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AQUAGRIS WP2 - Task 2.1:

State-of-the-art review

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Plan

General introduction

The aim of AquAgriS project is to coordinate research on environmental management reform to improve environmental, industrial and economic sustainability in the farming, fisheries and aquaculture (FFA) industries. In this project, “farming” refers to the breeding and raising of terrestrial animals and cultivation of plants; “fisheries” refers to the industrial capture of wild aquatic species; and “aquaculture” refers to the commercial culture of aquatic animals and plants. A major environmental concern is an increase in wastes produced because of FFA activities, which nececitate an explicit management of these wastes for sustainability (Greenpeace, 2008). There are many similarities between these FFA sectors, which provide an opportunity for the crossover of knowledge and information from related topics. As a key objective of this project is to define ways in which the FFA industries can become more sustainable and productive, the present state-of-the-art review is made in order to review the worldwide scientific literature and to document the latest European and worldwide progress in environmental management research and current technologies in the FFA industries. The purpose is to highlight the most important areas for future research, aiding technology transfer between centres of activity and to identify which technologies are at the forefront of current work in the FFA industries and what new technologies are emerging.

Waste could be defined as any material coming from the development of the activity or industry that is discharged to the environment with potential harmful effects on physical, chemical and/or ecological aspects of the environment. This encompasses two kinds of wastes:

- Organic matter and nutrients:

- ❖ Effluents, leakage and emissions from animal production and farmland (pesticides, fertilizer and NO_2^-/NO_3^- leaching into groundwater, etc.)
- ❖ Fertilizer and nitrogen leaching into groundwater
- ❖ Particulate organic matter (faeces or fish scales) and dissolved excretion released by the cultured organisms
- ❖ Wasted feed released into the surroundings by feeding (efficiency < 100%), accident, carelessness, bad husbandry,
- ❖ Dead animals,
- ❖ Human wastes released into the surrounding environment by the workers (trash, sewage, remains from meals)

- ❖ Escaped organisms into the environment and disease vectors (resulting in dead wild organisms and huge environmental impact)

- Exogenous material (not natural matter):

- ❖ Chemo-therapeutants and other exogenous molecules (vitamins, antibiotics, pigments, antifouling, pesticides, heavy metals such as copper and zinc)
- ❖ All materials involved in the construction, production and maintenance of the aquaculture/agriculture/fisheries activities

All farming operations produce waste. Intensive (conventional) farming uses significant inputs to maximise production. This can lead to considerable waste releases and to a variety of related environmental problems in soils as well as in surface and ground waters. Farming waste can affect continental and marine aquacultures through water pathways.

Modern intensive aquaculture systems are generally based on open and closed systems. In open systems, the water used as rearing medium is collected from natural sources, and, after use, it is often discarded into the environment along with its accumulated by-products (including high amounts of dissolved inorganic nitrogen and phosphorus). In closed systems, the accumulated by-products (for example, fish faeces, excretions, uneaten feed, etc) must be removed continuously to maintain optimal growth conditions and health of the cultivated species. A variable proportion of the water collected from natural sources is reused after specific treatment(s), which decrease its solid matter, nutrient, and pollutant content, while the remainder is discarded into the environment. Waste solids and dissolved substances management is one of the most important topics in these kind of aquaculture industry today. Extensive aquacultures usually involves the rearing of eggs and larvae in a hatchery before releasing into their natural habitat, either in pens or free to be caught at a later date. The organisms are kept in relatively low densities and very little input is added by the farmer. Extensive aquaculture is similar to free range terrestrial farms. Semi-intensive aquaculture lies in between the two systems.

Increased demand for fishmeal from a growing aquaculture sector has the potential to increase fishing pressure in industrial fisheries. The argument is that increased aquaculture production leads to higher feed demand, and then presumably to higher fishing effort in these fisheries. Collection of seed from the wild (by-catch of non-target species occurring in the collection of wild seed) is also causing concern. On the other hand, the relative waste contribution of aquaculture to marine systems is small but open pens can have local impacts affecting the environmental quality of the surroundings areas and with these, the fish wild stocks.

High concentrations of suspended solids (SS) and dissolved substances present in the effluents from agriculture and aquaculture activities are responsible for eutrophication, that is, the process of natural or anthropogenic enrichment of an ecosystem with inorganic nutrient elements (typically compounds containing nitrogen or phosphorus). The term is however often used to mean the resultant increase in the ecosystem's primary productivity - in other words excessive plant growth and decay - and even further impacts, including lack of oxygen and severe reductions in water quality and in fish and other animal populations. Thus, eutrophication can cause, among others effects, the proliferation of harmful (and toxic) algal blooms in inland and coastal waters. This is one example of the potential for negative impacts on the environment caused by discharging wastewater, which has been enriched with nitrogen and phosphorus, into streams, rivers, lakes, the sea or the soil.

A significant difference also exists between developing and developed countries as the legal, conceptual, economic and societal aspects are very different. Where possible, we will focus on case studies from India.

