2004) Integrated aquaculture: rationale, evolution and state of the art emphasizing seaweed biofiltration in modem mariculture. Aquaculture 231 (1-4): 361-391.

Newell RIE (2004) Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: A review. Journal of Shellfish Research 23:51-61.

Nizzoli D, Welsh DT, Fano EA, Viaroli P (2006) Impact of clam and mussel farming on benthic metabolism and nitrogen cycling, with emphasis on nitrate reduction pathways. Marine Ecology-Progress Series 315:151-165.

Nordvarg L, Hakanson L (2002)a Predicting the environmental response of fish farming in coastal areas of the Aland archipelago (Baltic Sea) using management models for coastal water planning. Aquaculture 206:217-243.

Nordvarg L, Johansson T (2002)b The effects of fish farm effluents on the water quality in the Aland archipelago, Baltic Sea. Aquacultural Engineering 25:253-279.

Olsen Y (2002) MARICULT Research Programme: background, status and main conclusions. Hydrobiologia 484:1-10.

Olsen, Y., O. Vadstein, D. Slagstad. 2005. Assimilative carrying capacity: Contribution and impacts on the pelagic system. In: Lessons from the past to optimize the future (Eds. Howell B., and Flos, R. Aquaculture Europe 2005, Trondheim, Norway, August 5-9, 2005. EAS Special Publication No 35. Extended abstract.

Olsen Y et al. (2006) A comparative study of responses in planktonic food web structure and function in contrasting European coastal waters exposed to experimental nutrient addition. Limnol. Oceanogr. 51: 488-503.

Olsen Y, Andersen T, Gismervik I, Vadstein O (2007) Protozoan and metazoan zooplankton-mediated carbon flows in nutrient-enriched coastal planktonic communities, Mar. Ecol. Prog. Ser., 331: 67-83.

Alonso-Rodriguez R and Paez-Osuna F (2003) Nutrients, phytoplankton and harmful algal blooms in shrimp ponds: a review with special reference to the situation in the Gulf of California. Aquaculture 219: 317-336.

Papatryphon E, Petit J, Kaushik SJ, van der Werf HMG (2004) Environmental impact assessment of salmonid feeds using Life Cycle Assessment (LCA). Ambio 33:316-323.

Palacios ME, Dabrowski K, Abiado MAG, Lee KJ, Kohler CC (2006) Effect of diets formulated with native Peruvian plants on growth and feeding efficiency of Red Pacu (*Piaractus brachypomus*) juveniles. Journal of the world aquaculture society37: 246-255.

Penczak T, Galicka W, Molinski M, Kusto E, Zalewski M (1982) The enrichment of a mesotrophic lake by carbon, phosphorus and nitrogen from the cage aquaculture of rainbow trout *Salmo gairdneri*. Journal of Applied Ecology 19:371-393.

Pergent-Martini C, Boudouresque CF, Pasqualini V, Pergent G (2006). Impact of fish farming facilities on Posidonia oceanica meadows: a review. Marine Ecology-An Evolutionary Perspective 27: 310-319.

Phillips MJ, Clarke R, Mowat A (1993) Phosphorus leaching from Atlantic salmon diets. Aquacultural engineering 12: 47-54.

Pitta P, Apostolaki ET, Tsagaraki T, Tsapakis M, Karakassis I (2006) Fish farming effects on chemical and microbial variables of the water column: A spatio-temporal study along the Mediterranean Sea. Hydrobiologia 563:99-108.

Pohle G, Frost B, Findlay R (2001) Assessment of regional benthic impact of salmon mariculture within the Letang Inlet, Bay of Fundy. Ices Journal of Marine Science 58:417-426.

Porrello S, Lenzi M, Tomassetti P, Persia E, Finoia MG, Mercatali I (2003)a Reduction of aquaculture wastewater eutrophication by phytotreatment ponds system II. Nitrogen and phosphorus content in macroalgae and sediment. Aquaculture 219:531-544.

Porrello S, Lenzi M, Ferrari G, Persia E, Savelli F, Tomassetti P (2005) Loading of nutrient from a land-based fish farm (Orbetello, Italy) at different times. Aquaculture International 13:97-108.

Porter G (2003) Protecting wild Atlantic salmon from impacts of salmon aquaculture. A country by country progress report. WWF. www.worldwildlife.org

Primavera JH (2006) Overcoming the impacts of aquaculture on the coastal zone. Ocean and Coastal Management 49: 531-545.

Qian PY, Wu MCS, Ni IH (2001) Comparison of nutrient release among some maricultured animals. Aquaculture 200: 305-316.

Rafiee G, Saad CR (2005) Nutrient cycle and sludge production during different stages of red tilapia (*Oreochromis* sp.) growth in a recirculating aquaculture system. Aquaculture 244: 109-118.

Revsbech NP, Jørgensen BB, Blackburn TH (1980) Oxygen in the sea bottom measured with a microelectrode. Science 207: 1355-1356.

Rishel KL, Ebeling JM (2006) Screening and evaluation of alum and polymer combinations as coagulation/flocculation aids to treat effluents from intensive aquaculture systems. Journal of the World Aquaculture Society 37:191-199.

Roggenbauer RH (2005) Comparing indoor wastewater-fed aquaculture with two conventional wastewater treatment plants using adopted sustainable process indices. International Journal of Sustainable Development and World Ecology 12:132-140.

Roy PB, Lall SP (2004) Urinary phosphorus excretion in haddock, *Melanogrammus aeglefinus* (L.) and Atlantic salmon, *Salmo salar* (L.). Aquaculture 233: 369-382.

Ruhonen K, Vielma J, Grove DJ (1999) Low-protein supplement increases protein retention and reduces the amounts of nitrogen and phosphorus wasted by rainbow trout fed on low-fat herring. Aquaculture Nutrition 5: 83-91.

Rysgaard S, Christensen PB, Sorensen MV, Funch P, Berg P (2000) Marine meiofauna, carbon and nitrogen mineralization in sandy and soft sediments of Disko Bay, West Greenland. Aquat Microb Ecol 21: 59-71.

Sá R, Pousão-Ferreira and Oliva-Teles A (2007) Growth performance and metabolic utilization of diets with different protein: carbohydrate ratios by white sea bream (*Diplodus sargus*,L.) juveniles. Aquaculture Research 38: 100-105.

Sara G, Scilipoti D, Milazzo M, Modica A (2006) Use of stable isotopes to investigate dispersal of waste from fish farms as a function of hydrodynamics. Marine Ecology-Progress Series 313:261-270.

Sarker SA, Satoh S, Kiron V (2005) Supplementation of citric acid and amino acid-chelated trace element to develop environment-friendly feed for red sea bream, *Pagrus major*. Aquaculture 248: 3-11.

Schneider O, Sereti V, Eding EH, Verreth JAJ, Klapwijk B (2007) Kinetics, design and biomass production of a bacteria reactor treating RAS effluent streams. Aquacultural Engineering 36:24-35.

Schneider O, Sereti V, Machiels MAM, Eding EH, Verreth JAJ (2006) The potential of producing heterotrophic bacteria biomass on aquaculture waste. Water Research 40:2684-2694.

Schuenhoff A, Mata L, Santos R (2006) The tetrasporophyte of *Asparagopsis armata* as a novel seaweed biofilter. Aquaculture 252:3-11.

Shao JP, Tian X, Ying B (2006) Dynamic changes of total bacteria and Vibrio in an integrated seaweed-abalone culture system. Aquaculture 252:289-297.

Shimoda T, Suryati E, Ahmad T (2006) Evaluation in a shrimp aquaculture system using mangroves, oysters, and seaweed as biofilters based on the concentrations of nutrients and chlorophyll a. Jarq-Japan Agricultural Research Quarterly 40:189-193.