A common plan was chosen for the FFA industries to strengthen their similarities. In a first part, wastes and their impacts on the environment will be categorised. In a second part, solutions to minimize the production of wastes will be listed. Finally, solutions to reduce waste once produced will be presented. In all cases, particular attention was paid to intensive or semi-intensive systems as they are responsible for most of the wastes produced in farming and aquaculture activities and also because intensive systems are the only ones that will support a global growing demand for organic feed in the future. It is worth to mention that fisheries (exploitation of the natural production) have different mode of production than farming or aquaculture, even if similarities could be found in the way wastes could be managed. Therefore, this review is based mainly on farming and aquaculture way of production with mention of fisheries where possible. Another ultimate part will concern tools, methods that are currently used or about to be developed in order to achieve mitigation measures.

I. Evaluating and measuring the impact on the environment

I.1. Agriculture

I.1.1 Introduction

The turnover of nutrients and other materials within agricultural systems is much greater than in natural ecosystems, and the potential for loss of environmentally active agents becomes correspondingly greater. In most instances, agriculture acts as a "diffuse" source, in other words, relatively low rates of loss or emissions take place from large areas of land into waters or the atmosphere. On occasion, agriculture may also act as a "mini" point source where high rates of loss occur from relatively small areas, for example, from spillages or seepages from stored materials (De Clercq *et al.*, 2001).

a) The nitrogen cycle

The nitrogen (N) cycle is the biogeochemical cycle that describes the transformations of nitrogen and nitrogen-containing compounds in nature (Nitrogen cycle, 2007). The major source of nitrogen is air, which is about 78 percent N_2 by volume. N is essential for many biological processes; for example, it is included in all amino acids, is incorporated into proteins and is present in the four bases that make up nucleic acids, such as DNA. Processing is necessary to convert gaseous N into forms useable by living organisms. All N obtained by animals can be traced to the eating of plants at some stage down the food chain. Plants get N from the soil by absorption at their roots in the form of either nitrate ions (NO_3^-) or ammonia (NH_3). NH_3 is produced in the soil by N-fixing organisms such as *Azobacter vinelandii* which produces the enzyme nitrogenase. Nitrogenase combines gaseous nitrogen with hydrogen to produce NH_3. Some N-fixing bacteria, such as *Rhizobium*, live in root nodules of leguminous plants (such as peas or beans). Here they form a symbiotic relationship with the plant, producing NH_3 in exchange for supplies of carbohydrate. Low nutrient containing soils can be planted with leguminous plants to enrich them with N.

Another source of ammonia is the decomposition of dead organic matter by saprophytic bacteria called decomposers, which produce ammonium ions (NH_4^+). In well-oxygenated soil, NH_4^+ is oxidized to nitrite (NO_2^-) by bacteria such as *Nitrosomonas europea* and then into nitrate (NO_3^-) by *Nitrobacter*. This conversion of ammonia into nitrates is called nitrification.

NH_4^+ can bind to soils and clays, whereas NO_3^-, due to their negative charge, cannot. They are also very soluble in water, and, after heavy rain, leaching (the removal of NO_3^- into rivers and lakes) can occur. In the presence of anoxic (low oxygen) conditions in soils, denitrification by bacteria such as *Thiobacillus denitrificans* can happen. This is the reverse of nitrification and results in NO_3^- being converted to nitrogen gas (N_2) and lost to the atmosphere.

There are three major ways to convert N_2 into a chemically more reactive species:

1. Biological fixation: some bacteria (associated with certain plants, leguminosae) and certain blue-green algae are able to fix N and assimilate it as organic nitrogen.
2. Technical N-fixation: in the Haber-Bosch process, N_2 is converted together with hydrogen gas (H_2) into NH_3.
3. Combustion of gasoline and fossil fuel (automobile engines and thermal power plants), which transfers elemental N_2 into nitrous oxide (NO).

Additionally, the formation of NO from N_2 and O_2 due to photons and lightning, are important for atmospheric chemistry, but not for terrestrial or aquatic N turnover.

b) The phosphorus cycle

The quantities and forms of phosphorus (P) in soils depend on the degree of weathering, the nature of soil parent materials and their management (Carton and Jarvis, 2001). P exists in both inorganic and organic forms in soils. Both organic and inorganic P are involved in transformations that release water soluble P from solid forms (and vice versa). The direction (whether mineralization or immobilisation) and magnitude of P transformations determine the physical and chemical status of P in the soil, and in turn, the potential of the soil system to supply P to plants or to contribute to pollution. A key difference between N and P in soil is the fact that P attaches strongly to the soil matrix and because of that, P does not generally leach in large quantities through the soil profile (except from organic soils such as peats or very sandy soils).

c) Environmental effects of nitrogen and phosphorus from agriculture

In this report, we focus on the effects of chemicals, mainly N and P, which are used as nutrients in the agricultural production systems.

It is typical for both intensive and extensive agriculture and horticulture that nutrients, which enter the system by mineral and organic fertilizers, are not completely used by crops and livestock. Therefore, parts of those nutrients are lost to the environment. Over the past 10 years there has been an increasing awareness about non-point source pollution and the linkage to agriculture. Diffuse pollution of water is one of the major concerns, both to groundwater and surface waters, which cause eutrophication. The definition of eutrophication means an increase in chemical nutrients - typically compounds containing N or P - in an ecosystem. Pollution of air through ammonia and NO emissions, dust and pesticides also give rise for concern (Factsheet, 2007).

From agriculture, part of N loss (50-80%) is recycled to water and soils, causing enrichment of groundwater, eutrophication of surface waters, in synergy with P, and contributing to “acid rain” damages on terrestrial flora and soils. Another part, up to 20-50%, is “denitrified into inert N_2 (and some N_2O, a greenhouse gas) by soil and sediment bacteria, or by natural chemical reduction in certain types of soils and groundwaters (EC, 2002).

Mineral fertilizers, containing N, directly introduce NH_4^+ and NO_3^- into groundwater by leaching and into surface waters by run-off and subsoil “drainage”. The extent of this depends on ground conditions at time of spreading. Organic N (in manure) uses the same “pathways”, plus additional losses to the atmosphere in the form of ammonia (volatilization) and N_2O (incomplete denitrification). Ammonia and N_2O losses ranges from 10% to 30% of the initial N excreted by animals, and are re-deposited on the soil and water bodies in rain (wet deposition) or directly (dry atmospheric deposition) (EC, 2002).

d) Situation in Europe

It is noted that the nitrogen “pressure” on EU 15 agricultural soils from animal husbandry (mainly cows, pigs, poultry and sheep) is estimated at approximately 7.6 million tons N annually spread on agricultural soils (COM (2007) 0120 final, 2007). Therefore, the total diffuse N “pressure”, when the additional 8.9 million tons N from mineral fertilizers are added, was approximately 16.5 million tons in 2003, compared to almost 18 million tons in 1999 and 17.4 million tons in 1995. Regionalised estimates of the application rate of N from manure shows amounts exceeding 170 $kg.ha^{-1}.yr^{-1}$ in Belgium (Flanders) and the Netherlands, but, also, at local level, in Italy, France (Brittany), Spain and Portugal (Fig. 1).

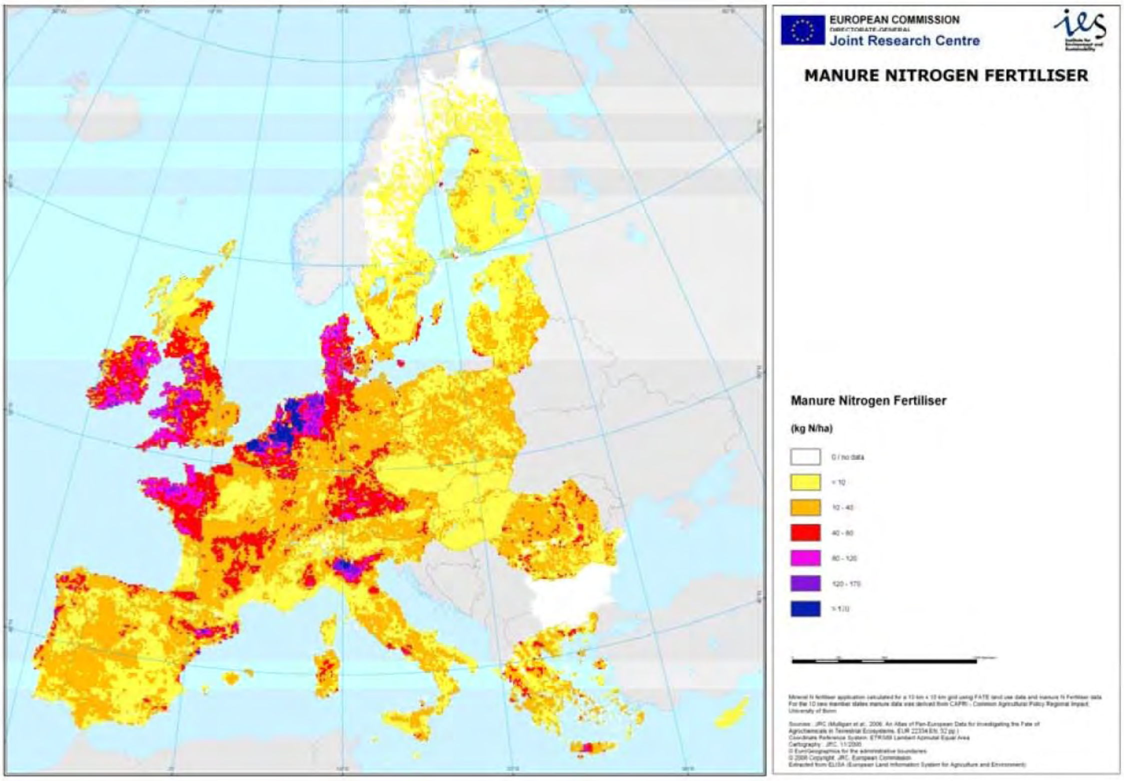


Figure 1: Manure N fertilizer application in 2000 (Mulligan *et al.*, 2006; COM (2007) 0120 final, 2007).