Singh RK and Balange A (2005) Effect of restricted feeding regimes on compensatory weight gain and body tissue in fry of the Indian major carp *Cirrhinus mrigala* (Hamilton, 1822). Israeli Journal of Aquaculture 57 (4): 250-254.

Smith AC, Martin JL, Ehrman JM, Kaczmarska I (2001) Ten-year record of *Thalassiosira nordenskioeldii* population dynamics: comparison of aquaculture and non-aquaculture sites in the Quoddy Region. Ices Journal of Marine Science 58:391-397.

Sorokin YI, Sorokin PY, Ravagnan G (2006) Hypereutrophication events in the Ca'Pisani lagoons associated with intensive aquaculture. Hydrobiologia 571:1-15.

Soto D, Norambuena F (2004) Evaluation of salmon farming effects on marine systems in the inner seas of southern Chile: a large-scale mensurative experiment. J. Appl. Ichtyol. 20: 493-501.

Stenton-Dozey J, Probyn T, Busby A (2001) Impact of mussel (Mytilus galloprovincialis) raft-culture on benthic macrofauna, in situ oxygen uptake, and nutrient fluxes in Saldanha Bay, South Africa. Canadian Journal of Fisheries and Aquatic Sciences 58:1021-1031.

Sterner RW, Elser JJ (2002) Ecological stochiometry. Princeton University Press.

Stewart ARJ, Grant J (2002) Disaggregation rates of extruded salmon feed pellets: influence of physical and biological variables. Aquaculture research 33: 799-810.

Sugiura SH, Merchant DD, Kelsey K, Wiggins T, Ferraris RP (2006) Effluent profile of commercially used low-phosphorus fish feeds. Environmental pollution 140: 95-101.

Sumagaysay-Chavoso NS (2003) Nitrogen and phosphorus digestibility and excretion of different-sized groups of milkfish (*Chanos chanos* Forsskal) fed formulated and natural food-based diets. Aquaculture Research 34: 407-418.

Sumagaysay-Chavoso NS, San Diego-McGlone ML, David LT (2004) Environmental capacity of receiving water as basis for regulating intensity of milkfish (*Chanos chanos* Forsskal) culture. Journal of Applied Ichthyology 20:476-487.

Sweilum MA, Abdella MM, El-Din SAS (2005) Effect of dietary protein-energy levels and fish initial sizes on growth rate, development and production of Nile tilapia, *Orechromis niloticus* L. Aquaculture Research 36: 1414-1421.

Tacon AGJ, Forster IP (2003) Aquafeeds and the environment: policy implications. Aquaculture 226:181-189.

Tantikitti C, Sangpong W, Chiavareesajja S (2005) Effects of defatted soybean protein levels on growth performance and nitrogen and phosphorus excretion in Asian seabass (*Lates calcarifer*). Aquaculture 248: 41-50.

Tett P, Gilpin L, Svendsen H, Erlandsson CP, Larsson U, Kratzer S, Fouilland E, Janzen C, Lee JY, Grenz C, Newton A, Ferreira JG, Fernandes T, Scory S (2003) Eutrophication and some European waters of restricted exchange. Continental Shelf Research 23:1635-1671.

Thingstad TF, Skjoldal EF, Bohne RA (1993) Phosphorus cycling and algal-bacterial competition in Sandsfjord, western Norway. Mar. Ecol. Prog. Ser. 99(3): 239-259.

Tibbetts SM, Lall SP, Milley JE (2005) Effects of dietary protein and lipid levels and DP DE^{-1} ratio on growth, feed utilization and hepasomatic index of juvenile haddock, (*Melanogrammus aeglefinus* L.) Aquaculture Nutrition 11: 67-75.

Tlusty MF, Preisner K (2005) Organic matter production of American lobsters (Homarus americanus) during impoundment in Maine, United States. New Zealand Journal of Marine and Freshwater Research 39:471-484.

True B, Johnson W, Chen SL (2004) Reducing phosphorus discharge from flow-through aquaculture I: facility and effluent characterization. Aquacultural Engineering 32:129-144.

Tuya F, Sanchez-Jerez P, Dempster T, Boyra A, Haroun RJ (2006) Changes in demersal wild fish aggregations beneath a sea-cage fish farm after the cessation of farming. Journal of Fish Biology 69:682-697.

Velvin R (1999) Environmental effects from Fish farming. In Poppe T (Ed.), Textbook of Fish Health and Fish Diseases. Universitetsforlaget, Oslo, Norway, pp. 340-347. In Norwegian.

Vezzulli L, Chelossi E, Riccardi G, Fabiano M (2002) Bacterial community structure and activity in fish farm sediments of the Ligurian sea (Western Mediterranean). Aquaculture International 10: 123-141.

Vezzulli L, Pruzzo C, Fabiano M (2004) Response of the bacterial community to in situ bioremediation of organic-rich sediments. Marine Pollution Bulletin 49:740-751.

Vita R, Marin A, Madrid JA, Jimenez-Brinquis B, Cesar A, Marin-Guirao L (2004) Effects of wild fishes on waste exportation from a Mediterranean fish farm. Marine Ecology-Progress Series 277:253-261.

Vizzini S, Mazzola A (2006) The effects of anthropogenic organic matter inputs on stable carbon and nitrogen isotopes in organisms from different trophic levels in a southern Mediterranean coastal area. Science of the Total Environment 368:723-731.

Vizzini S, Savona B, Caruso M, Savona A, Mazzola A (2005) Analysis of stable carbon and nitrogen isotopes as a tool for assessing the environmental impact of aquaculture: a case study from the western Mediterranean. Aquaculture International 13:157-165.

Vollenweider RA, Kerekes J (1980) The loading concept as basis for controlling eutrophication philosophy and preliminary results of the OECD program on eutrophication. Progress in Water Technology 12(2): 5-38.

Wallace RA, Sanders GP, Ferl RJ (1991) Biology: The science of life. 3rd ed. Harper Collins Publishers, New York.

Wang Y, Guo J, Bureau DP, Cui Z (2006) Replacement of fish meal by rendered animal protein ingredients in feeds for cuneate drum (*Nibea miichtioides*). Aquaculture 252: 476-483.

Webb AP, Eyre BD (2004) Effect of natural populations of burrowing thalassinidean shrimp on sediment irrigation, benthic metabolism, nutrient fluxes and denitrification. Mar Ecol Prog Ser 268: 205-220.

Wilkie MP (2002) Ammonia excretion and urea handling by fish gills: Present understanding and future research challenges. Journal of Experimental Zoology 293: 284-301.

Wilson RP (1994) Utilization of dietary carbohydrate by fish. Aquaculture 124: 67-80.

Wilson S, Blake C, Berges JA, Maggs CA (2004). Environmental tolerances of free-living coralline algae (maerl): implications for European marine conservation. Biological Conservation 120: 279-289.

Wu RSS (1995) The environmental impact of marine fish culture: towards a sustainable future. Mar. Pollut. Bull. 31: 159-166.

Xie B, Ding ZH, Wang XR (2004) Impact of the intensive shrimp farming on the water quality of the adjacent coastal creeks from Eastern China. Marine Pollution Bulletin 48:543-553.

Xu Z, Lin X, Lin Q, Yang Y, Wang Y (2007) Nitrogen, phosphorus, and energy waste outputs of four marine cage-cultured fish fed with trash fish. Aquaculture 263: 130-141.