Manure N application rates at regional level between 120 and 170 kg.ha^{-1} are also found in Denmark, United Kingdom (England), a few counties of Ireland and in Southern Germany. All the mentioned areas above also have the highest P application rates from livestock manure (above 90 kg P.ha^{-1}.yr^{-1} for the most intensive areas) and total N and P application rates (manure plus chemical fertilizers) with values exceeding respectively 240 kg N.ha^{-1}.yr^{-1} and 90 kg P.ha^{-1}.yr^{-1} (Fig. 2).

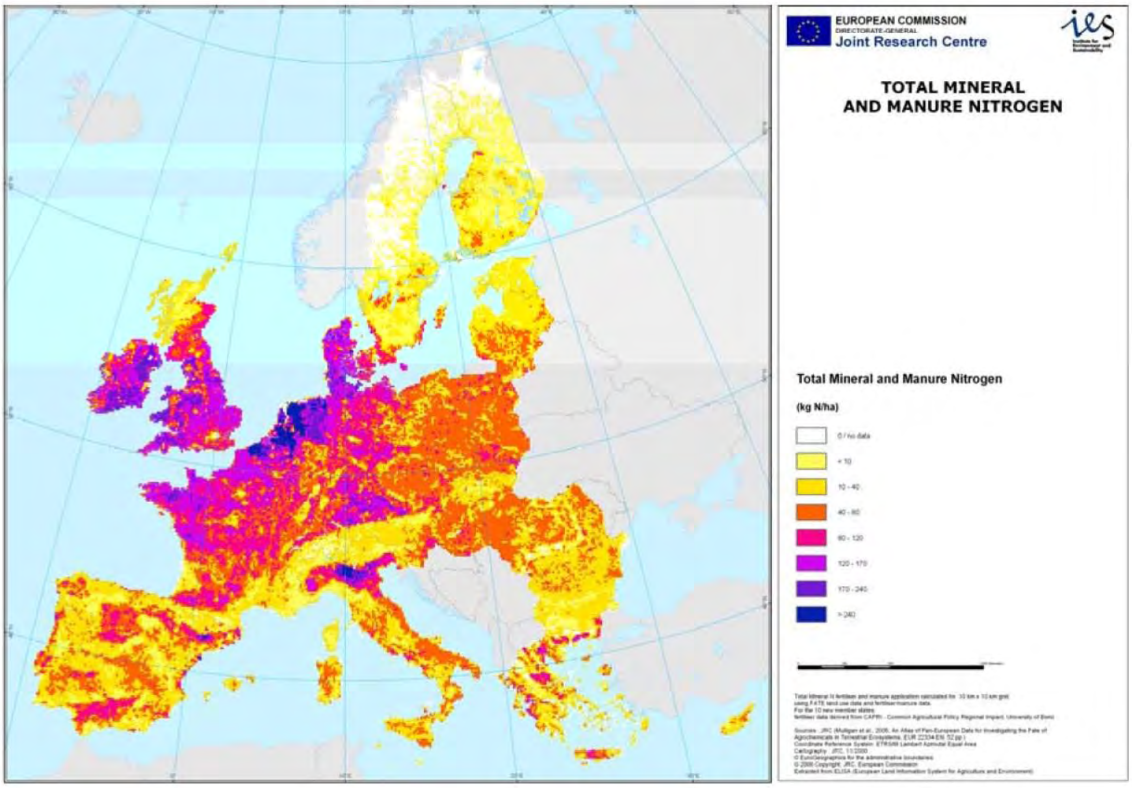


Figure 2: Total N application, manure and chemical fertilizer in 2000 (Mulligan *et al.*, 2006; COM (2007) 0120 final, 2007).

Agriculture is a significant N contributor to the aquatic environment. According to recent studies (EEA, 2005a,b; Mulligan *et al.*, 2006), updating information on the contribution from the different sectors to water pollution, agriculture is typically responsible of 50-80% of the total load. The relevance of N discharge from agriculture into the natural environment has been confirmed by the data reported by several Member States (Belgium, Germany, Denmark, Finland, France, Luxembourg, the Netherlands and United Kingdom) in their reports on implementation of the Nitrate Directive (91/676/EEC). Agriculture represents approximately 62% of N load to surface water, ranging from a minimum load of 18% from the agriculture in Portugal to a maximum load of 97% from agriculture in Denmark. Higher proportions are found for Member States that have established efficient urban wastewater and industrial wastewater treatment systems, thereby drastically reducing N loads from those point load sources.

The contribution from agriculture to N and P losses to water is also confirmed by reports under the Water Framework Directive (2000/60/EC). In 2005, several Member States dentified eutrophication and related contributions from agricultural sources as among the major threats to the achievement of good water status (COM (2007) 0120 final, 2007). A comprehensive analysis regarding "Nitrate, agriculture and the environment" can be found in a book edited by Addiscott (2005).

An indicator of N pressures from agricultural sources is the "gross nutrient balance", which represents the difference between N inputs (from mineral fertilizers, manure, atmospheric depositions, fixation by leguminous crops and other minor sources) and N outputs (uptake by crops, grassland and fodder crops) per hectare of utilised agricultural land. According to the European Environmental Agency calculations, the gross N balance at EU 15 level in 2000 was 55 $kg.ha^{-1}$, a decline of 16% compared to 1990, with a range from 37 $kg.ha^{-1}$ (Italy) to 226 $kg.ha^{-1}$ (the Netherlands). Gross N balance surplus decreased in all Member States except Ireland and Spain (EEA, 2005a). Relatively small surpluses in N gross balance at national level underestimate surpluses in specific regions. An estimate of gross N balance calculated at regional level by the CAPRI database with reference to year 2001 (Britz, 2005) shows the heterogeneity between EU regions, with surplus ranging from 0 up to 300 kg $N.ha^{-1}$. The maximum was reached in areas with a high density of livestock rearing but also in regions of intensive fruit and vegetable cropping, or cereals and maize with unbalanced fertilisation. The highest national N surpluses are found in regions of the Netherlands and Belgium (> 150-200 kg $N.ha^{-1}$). The same levels of surplus, however, can be found in Brittany (France) and in Vechta Kloppenburg (Lower Saxony, Germany). Surpluses of the order of 100-150 kg $N.ha^{-1}.yr^{-1}$ are also found in Member States with relatively low national surplus such as Spain (Catalonia), Italy (Lombardi) and United Kingdom (Northern Ireland, Wales and West England). Greater livestock density leading to increased animal housing, manure storage and spreading, has resulted in more ammonia volatilisation and atmospheric deposition on neighbouring soils and waters with values up to 50-60 kg $N.ha^{-1}.yr^{-1}$ being recorded in such regions.

The implementation of element balances as tools to meet environmental targets for nutrient management in agriculture assumes that there is a relation – direct or indirect – between nutrient surpluses and their environmental impact. There are however, both theoretical reasons and practical examples to suggest that the relation is not so simple (Öborn *et al.*, 2003).

I.1.2 Leakage of plant nutrients (N and P) to water

a) Nitrogen

N leaching (NO_3^-) from land to groundwater and surface waters normally occurs every year in regions with a humid climate (Johnsson and Hoffmann, 1998). The extent to which N leaching takes place depends on both natural prerequisites such as precipitation, temperature and soil type, and on human activities. N losses from growing forests and permanent grassland are often small. However, when the soil is fertilized and cultivated, N leaching can reach considerable levels, especially in intensively cultivated areas with high precipitation and light textured soils.

The general opinion is that leaching of nitrogen from arable land increased considerably in the decades of the post-war period. Some researchers have tried to quantify the increase. Boers *et al*. (1997) estimated an increase of 300-400% between 1950 and the 1980's for the Netherlands. Shröder (1985) estimated an average increase of slightly more than 100% from 1950 to 1980 for Danish agriculture. The average increase for Sweden was calculated to be about 60% from the 1930's to the 1970's (Hoffmann, 1999). Since the mid-19th C., the nitrogen inputs to the Baltic Sea have increased four-fold (Larsson *et al*., 1985).

Hoffmann (1999) also calculated the leaching of N from Swedish agriculture in a historical perspective, where year 1865 was set to be the start of the study. Results indicated that average leaching of NO_3^- in the mid-18th C. was approximately the same as it was at the 1990's. The three main explanations for this were: 1. Large areas of bare fallow typical for the farming practice in the mid-18th C. 2. Enhanced N mineralisation and concentration of NO_3^- in soils at the mid-18th C. from newly cultivated land. 3. Low yield, *that is*, poor N utilisation during the mid-18th C. From 1865 to 1935 the acreage of arable land increased by 1.3 million hectares in Sweden. This was possibly due to extensive draining projects of old grasslands on waterlogged land, lakes, bogs and forested land. This caused a substantial drop in N retention and the probable increase in net load to the sea might thus have been more affected by this decrease in retention than the actual increase in gross load (Hoffmann, 1999).