Xue M, Luo L, Wu X, Ren Z, Gao P, Yu Y, Pearl G (2006) Effects of six alternative lipid sources on growth and tissue fatty acid composition in Japanese sea bass (*Lateolabrax japonicus*). Aquaculture 260: 206-214.

Yang YF, Fei XG, Song JM, Hu HY, Wang GC, Chung IK (2006) Growth of Gracilaria lemaneiformis under different cultivation conditions and its effects on nutrient removal in Chinese coastal waters. Aquaculture 254:248-255.

Zhou Y, Yang HS, Zhang T, Liu SL, Zhang SM, Liu Q, Xiang JH, Zhang FS (2006)a Influence of filtering and biodeposition by the cultured scallop Chlamys farreri on benthic-pelagic coupling in a eutrophic bay in China. Marine Ecology-Progress Series 317:127-141.

Zhou Y, Yang HS, Hu HY, Liu Y, Mao YZ, Zhou H, Xu XL, Zhang FS (2006)b Bioremediation potential of the macroalga Gracilaria lemaneiformis (Rhodophyta) integrated into fed fish culture in coastal waters of north China. Aquaculture 252:264-276.

Zhou Y, Yang HS, Zhang T, Qin PB, Xu XL, Zhang FS (2006)c Density-dependent effects on seston dynamics and rates of filtering and biodeposition of the suspension-cultured scallop Chlamys farreri in a eutrophic bay (northern China): An experimental study in semi-in situ flow-through systems. Journal of Marine Systems 59:143-158.

Yang L, Zhou HD, Moccia R (2006) Membrane filtration coupled with chemical precipitation to treat recirculating aquaculture system effluents. Journal of Environmental Quality 35:2419-2424.

Yap LG, Azanza RV, Talaue-McManus L (2004) The community composition and production of phytoplankton in fish pens of Cape Bolinao, Pangasinan: a field study. Marine Pollution Bulletin 49: 819-832.

Yokoyama H (2002) Impact of fish and pearl farming on the benthic enviroments in Gokasho Bay: Evaluation from seasonal fluctuations of the macrobenthos. Fisheries Science 68:258-268.

Yokoyama H (2003) Environmental quality criteria for fish farms in Japan. Aquaculture 226:45-56.

Yokoyama H, Inoue M, Abo K (2004) Estimation of the assimilative capacity of fish-farm environments based on the current velocity measured by plaster balls. Aquaculture 240: 233-247.

Yokoyama H, Katsuyuki A, Ishihi Y (2006) Quantifying aquaculture-derived organic matter in the sediments in and around a coastal fish farm using stable carbon and nitrogen isotope ratios. Aquaculture 254: 411-425.

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Appendix Table 1

Short summaries of the most relevant articles retrieved from the ISI Web of Science databases using the search phrase "Aquaculture AND Eutrophication" for the years 2000-2006.

Summary	Reference
Taxonomic composition, abundance and biomass of mesozooplankton in the Zhelin Bay - an estuary with intensive aquaculture "Eutrophication due to aquaculture and other sources jeopardize aquaculture" (China). Species diversity, biomass, and total abundance of mesozooplankton were greater in the samples collected outside the bay than inside the bay	Dong et al. (2006)
Hypereutrophication events in the Ca'Pisani lagoons associated with intensive aquaculture Hypereutrophication event in Italian lagoons associated with intensive aquaculture. Invalidation of ecosystems in hypereutrophic lagoons due to overloading of organic matter, nutrients, and labile sulphides.	Sorokin et al. (2006)
Influence of filtering and bio deposition by the cultured scallop *Chlamys farreri* on benthic-pelagic coupling in a eutrophic bay in China Suspended filter feeders (scallops) enhance pelagic-benthic coupling and exert basin-scale impacts. Bivalve filtering-biodeposition enhances the deposition of suspended material and the flux of C, N and P to the benthos.	Zhou et al. (2006)a
A dynamic CSTT model for the effects of added nutrients in Loch Creran, a shallow fjord Simple 3 box model to assess the capacity of fjords to assimilate nutrients from fish farms. The model's biological state variables are chlorophyll, dissolved inorganic nitrogen and dissolved inorganic phosphorus, and it includes a simple run-off model to convert rainfall into river discharge.	Laurent et al. (2006)
Reduction of land based fish-fanning impact by phytotreatment pond system in a marginal lagoon area Phytoplankton pond reduce fish fanning impact in marginal lagoon area on Italian west coast	Gennaro et al. (2006)
Growth of *Gracilaria lemaneiformis* under different cultivation conditions and its effects on nutrient removal in Chinese coastal waters Seaweeds on rafts could be effective in removing waste nutrients	Yang et al. (2006)
Fish farming effects on chemical and microbial variables of the water column: A spatio-temporal study along the Mediterranean Sea	Pitta et al. (2006)

Dilution and grazing are probably both responsible for the lack of detectable signs of eutrophication	
Temporal changes in the polychaete infaunal community surrounding a Hawaiian mariculture operation	Lee et al. (2006)
Reduced infaunal diversity due to fish farm waste deposition on Hawaii. Such effects may be diluted at open-ocean locations	
Bioremediation potential of the macroalga *Gracilaria lemaneiformis* (Rhodophyta) integrated into fed fish culture in coastal waters of north China	Zhou et al. (2006)b
Exploit fish farm nutrients as a resource. Extrapolation suggest that 1 ha cultivation of seaweed can sequester 0.22 tonnes of nitrogen and 0.03 tonnes of phosphorus annually	
Dynamic changes of total bacteria and *Vibrio* in an integrated seaweed-abalone culture system	Shao et al. (2006)
Bacterial community modified in polyculture of fish and seaweeds	
Exploring Northeast American and Asian species of *Porphyra* for use in an integrated finfish-algal aquaculture system	Carmona et al. (2006)
Use of *Porphyra* for bioremediation of affected coastal waters. Under the experimental conditions, *Porphyra* spp. removed 70-100 % of N within 3-4 days at N concentrations up to 150 µmol l^{-1}, but was less efficient at removing inorganic phosphorus (35-91 % removal).	
Density-dependent effects on seston dynamics and rates of filtering and biodeposition of the suspension-cultured scallop *Chlamys farreri* in a eutrophic bay (northern China): An experimental study in semi-in situ flow-through systems	Zhou et al. (2006)c
Scallops enhance pelagic-benthic coupling by enhancing the deposition of suspended organic material.	
Inorganic nitrogen control in wastewater treatment ponds from a fish farm (Orbetello, Italy): Denitrification versus *Ulva* uptake	Bartoli et al. (2005)
Ulva based phytotreatment. Effective strategy for dissolved inorganic nitrogen removal. Assimilation of up to 50 % of DIN in effluent from land based fish farm.	
Environmental consequence analyses of fish farm emissions related to different scales and exemplified by data from the Baltic - a review	Gyllenhammar and Hakanson (2005)
Environmental consequence analysis of fish farm emissions in the Baltic Sea. At the smallest scale (< 1 ha) the "footprint" expressing the impact area in the sediment below cages correspond approximately to the size of a football field if the annual fish production is about 50 tonnes. At the local scale (1 ha to 1 km^2 there exist a simple load diagram (effect-load-sensitivity), which makes possible an estimate of the maximum allowable fish production. At the regional scale (100-10000 km^2) it is possible to create negative nutrient pulses by using more than 1.1 g wet weight regionally caught wild fish per g feed for	

the cultivated fish. At the international scale (> 10000 km^2) related to the Baltic Proper, the contribution from fish farms to the overall nutrient fluxes are very small.

Quantification of in situ nutrient and heavy metal remediation by a small pearl oyster (*Pinctada imbricata*) farm at Port Stephens, Australia	Gifford et al. (2005)
Use of pearl oyster in bioremediation of coastal waters. Each tonne of pearl oyster material harvested resulted in approximately 703 g metals, 7452 g nitrogen and 545 g phosphorus being removed from the waters of Port Stephens.	
Coastal water quality assessment in the Yucatan Peninsula: management implications	Herrera-Silvera et al. (2004)
Coastal water quality assessment. Establish base lines for selected water quality parameters.	
Loading of nutrient from a land-based fish farm (Orbetello, Italy) at different times	Porrello et al. (2005)
Loading of nutrients from a land based fish farm into lagoon. Sequence of N and P excretion. Total dissolved nitrogen was excreted shortly after feeding, with a peak 4 h after the end of feeding. Most of the total dissolved phosphorus was excreted after many hours, with a peak 8 h after the end of feeding.	
Development of mariculture and its impacts in Chinese coastal waters	Feng et al. (2004)
China has the highest mariculture production in the world. 11 315 000 tonnes on 1 286 000 ha. Increasing concern about impact on the coastal environment. Authors suggest moving facilities to deeper waters and use of fallowing.	
Response of the bacterial community to in situ bioremediation of organic-rich sediments	Vezzulli et al. (2004)
Assessing the potential of bioremediation of sediment eutrophication by bio-augmentation (bio-fixed microorganisms) and bio-stimulation (oxygen release compounds).	
Study on limiting nutrients and phytoplankton at long-line-culture areas in Laizhou Bay and Sanggou Bay, north-eastern China	Liu et al. (2004)
Complicated correlations between phytoplankton and nutrients together with violent fluctuations in dissolved inorganic concentrations indicate fragile ecosystem stability. Exposed locations and less crowded settings of rafts allow better water exchange and reduced effect of aquaculture on the environment.	
Reducing phosphorus discharge from flow-through aquaculture I: facility and effluent characterization	True et al. (2004)
Transport and removal are important factors affecting P discharge	
Environmental capacity of receiving water as basis for regulating	Sumagaysay-Chavoso et al.

intensity of milkfish (*Chanos chanos* Forsskal) culture	(2004)
Environmental capacity in terms of the maximum amount of dissolved inorganic nitrogen or dissolved inorganic phosphorus input to the system was predicted using regression analysis and following set criteria for nutrients, i.e. nitrite, nitrate and phosphate. At present, the estuarine water quality has already reached its environmental capacity during dry months. About 945 ha of commercial milkfish ponds are operating upstream, mostly as extensive systems	
Sustainable impact of mussel farming in the Adriatic Sea (Mediterranean Sea): evidence from biochemical, microbial and meiofaunal indicators	Danovaro et al. (2004)
The indicators based on the biochemical compositions of the sediment organic matter and the microbial parameters also show no evidence of the eutrophication process, except as a slight increase in the bacterial density in the sediments beneath the farm during the period of highest mussel stocks	
Environmental impact assessment of salmonid feeds using Life Cycle Assessment (LCA)	Papatryphon et al. (2004)
The assessment revealed that the use of fishery resources and nutrient emissions at the farm (such as eutrophication potential) contribute most to the potential environmental impacts of salmonid aqua feeds	
Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: A review	Newell (2004)
Natural and aquaculture-reared stocks of bivalves are potentially a useful supplement to watershed management activities intended to reduce phytoplankton production by curbing anthropogenic N and P inputs to eutrophied aquatic systems	
Environmental impact of aquaculture-sedimentation and nutrient loadings from shrimp culture of the southeast coastal region of the Bay of Bengal	Das et al. (2004)
Nutrient loading was found to decrease with distance from the shrimp farm discharge unit in estuarine water	
Solving the coastal eutrophication problem by large scale seaweed cultivation	Fei (2004)
Eutrophication is becoming a serious problem in coastal waters in many parts of the world. It induces the phytoplankton blooms including Red Tides, followed by heavy economic losses to extensive aquaculture areas. Some cultivated seaweeds have very high productivity and could absorb large quantities of N, P, CO_2, produce large amounts of O_2 and have excellent effect on decreasing eutrophication . The author believes that seaweed cultivation in large scale should be a good solution to the eutrophication problem n coastal waters	
Impact of the intensive shrimp farming on the water quality of the adjacent coastal creeks from Eastern China	Xie et al. (2004)
With water quality criteria based on local laws, a significant pollution	

was observed in the area, with inorganic nitrogen being the most significant pollutant, followed by chemical oxygen demand and inorganic phosphorus

Aquaculture - profitable environmental remediation? — Gifford et al. (2004)

This paper proposes pearl oyster deployment as a novel bioremediation technology for impacted sites to remove toxic contaminants, reduce nutrient loads and lower concentrations of microbial pathogens

Eutrophication and some European waters of restricted exchange — Tett et al. (2003)

Regions of restricted exchange (RREs) are an important feature of the European coastline. They are historically preferred sites for human settlement and aquaculture and their ecosystems, and consequent human use, may be at risk from eutrophication

Food poisoning associated with biotoxins in fish and shellfish — Brett (2003)

In recent times, the number of blooms of algae that produce toxins has increased in frequency, intensity and geographical distribution. This may be a result of increased awareness, aquaculture, eutrophication, or transport of cysts in ship ballast

Environmental management of marine aquaculture in Tasmania, Australia — Crawford (2003)

A new project is investigating system-wide effects of salmon farming on the environment, in particular, increased release of nutrients into waterways. This includes monitoring dissolved oxygen, nutrients and phytoplankton, modelling the system, and investigating ecological indicators of eutrophication

Aquafeeds and the environment: policy implications — Tacon and Forster (2003)

Aquaculture feeds and feeding regimes can play a major role in determining the quality and potential environmental impact or not on finfish and crustacean farm effluents. This is particularly true for those intensive fanning operations employing open aquaculture production systems, the latter including net cages/pen enclosures placed in rivers, estuaries or open water bodies, and land-based through-flow tank, raceway or pond production systems

Environmental impacts of coastal Aquaculture in eastern Mediterranean bays the case of Astakos Gulf, Greece — Belias (2003)

No clear eutrophication incidents have been identified, although the water column near the fish farms was enriched in nutrients and organic carbon. The most significant influence concerns the near bottom water layer. The environmental impact depend on the amount of food given to fishes, the mode of feeding, the fish density in cages, the annual production and the years of unit operation

A model of fish preference and mortality under hypoxic water in the coastal environment — Karim et al. (2003)

Eutrophication and associated occurrence of hypoxic condition could cause significant damage to marine ecosystems, resulting in