For the period 1985-1997 several countries in Europe have calculated theoretical decreases in N leaching from arable land. Sweden had a decrease of 25% (Johnsson and Hoffmann, 1998). Denmark had a decrease up to 35% (Andersen *et al.*, 1998). Norway had a decrease of 19% (Vagstad, 1996) and Germany a decrease between 18 and 26% (Werner and Wodsak, 1995; Kersebaum *et al.*, 1996). On the other hand, the Netherlands calculated an increase in N leaching of 38% (Oenemna and Roest, 1998). As with all large-scale estimates, the values are somewhat uncertain but they do, nevertheless, indicate a trend towards less N leaching (Hoffmann, 1999). A rapid large-scale reduction of N leaching from arable land is possible if drastic changes in land use occur or if fertilizer and manure rates were reduced from superoptimal rates. However, when fertilizer N is applied in rates adaptable to crop needs (Macdonald *et al.*, 1989), the N leaching rates will not be rapidly reduced by limiting the use of fertilizer N in the short term (Addiscott, 1988).

Most arable soils have been fertilised for decades with different amounts of manure and fertilizer N, which influence the size and mineralisation capacity of the N pool in soil. In a long-term winter wheat experiment at Rothamsted (United Kingdom), soil delivery of plant available N (which can be leached when no plant can utilize this amount) increased as a result of mineral N fertilizers applied historically over a long period (Shen *et al.*, 1989). However, input of fertilizer N is not necessarily a crucial factor for N leaching in the short term, providing that it is not used at a superoptimal rate. It is the N mineralisation from the N pool in the soil that is important for N leaching, which was demonstrated by Lawes as long ago as 1881 (Lawes *et al*., 1881). In a long-term perspective, the agricultural system influence the organic matter content including the carbon (C) and N contents in soil. The organic matter content in soil will, after a long time, reach a steady state, depending on amounts of applied C and N and removed C and N in the crop rotation. In a long-term 6-year crop rotation with 3 years of clover/grass ley and 3 years of cereals, the C contents in soil increased from 3% to 3.5% over a 30-year period. The high C content at start depended on a permanent grassland as preceding crop. A 6-year crop rotation with only cereals after the permanent grassland decreased the C contents in soil from 3% to 2% during 30 years (Johnston, 1994). If organic N and thereby the N mineralisation capacity have increased in arable soils, reducing fertilizer N rates from suboptimal or normal levels will only result in a very slow decrease in N leaching due to inertia in the system (Löfgren and Olsson, 1990).

b) Phosphorus

P is one of the essential elements for plant, livestock and humans. P, together with N, is also often the limiting nutrient for primary production in lakes and streams. This means that an increased P level in a lake increases primary production and oxygen demand and the lake becomes more eutrophic (Correll, 1998). The anthropogenic input of P to freshwater was 3 000 tons.yr^{-1} in Sweden, where most of the eutrophic lakes were situated in southern and central parts (Larsson, 1997). However, P leaching losses also reach the sea. Since the mid-19th C., the P inputs to the Baltic Sea have increased eight-fold (Larsson *et al.*, 1985).

Sweden, as well as many other countries, experienced a major increase in fertilisation of arable land in the middle of the 20th C. The high inputs of fertilizer P caused a build-up of P in the soil. The average P accumulation in Swedish arable soils since the 1950s has been estimated at 600-700 kg $P.ha^{-1}$ in the topsoil (Andersson *et al.*, 1998). Although fertilizer loads in total have been considerably reduced in Sweden during the last 20 years, the application is not evenly distributed between fields (SCB, 2006). Even with reduced P application, some soils still have a surplus of P that may cause environmental effects over many years. The average losses of P from cultivated fields, 0.3 kg $P.ha^{-1}.yr^{-1}$ (Ulén *et al.*, 2001) are very low in proportion to P application and P uptake by the crops grown but it is enough to cause increased eutrophication in many rivers and lakes in Sweden. Ulén *et al.* (2001) found in 15 agricultural fields that despite reduction in P inputs and finally a negative P balance in most of the fields, P losses was more or less constant over a period of 21 years. The authors also found that two of the 15 fields contributed about half of the total P loss from all fields.

The main transport pathways for P removal from agricultural soils are surface run-off (overland flow), interflow (lateral flow below the soil surface), matrix flow and preferential flow (Haygarth and Sharpley, 2000). Matrix and preferential flow pathways refer to vertical water and solute transport. The evidence of P losses via surface run-off and interflow is well documented (Svendsen and Kronvang, 1991; Sharpley *et al*., 1994; Catt *et al.*, 1998). Both dissolved and particulate P are transported via these pathways and their distribution depends on the present land use and management of the upstream fields. In general, particulate P dominates in surface run-off from cultivated fields, whereas the overland flow from grass- and forestland mobilizes little sediment and, therefore, the particulate P contribution to the total P losses decreases. Surface run-off driven P losses are closely connected with the erosion issue, and erosion countermeasures are also often effective for P loss reduction. The amount of transported P depends on soil properties (infiltration capacity, soil P content, soil texture and erodibility), site characteristics (slope, vegetation cover and roughness coefficient) and precipitation duration and intensity.