considerable economic losses to fisheries and aquaculture and is a major source of stress that fish often have to contend with in order to survive. Shallow coastal areas are of great importance for the special nursery of fish and shellfish and land reclamation in these areas cause strong damage to fisheries	
Cultivation of seaweeds: current aspects and approaches In comparison with total annual commercial production of fish, crustaceans and molluscs of about 120 x 10^6 tonnes, of which one-third is produced by aquaculture, the production of seaweeds is about 10 x 10^6 tonnes wet weight; the majority of this comes from culture-based systems. Mass culture of commercially valuable seaweed species is likely to play an increasingly important role as a nutrient-removal system to alleviate eutrophication problems due to fed aquaculture	Luning and Pang (2003)
Reduction of aquaculture wastewater eutrophication by phytotreatment ponds system II. Nitrogen and phosphorus content in macroalgae and sediment The presence of intensive fish farms in the Orbetello Lagoon (Italian West Coast) has produced a heavy impact on the environment and has led to a large-scale cyclic development of opportunistic macroalgae	Porello et al. (2003)a
Reduction of aquaculture wastewater eutrophication by phytotreatment ponds system I. Dissolved and particulate nitrogen and phosphorus Lagooning had a strong effect on the effluent despite the short residence time	Porello et al. (2003)b
The influence of fish cage aquaculture on pelagic carbon flow and water chemistry in tidally dominated mangrove estuaries of peninsular Malaysia A crude estimate of the contribution of fish cage inputs to the estuaries shows that fish cages contribute only approximately 2 % of carbon, but 32-36 % of nitrogen and 83-99 % of phosphorus to these waters, relative to phytoplankton and mangrove inputs. Preliminary sampling did not reveal any large-scale eutrophication due to the cages	Alongi et al. (2003)
Characterization of effluent from an inland, low-salinity shrimp farm: what contribution could this water make if used for irrigation The potential benefit of having nutrient enriched wastewater to irrigate field crops was substantial, supplying between 20 and 31 % of the necessary nitrogen fertilizer for wheat production	McIntosh and Fitzsimmons (2003)
The continental shelf benthic ecosystem: current status, agents for change and future prospects Continental shelf benthic ecosystems play an important role in the economy of many coastal states through provision of food, non-living resources and through control of climate. Changes in the status of these ecosystems can be expected to have important economic and social consequences. Agents that could induce change include climate and	Hall (2002)

oceanography, hydrology (river discharge), land-use and waste disposal practices, fishing, aquaculture and extraction of non-living resources

Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences — Anderson et al. (2002)

The sources of nutrients potentially stimulating algal blooms include sewage, atmospheric deposition, groundwater flow, as well as agricultural and aquaculture runoff and discharge. On a global basis, strong correlations have been demonstrated between total phosphorus inputs and phytoplankton production in freshwaters, and between total nitrogen input and phytoplankton production in estuarine and marine waters. Shifts in species composition have often been attributed to changes in nutrient supply ratios, primarily N:P or N:Si. Recently this concept has been extended to include organic forms of nutrients, and an elevation in the ratio of dissolved organic carbon to dissolved organic nitrogen (DOC:DON) has been observed during several recent blooms. Alternate modes of nutrition such as heterotrophy and mixotrophy are now recognized as common among HAB species. Nutrient enrichment has been strongly linked to stimulation of some harmful species, but for others it has not been an apparent contributing factor. The overall effect of nutrient over-enrichment on harmful algal species is clearly species specific

Marine eutrophication and benthos: the need for new approaches and concepts — Grall and Chauvaud (2002)

We have to better understand coastal eutrophication and develop tools for quantifying the impacts; in order to achieve this goal, some possible directions proposed are: integrated studies leading to new concepts, model development based on these concepts and finally comparison of various ecosystems on a global scale

Predicting the environmental response of fish farming in coastal areas of the Åland archipelago (Baltic Sea) using management models for coastal water planning — Nordvarg and Hakanson (2002)

Eutrophication is recognised as a major threat to marine ecosystems. Fish farms contribute to eutrophication since they emit various forms of nutrients. Consequently, it is important to determine environmental impact caused by fish farms

High ammonium production from sediments in hypereutrophic shrimp ponds — Burford and Longmore (2001)

Intensive shrimp ponds are hypereutrophic ecosystems with high nutrient loading rates. The discharges of pond water mean that sediment processes in shrimp ponds ultimately affect the nitrogen loads discharged into the aquatic environment, exacerbating the potential for eutrophication caused by shrimp farming activities

The effects of fish farm effluents on the water quality in the Åland archipelago, Baltic Sea — Nordvarg and Johansson (2002)

The main aim of this study was to increase our knowledge on how fish farm effluents influences the surface water quality in the Åland

archipelago (Baltic Sea). Extensive field studies were conducted between 1997 and 1999 in nine defined coastal areas. The most significant effects on total phosphorus concentrations, periphytic growth and phytoplankton standing crop was observed in two coastal bays with small to moderate fish farm production (approximately 70 tonnes per year). The significant effects were mainly due to the small size of the bay (0.48 and 0.73 km^2) and shallow mean water depth (3 and 3.8 m), rather than long water retention times (2 and 6 days). Due to a very large fish farm production (800 tonnes per year) we also observed significant effects in a coastal area of moderate size (8 km^2).

Ten-year record of *Thalassiosira nordenskioeldii* population dynamics: comparison of aquaculture and non-aquaculture sites in the Quoddy Region — Smith et al. (2001)

The analysis suggests that the diatom *Thallassiosira nordenskioldii* has increased its contribution to the phytoplankton community at the sites located near aquaculture activity, one of which also became less saline. These changes may be related to aquaculture activities, although the causative factor(s) is yet unidentified. The trends suggest that this diatom is sensitive to changes in the local environment

Our evolving conceptual model of the coastal eutrophication problem — Cloern (2001)

How does anthropogenic nutrient enrichment cause change in the structure or function of nearshore coastal ecosystems? The early, phase one conceptual model was influenced by limnologists who studied lake eutrophication by the 1960s. This model emphasized changing nutrient input as a signal, and responses to that signal as increased phytoplankton biomass and primary production, decomposition and depletion of oxygen in bottom waters. The contemporary, phase II conceptual model reflects the differences in the responses between lakes and coastal-estuarine ecosystems to nutrient input.

Sediment mineralization, nutrient fluxes, denitrification and dissimilatory nitrate reduction to ammonium in an estuarine fjord with sea cage trout farms (Horsens fjord, Denmark) — Christensen et al. (2000)

The ecological consequence of a shift in the relative importance of processes in response to organic matter loading was a reduced nitrogen removal by denitrification and an increased efflux of NH_4 to the water column, resulting in stimulation of pelagic primary production and increased nitrogen retention by the ecosystem. The excess nitrogen input to the fjord from the trout farms corresponded to approximately 12 % of the total nitrogen load from land during the summer months

Appendix Table 2

Short summaries of the most relevant articles retrieved from the ISI Web of Science databases using the search phrase "Aquaculture AND Pelagic AND Impact" for the years 2000-2006.

Summary	Depth	Current speed	Reference
Ammonium (NH_4) concentration was measured downstream from a well flushed salmonid farm with a standing fish stock of 576 mt. The dissolved N-loading from the farm was 101 kg day^{-1}. The NH_4 concentration sampled at 3 m depth was 10 µM in the cages, decreased to 8 µM at 5 m downstream, 4 µM at 10 m, and then the decrease slowed down but approached the background level of approximately 1.5 µM.	11 m	0-35 cm s^{-1} (tidal cycle)	Merceron et al. (2002)
Macroalgal and phytoplankton bioassays were used to monitor nutrient release from sea bream and sea bass fish farms in the Mediterranean. The feed input varied between 520 and 2749 mt per year. For both types of assays growth was stimulated compared with the reference sites for distances up to 150 m downstream in the dominating current direction. The incubation time was 3-6 days.	15-39 m		Dalsgaard and Krause-Jensen (2006)
43 salmon farm-sites in Chile were studied. 29 facilities had an annual production in the range 1500-2500 tonnes, while the remaining 14 had a production of <500 tonnes per year. Nutrients and chlorophyll a were measured in the water column from integrated samples taken from 15 cm, 10 m and 15 m depth. They measured close to the cages and at a site 1 km away. They found no significant effects on nitrate, ammonia, phosphate and chlorophyll a concentration compared with the reference site.	15-94 m	3-30 cm s^{-1}	Soto and Norambuena (2004)
This is a study of 5 medium sized sea bream/sea bass cage farms at semi-offshore sites in the Mediterranean. The standing stock of fish biomass varied between 43 and 550 mt. They measured nitrate and phosphate at the farm site and at a control-site located 3 nautical miles (5.6 km) away in opposite direction to the prevailing current. The authors found no significant differences between farm and control sites.	21-37 m		Maldonado et al. (2005)
In the Baltic Sea some cage farms are located in small and shallow coastal bays (0.48-0,73 km^2 and 3-4 m depth) with water retention times of 2-	3-4 m		Nordvarg and Johansson

6 days. In these areas significant increases in total phosphorus concentration and phytoplankton biomass was observed, even if the aquaculture production was low (≈ 70 mt year^{-1}).			(2002)
Three aquaculture sites along the Mediterranean (Spain, Italy and Greece) were studied. Annual productions raged between 260 and 1150 tonnes of sea bream and sea bass. Samples for analysis of nutrients, chlorophyll a, particulate organic carbon and nitrogen, heterotrophic bacteria and cyanobacteria, were taken during the warm and most productive period, when impact from fish cages is greatest. Samples were collected along a transect of four stations, starting at the edge of the cages and downstream of the prevailing current at 8-30 m, 45-60 m and at a distance >450 m, which constituted the reference station. No significant changes were detected in integrated samples from the whole water column (0-30 m). Changes were found in surface samples in the immediate vicinity of the cages, with increased concentration of ammonium, POC and PON, and decreased cyanobacteria densities.	Ca 30 m		Pitta et al. (2006)
Bolinao Bay in the Philippines is a site for extensive and intensive aquaculture of milkfish (*Chanos chanos*) in cages and net-pens. During the wet monsoon season salinity decreases and nutrient concentration increases due to increased river transport. There is a seasonal succession of phytoplankton in the area from diatoms in the dry season to dinoflagellates in the wet season. In 2002 a bloom of the dinoflagellate *Prorocentrum minimum* was recorded for the first time during the dry season (January/February). As the bloom declined the oxygen concentration decreased to 2 mg per litre in the seawater due to microbial breakdown of the dying bloom. This led to a massive fish kill in aquaculture facilities and also reef fish, octopus and urchins were washed on the shore. The net-pen structures diminish the speed of the water with 45-70 %, and in addition the current speeds are lower in the dry season. Inside the pens a maximum speed of 8.5 cm s^{-1} was measured during the wet season, and 0.9-2.9 cm s^{-1} during the dry season. More than a thousand fish pens and cages were maintained on an area about 24 km long and 3 km wide (an inshore	46 m (average)	1-10 cm s^{-1}	Yap et al. (2004) Azanza et al. (2005)

channel) at this time.	
The effects of fish-farming on local landings in five regions of the Aegean and Ionian Seas (eastern Mediterranean basin) were analysed, together with certain parameters related to environmental changes and fishing fleet activity. Data from time series of landings, fish farming production, fishing fleet, temperature and rainfall (1984-2001) were analysed based on the Minimum-maximum Auto-correlation Factor Analysis (MAFA). Results suggested that increased fish-farming activity in enclosed, oligotrophic areas could imply an increase in fisheries landings. Intense fish farming could be an important explanatory factor regarding changes in fisheries production at certain areas.	Machias et al. (2006)

Appendix Table 3

Short summaries of the most relevant articles retrieved from the ISI Web of Science databases using the search phrase "Aquaculture AND Benthic AND Impact" for the years 2000-2006.

Summary	Reference
Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages (Tasmania)	Macleod et al. (2006)
Rate and extent of recovery was affected by farm location, initial impact of the sediments, and length of fallow period. Initial recovery was faster at the more sheltered site than at the more exposed site, possibly reflecting differences in environmental resilience with the more sheltered location better able to assimilate organic inputs. The findings of this study suggest that the recovery response of benthic communities can be predicted once baseline conditions are understood	
Assessment of site specific benthic impact of floating cage fanning in the eastern Hios island, Eastern Aegean Sea, Greece	Klaoudatos et al. (2006)
Marked changes in species numbers, diversity and abundance of benthic fauna between farm and control sites. Species richness, diversity and evenness were higher at the control sites whereas numerical abundance was higher at the farm sites. K-dominance curves suggest a minor impact on the benthic community at the farm sites and temporal changes on macrobenthic assemblages	
The effects of anthropogenic organic matter inputs on stable carbon and nitrogen isotopes in organisms from different trophic levels in a southern Mediterranean coastal area	Vizzini and Mazzola (2006)
Benthic component more sensitive to organic pollution than nektobenthic species. Effects of anthropogenic activities detectable over a wide area. Study site characterised by wide waste dispersal, which brings a reduction in impact in the area directly affected and enlarges the area of moderate impact	
Effects of mussel fanning on macrobenthic community structure in Southeastern Brazil	da Costa and Nalesso (2006)
Mussel culture had no negative impact on the macrobenthic community	
Density-dependent effects on seston dynamics and rates of filtering and biodeposition of the suspension-cultured scallop Chlamys farreri in a eutrophic bay (northern China): An experimental study in semi-in situ flow-through systems	Zhou et al. (2006)c
Scallops enhance pelagic-benthic coupling	
Changes in demersal wild fish aggregations beneath a sea-cage fish farm after the cessation of farming	Tuya et al. (2006)
Changes in benthic fauna and fish assembly	

A review of the impacts of salmonid farming on marine coastal ecosystems in the southeast Pacific	Buschmann et al. (2006)
Current evidence indicates a significant loss of benthic biodiversity and localized changes in physico-chemical properties. Presence of salmon farms significantly increases the pulses and density of dinoflagellates. It is urgent that an ecosystem approach be implemented to assess all impacts of salmonid farming on coastal ecosystems in southern Chile	
Impact of clam and mussel farming on benthic metabolism and nitrogen cycling, with emphasis on nitrate reduction pathways	Nizzoli et al. (2006)
Nitrate reduction processes eliminated fixed N from the clam farm sediments via coupled nitrification-denitrification, while the dominance of dissimilatory nitrate reduction to ammonium at the mussel farm resulted in a net N input to the sediment compartment. Large differences in the impact of clam and mussel farming can be explained by the fact that infaunal clams stimulate both organic matter and oxygen to the sediment, while suspended mussels enhance only organic matter inputs	
Use of stable isotopes to investigate dispersal of waste from fish farms as a function of hydrodynamics	Sara et al. (2006)
Increasing water current velocities enlarge the relative area of influence of the cages, particularly on sediments	
Benthic impacts of fish farming: Meta-analysis of community and geochemical data	Kalantzi and Karakassis (2006)
Overall conclusion: the complicated interactions between variables and the lack of data, such as current speed, makes it difficult to set common or uniform environmental quality standards for benthic effects of fish farming. Differences between geographic regions, depth zones and sediment types should be considered.	
Impact of suspended and off-bottom Eastern oyster culture on the benthic environment in eastern Canada	Mallet et al. (2006)
Oyster biomass induced increased sedimentation of organic matter at the oyster table site, but there was no indication of organic enrichment in the sediment	
Settling velocity of faecal pellets of gilthead sea bream (*Sparus aurata* L.) and sea bass (*Dicentrarchus labrax* L.) and sensitivity analysis using measured data in a deposition model	Magill et al (2006)
The use of single mean settling velocity in deposition models does not accurately predict the extent of the benthic flux. Species-specific faecal settling rates should be used when modelling polycultures	
Vertical variability of wild fish assemblages around sea-cage fish farms: implications for management	Dempster et al. (2005)
Wild fish consume lost food and assimilate nutrients. Therefore coastal managers should prohibit fishing of large plaktivorous species at farms to fully harness their ability to ameliorate benthic impacts	