P export via matrix and preferential flow pathways has gained increased attention during the last years (Djodjic, 2001). Matrix flow stands for uniform water movement through the whole pore volume of a soil profile, whereas preferential flow of water occurs along preferential pathways such as macropores, cracks and wormholes (Beven and Germann, 1982), where only a small part of the total soil pore volume is involved in water and solute displacement. P concentrations in percolating water are assumed to be small, due to the high sorption potential of P-deficient subsoils. However, some studies have shown other results. Sharpley and Rekolainen (1997) identified several situations where soils are more vulnerable to P export through the soil profile: (1) peaty soils contening organic acids, and sandy soils with a low P sorption capacity, (2) waterlogged soils where the conversion of Fe(III) to Fe(II) lead to P release or organic P mineralization, and (3) soils with preferential flow through macropores and earthworm holes. Even in the case of flow along preferential pathways, the distribution between dissolved P forms and particulate P forms varies. Simard *et al.* (2000) concluded that P concentrations and forms in drainage water from agricultural fields were influenced by soil texture, tillage intensity and frequency, cropping systems and seasonal variability.

c) Potential of N and P losses in different management systems

Low input grazing-based meat production is often practised on land where alternative land use is restricted and where the grazing may contribute to the maintenance of high species diversity of the wild flora and fauna (Gimmingham 1972; Grime 1979; Bakker 1989; Vestergaard Petersen 1996; Johansson and Norderhaug 1998). In most areas of Europe seminatural grasslands are not considered to be very economically productive, but are retained in countries such as the UK, the Netherlands and Sweden because farmers receive subsidies from the European Union to maintain these systems (Dahlin *et al.*, 2005). Although inputs through atmospheric deposition may be considerable in certain regions, external nutrient inputs into low input systems are generally small, as area stocking rates and productivity are low. For example, whereas 200-400 kg $N.ha^{-1}$ may be added annually in higher input grazing-based meat or milk production, N inflow into low input systems is often only one-tenth of that. Nevertheless, in such systems considerable nutrient losses may occur, although not to the same extent as higher input production systems (Dahlin *et al.*, 2005). However, there are several possible management practices that can improve nutrient utilization and reduce losses in low input grazing systems. Limiting access for the animals to vulnerable areas (including fencing-off water courses) or areas with a high potential for N leaching during the wet season may thus limit losses or divert losses to less vulnerable areas. Housing of animals in winter often reduces nitrate losses, but simultaneously increases total ammonia losses (de Klein and Ledgard 2001). A rotational grazing favour an even distribution of excreta and urine, which decrease the risk of harmful plant nutrient hot spots and nutrient losses. Rotational grazing also makes it possible with higher stocking rates, compared with continuous grazing (Chen *et al.*, 2001). Where grasslands are not being managed with nature conservation as a specific goal, they may be reseeded to improve their nutritive value. As reseeding produces a flush of mineral N and soluble organic N in the soil (Bhogal *et al.*, 2000), spring reseeding is prefered, as this has been found to minimize N leaching (Shepherd *et al.*, 2001).

In Europe, there is an increased specialization in agriculture at the expense of the more "traditional", mixed farming systems. The European Union (EU) defines mixed crop and livestock farming systems as those that have less than two-thirds of their total income related to one sector of production (EEC, 1985). The widespread availability of mineral fertilizers has also encouraged the development of specialist arable farms. One important aspect of this specialization has been increased nutrient capital of agricultural land resulting in enhanced nutrient cycling and losses (Kleinhanss *et al.*, 1997; Magdoff *et al.*, 1997; Oomen *et al.*, 1998). Another consequence of the specialization of farms has been the decoupling of on-farm nutrient flow (field – harvested feed and forage – animal manure – field) that are commonly associated with mixed systems, owing to a reduced reliance on manure as a nutrient source for maintaining soil fertility in arable areas. The intensification that has occured within many livestock enterprises is only possible through the importation of feed, which brings associated problems of utilization and recycling of nutrients contained in the resulting waste materials (Watson *et al.*, 2005). Mixed farming systems includes a broad range of farm types. At one end of the scale, in systems largely based on livestock production, the main inflow of N and P to the farm is through purchased feed and livestock while the main outflow is through livestock produced and gaseous losses (N). At the other end of the scale, in systems largely based on crop production and export, the main inflow of N and P is through purchased fertilizers and seeds while the main outflow is through crop produced. The nutrient leaching and run-off can be very large in such a kind of crop production systems (Oomen *et al.*, 1998; Watson *et al.*, 2005).

In mixed farming systems it is not only the management practices of the external inputs and outputs of N and P that have to be improved to increase nutrient use efficiency (NUE) and reduce nutrient losses. Also, the internal flows of N and P within the farming system are highly important. Restricting application rates alone, according to the Nitrate Directive (91/676/EEC), will not automatically improve NUE and thus reduce losses. One example is that the balance between the use of imported versus home produced feed has a significant influence on the nutrient cycling within the farm. One of the challenges for improving NUE on mixed farms is reducing the uncertainty of nutritive value associated with the use of home produced feed and forage, and that of its impact on manure production and composition. This includes choices related to housing, expecially bedding, and manure management (Watson *et al.*, 2005).