Effects of nutrient enrichment derived from fish farming activities on macro-invertebrate assemblages in a subtropical region of Hong Kong	Gao et al. (2005)
Increased total organic carbon, total nitrogen and phosphorus, and greatly reduced N:P ratio at the farm site compared with reference stations. The diversity of the macro-fauna was significantly reduced and faunal diversity was negatively correlated with nutrient level	
Effects of mussel culture husbandry practices on various benthic characteristics	Miron et al. (2005)
No strong relationship between benthic parameters and husbandry practices	
Organic matter production of American lobsters (*Homarus americanus*) during impoundment in Maine, United States	Tlusty and Preisner (2005)
The level of organic matter production is below the level produced by other aquaculture operations, and that at which benthic impacts might be expected	
The environmental impact of Mediterranean cage fish farms at semi-exposed locations: does it need a re-assessment?	Maldonado et al. (2005)
Only one of five farms had a detectable impact on benthic macro-invertebrate community immediately under the cages. Medium sized farms on semi-exposed locations have less environmental impact than traditional farms located in shallow, sheltered sites	
Acoustical and sedimentological characterization of substrates in and around sheltered and open-ocean mussel aquaculture sites and its bearing on the dispersal of mussel debris	Hartstein (2005)
Lack of mussel debris indicates there is sufficient energy to transport and dissipate the shell and biodeposits over a wide area with little impact on the natural sediment	
Analysis of stable carbon and nitrogen isotopes as a tool for assessing the environmental impact of aquaculture: a case study from the western Mediterranean	Vizzini et al. (2005)
Isotopes revealed widespread aquaculture waste in the study site and a large contribution of waste to the benthic food web. In contrast, both nekton-benthic and pelagic organisms seemed to be less influenced by discharge	
Preliminary study on the effects of exclusion of wild fauna from aquaculture cages in a shallow marine environment	Felsing et al. (2005)
In natural coastal systems of western Australia or comparable environments wild fish are potentially important consumers of cage aquaculture waste materials	
Effects of wild fishes on waste exportation from a Mediterranean fish farm	Vita et al. (2004)
Wild fishes play an important role in the recycling of organic matter of	

the sediment, and the trophic role of wild fishes should be considered when evaluating the environmental impact of fish farms

Sustainable impact of mussel farming in the Adriatic Sea (Mediterranean Sea): evidence from biochemical, microbial and meiofaunal indicators — Danovaro et al. (2004)

Biochemical composition of the sediment organic matter and microbial parameters show no evidence of eutrophication. No effects on the benthic faunal indicators

Assessment of long term change in sediment condition after organic enrichment: defining recovery (Tasmania) — Macleod et al. (2004)

Comparison with results from impact studies in Scotland, Canada and Norway suggests that the sediments were considerably less impacted in Tasmania than in northern temperate areas. Establishment of a local baseline standard is important to evaluate both impact and recovery

Dynamic response of a mud snail *Nassarius sinusigerus* to changes in sediment biogeochemistry — Eden et al. (2003)

Distribution of the snail *Nassarius sinusigerus* is determined by the balance between the attraction of the organically rich sediments below cages and deterrence due to deleterious sediment geochemistry (anoxia and sulphides)

Environmental quality criteria for fish farms in Japan — Yokoyama (2003)

Environmental deterioration around fish farms has been widespread in Japanese coastal areas. Law to ensure sustainable aquaculture production was enacted in 1999. Criteria based on three indicators; 1) dissolved oxygen content in the water of the cages, 2) acid volatile sulphide content in the sediment, 3) the occurrence of macrofauna under the fish cages. Biomass of macrobenthos peaked in sediments containing 1.2 mg/g total nitrogen, where the majority of aerobic mineralization is supposed to occur. Animals were scarcely found in sediments with acid volatile sulphide > 1.7 mg/g, suggesting this is a critical condition

Environmental management of marine aquaculture in Tasmania, Australia — Crawford (2003)

Local impacts on the seabed around salmon farms are monitored using video footage, analysis of benthic invertebrate infauna, and chemical measures (redox and organic matter). Monitoring of shellfish farms is minimal because our research has shown that shellfish culture is having little impact on the environment

Organic enrichment of sediments from salmon farming in Norway: environmental factors, management practices, and monitoring techniques — Carroll et al. (2003)

To determine the relationship between environmental variables, management regimes, and levels of environmental impact

Environmental impacts of coastal Aquaculture in eastern Mediterranean bays the case of Astakos Gulf, Greece	Belias et al. (2003)
A strong debate has risen about the environmental impacts of aquacultures in the coastal areas. A sludge blanket enriched in colloidal organic carbon and trace metals from unused fish food cover parts of the seabed. The biodegradation of this sludge leads to anoxia, formation of undesirable gases and remobilization of metals, and extinction of benthic fauna. Hydrology and geomorphology are critical factors for environmental quality	
Effects of shellfish farming on the benthic environment	Crawford et al. (2003)a
The benthic infauna did not show clear signs of organic enrichment. Shellfish fanning have little impact, and much less than salmon fanning	
MARICULT Research Programme: background, status and main conclusions	Olsen (2002)
Negative ecological effects caused by enhanced nutrient supply to pelagic ecosystems may occur if the primary production is not being grazed efficiently by zooplankton or benthic grazers	
Bacterial community structure and activity in fish farm sediments of the Ligurian sea (Western Mediterranean)	Vezzulli et al. (2002)
Data suggest a functional stress of bacterial degradation rates and represent a potential valuable environmental index of imbalance between supply and removal of organic matter in eutrophicated environments	
Impact of fish and pearl farming on the benthic environments in Gokasho Bay: Evaluation from seasonal fluctuations of the macrobenthos	Yokoyama (2002)
Large impact of fish farming on the macrofauna, whereas pearl farming causes less effect on the benthic fauna	
Heterotrophic bacteria community and pollution indicators of mussel - farm impact in the Gulf of Gaeta (Tyrrhenian Sea)	La Rosa et al. (2001)
Sediment enrichment of organic compounds and the consequent modification of the characteristics of the benthic environment at the mussel farm determined an increase in aerobic heterotrophic bacteria, and particularly of *Vibrio* density. These bacteria could be good indicators of organic enrichment	
Impacts of biodeposits from suspended mussel (*Mytilus edulis* L.) culture on the surrounding surficial sediments	Chamberlain (2001)
At one site effects were restricted to a radius of 40 m around the farm, while at a second site there were no observed effects of mussel biodeposits on the benthos. We propose that variation in the dispersion caused by local current patterns had a significant influence on the impact	
Assessment of regional benthic impact of salmon mariculture within	Pohle et al. (2001)

the Letang Inlet, Bay of Fundy

Impact on the benthic fauna was evaluated in two areas with intensive salmon net-pen cultures and one without aquaculture. Increased sediment organic matter content and decrease in diversity of benthic organisms was observed

Monitoring for benthic impacts in the southwest New Brunswick salmon aquaculture industry — Janowicz M, Ross J (2001)

Environmental Monitoring Program (EMP) was adopted in 1995. This is a requirement of the aquaculture site licence and is conducted annually at each farmed site

Mitigating the environmental effects of mariculture through single-point moorings (SPMs) and drifting cages — Goudey et al. (2001)

By spreading out the accumulation of organic matter, one can prevent the local environment from being overwhelmed

Impact of mussel (*Mytilus galloprovincialis*) raft-culture on benthic macrofauna, in situ oxygen uptake, and nutrient fluxes in Saldanha Bay, South Africa — Stenton-Dozey et al. (2001)

Macrofauna biomass was reduced and trophic groups and taxa altered

Regulating the local environmental impact of intensive, marine fish farming II. The monitoring programme of the MOM system (Modelling-Ongrowing fish farms-Monitoring) — Hansen et al (2001)

A programme for monitoring the impact from marine fish farms (Norway)

Impact of cage farming of fish on the seabed in three Mediterranean coastal areas — Karakassis et al. (2000)

Similar patterns of succession from the impacted to the normal zones were found, although macrofaunal composition differed among sites. The impacts of fish farming on benthos in the Mediterranean vary considerably depending on site characteristics

Predicting impacts and recovery of marine-farm sites in Stewart Island, New Zealand, from the Findlay-Watling model — Morrisey et al. (2000)

Estimated input of carbon and supply of oxygen to the seabed below an operating salmon farm provided predictions on the severity of impact that compared reasonably well with the observed anoxic nature of the sediment and presence of patches of *Beggiota*-like growths. Accumulation of zinc and copper (derived from food and antifoulants) in sediments below farms may significantly affect recolonisation of sediments by benthic organisms

Impact of fish farming facilities on *Posidonia oceanica* meadows: a review — Pergent-Martini et al. (2007)

The disturbances caused by these fish farms were measured by means of both abiotic (light, sediment, interstitial water) and biotic variables (meadow density, leaf biometry, lepidochronology, primary production,

epiphytes, reserve carbohydrates in the rhizomes), in function of increasing distance from cages and/or inside a geographically close reference site. The results showed significant degradation of these seagrass meadows in all the sectors investigated. When fish farming cages were placed above a *P. oceanica* bed, the meadow was severely degraded or disappeared and the sediment showed a strong increase in organic matter that could lead to anoxia phenomena. The irreversible impact of fish farming projects on P. oceanica meadows requires the application of the precautionary principle. Several recommendations (site selection, preliminary studies and monitoring over time) are suggested in order to enable piscicultural activities to be incorporated in a global process of Integrated Coastal Zone Management.

Seagrass (*Posidonia oceanica*) vertical growth as an early indicator of fish farm-derived stress Marba et al. (2006)

The usefulness of vertical rhizome growth as an early indicator of fish farm impacts to *Posidonia oceanica* meadows was tested by comparing annual estimates of vertical rhizome growth, quantified retrospectively, at distances ranging between 5 and 1200 m from fish cages at four Mediterranean locations (Cyprus, Greece, Italy and Spain). The reconstructed vertical rhizome growth spanned from 19 to 25 years of growth, depending on sites, and the average vertical rhizome growth before the onset of fish farm operations ranged between 4.48 and 8.79 mm yr(-1). The vertical rhizome growth after the onset of farming activities declined significantly (t-test, $P < 0.05$) from the control station (at > 800 m from the farm. Moreover, vertical growth significantly (t-test, $P < 0.05$) declined by about twofold following the onset of fish farm operations for the extant meadow nearest to the cages, as well as those supporting intermediate impacts at distances 35-400 m front the cages. The results obtained confirm that fish farm activities strongly affect seagrass health on the surrounding meadows, and clearly demonstrate the value of reductions in vertical rhizome growth as an early warning symptom of stress and impacts to P. oceanica meadows.

Interactions of Atlantic salmon in the Pacific northwest environment II. Organic wastes. The second paper evaluating the environmental risks associated with the culture of Atlantic salmon in the Pacific Northwest addresses organic wastes released from net-pens. It begins with the minor effects caused by dissolved nutrients from organic waste in the water column, and then follows in some detail with the more significant environmental effects created by the deposition of waste onto the benthos under and adjacent to salmon farms. The information is supplemented with new data from case studies to elucidate the relationship between organic inputs from salmon farms, which cause changes in sediment chemistry resulting in predictable biological responses. The paper continues with a discussion of chemical and biological remediation of sediments near salmon farms, and ends with some conclusions on the varying degrees of risk from the presence and accumulation of organic wastes. Brooks and Mahnken (2003)

Comparison between some procedures for monitoring offshore cage culture in western Mediterranean Sea: Sampling methods and impact indicators in soft substrata Two sampling methods, an array of physical-chemical and biological indicators, and uni-multivariate statistical procedures were compared on the basis of sensitivity to detect any impact and cost-effectiveness criteria, applied to environmental monitoring of fish farming. Sampling was conducted in a western Mediterranean gilthead seabream (*Sparus aurata*) and meagre (*Argyrosomus regius*) farm (San Pedro del Pinatar, SE Spain). Both sampling methods (Van Veen grab vs. SCUBA diving) provided similar results for all the parameters except redox potential. The most sensitive chemical parameters of the sediment were total phosphorus, total ammonium nitrogen and acid volatile sulphide, but the latter was the only one which correlated with the macrobenthic community structure. The application of univariate methods to biological indicators only showed a clear trend in two (AMBI and BENTIX) of the five indices used. Both biotic indices, although statistically correlated with the spatial pattern of the macrobenthic community, provided different quality status categorization that could lead to misinterpretation and so, validation in offshore conditions is required. Nevertheless, multivariate analysis of macrofauna confirmed a noticeable spatial pattern, at the same time that this statistical strategy has been proven and validated in a wide range of marine community studies.

Aguardo-Gimenéz et al. (2007)

Biogeochemical conditions in sediments enriched by organic matter from net-pen fish farms in the Bolinao area, Philippines.

Holmer et al. (2003a)

Sedimentation and sediment metabolism was measured at eight active milkfish fish pens and at one abandoned site. The sediment metabolism decreased with increasing rates of sedimentation indicating that the microbial activity reached a saturation in the fish pen sediments. Anaerobic processes dominated the organic matter decomposition, and sulphate reduction rates are among the highest measured in fish farm sediments. Presence of methane bubbles in the sediments suggests that sulphate reduction and methanogenesis coexisted.

Sulphide invasion in the seagrass Posidonia oceanica at Mediterranean fish farms: assessment using stable sulphur isotopes

Frederiksen et al. (2007)

Seagrass mortality was induced by changes in sediment biogeochemical conditions. Stable sulphur isotopic signals suggested an invasion of sulphide into the seagrasses, probably causing increased mortality of this sensitive seagrass. Due to its slow growth, recovery of lost seagrass meadows is unlikely

Stimulation of sulphate reduction rates in Mediterranean fish farm sediments inhabited by the seagrass *Posidonia oceanica*

Holmer and Frederiksen (2007)

Sulphate reduction rates and biogeochemical parameters of fish farm sediments across the Mediterranean were investigated in the order to

evaluate the potential effects of organic matter inputs on habitat quality for the common seagrass *Posidonia oceanica*. Results show that although the organic matter accumulation was minor at the sites (POC < 2.8% DW), the sulphate reduction rates were high, in particular at the largest farm in Italy (up to 212 mmol m(-2) d(-1)), similar to rates found at shallower, temperate fish farm sites, where higher sedimentation rates can be expected. Sulphate reducing bacteria in these low-organic, carbonate-rich Mediterranean sediments respond strongly to organic matter loadings and cause habitat degradation. Sulphate reduction rates measured in the P. oceanica sediments were among the highest recorded (7.8-42.0 mmol m^{-2} d^{-1}) similar to rates found in degrading meadows impacted by organic matter loadings. As sulphate reduction rates were correlated with the sedimentation rates along the transects rather than organic matter pools this suggests mineralization processes were controlled by organic matter loading in fish farm sediments. The vegetated sediments near the net cages were more reduced due to accumulation of sulphides compared to control sites, which is a possible contributing factor to the observed seagrass decline in the farm surroundings. It is recommended that Mediterranean fish farms are placed in areas with rapid dispersal of particulate waste products to minimize organic matter loading of the sediments and thereby preserve habitat quality for benthic fauna and flora.