I.1.3 Ammonia emissions from stable, manure storage and spreading of fertilizers

International agreements, like the Gothenburg protocol (UNECE, 2004), are aiming to reduce global environmental problems such as eutrophication and acidification by reducing emissions of pollutants to air. Ammonia is such a pollutant, which mainly originates from manure handling in stables, storages and applied on land. In Sweden 2001, 53,800 tonnes ammonia were released to air (SCB, 2003), where 85% originated from agriculture. The same situation is prevailing in the Netherlands with 90% (van der Hoek, 2007) of the ammonia originating from agriculture and the agriculture commonly account for more than 80% of the total NH_3 emissions in the European countries (EMEP, 2005). For the European Union Member States, the EU Directive 2001/81/EC sets upper limits for each Member State for the four components responsible for acidification, eutrophication and ground-level ozone. This is the so-called National Emission Ceilings Directive (NEC; EC, 2001). In addition to adverse environmental effects, ammonia volatilization means substantial losses of fertilizer value.

Ammonia forms salts with acidic gases in the atmosphere (Ferm, 1998). These salt particles can be transported long distances and deposed to ground and water. Modelled origin of deposited nitrogen in Europe during 1993, showed that about 15% of the NHx emissions were deposed to the seas (Barett *et al.*, 1995). About half of the ammonia is deposed locally (within 50 km from the source) and the rest is transported longer distances with an average half -life corresponding to ca. 400 km at the present pollutant level (Ferm, 1998).

The main sources of ammonia emissions are manure storage and spreading of manure in most European countries. The goal for Sweden is 51,700 tonnes ammonia emissions per year by 2010, which means at least a reduction of 15% from the 1995 levels. Under the Gothenburg protocol, signatories will have to report NH_3 emissions annually. In the national NH_3 emissions inventories, emission factors are used for stable, storage and spreading. In the European Agricultural Gaseous Emissions Inventory Researchers Network (EAGER) (Menzi *et al.*, 2004), national emission factors, inventories, and models have been compared. The comparison of emission factors proved that they mostly agree quite well between countries and existing differences are often explicable by farming practice. The biggest uncertainties exist for solid manure (Menzi *et al.*, 2004; Reidy *et al.*, 2007). In slurry systems, NH_3 losses are highest in stable and after spreading while in solid manure systems the highest losses occur during storage (Reidy *et al.*, 2007). The emission factors for storage and spreading of manure for Sweden are presented by Karlsson and Rodhe (2002).

The ammonia volatilisation is influenced by many factors like climate conditions (temperature, wind speed), manure properties (pH, viscosity, ammonium (NH_4-N) concentration), and handling technology (Svensson, 1991; Sommer *et al.*, 2001).

a) Stable

In stables, the most important factors are number of animals and animal size, manure properties like pH, temperature, dry matter content and carbon:nitrogen ratio (C:N), size of emitting surfaces with manure, storage time and used ventilation technology (Sannö *et al.*, 2003). Swensson and Gustafsson (2002) showed that the NH_3 release is higher from a lose house system with slurry handling than a tight house system for cows with solid manure handling partly due to larger surfaces with slurry in the lose house system. There was also a clear effect of the content of crude protein, in the total feed ration on ammonia release in tied stall barn with liquid manure (Swensson, 2002). For solid manure, the choice of bedding material could influence the ammonia losses, both in stable and in storage (Jeppsson, 1999).

b) Storage

The main influencing factors on the ammonia losses from storages are manure properties (pH, dry matter content) temperature and wind conditions, filling technology, storage time, and for slurry storage ratio surface:volume, crust formation and mixing methodology (Svensson, 1991).

There may be large losses of ammonia when manure is stored, especially from solid manure or uncovered storages with urine water (Sommer *et al.*, 1993; Karlsson, 1996a; Karlsson and Rodhe, 2002; Smith *et al.*, 2007). The Swedish emission factors for storage of manure from dairy cows and pigs are: 1) 20% of total N if stored as solid manure 2) 1 to 9% of total N if stored as slurry and 3) 5 to 40% of total N if it is urine (Karlsson and Rodhe, 2002). If the manure is composted, the emission factors are also high (30%).

c) Spreading

Variables significantly affecting NH_3 volatilization after spreading of slurry are soil water content, air temperature, wind speed, slurry type, dry matter content of slurry, total ammoniacal nitrogen content of slurry (TAN=NH_3+NH_4^+), application method and rate, slurry incorporation and measuring technique (Sommer *et al.*, 2001; Søgaard *et al.*, 2002). In laboratory experiments, Svensson (1993) showed an exponential increase of NH_3 equilibrium concentration above soil after broadcast spreading of pig slurry with increased soil surface temperature. The NH_3 emission was proportional to the applied amount of manure TAN, as with the double rate of TAN, NH_3 equilibrium concentration was doubled. Additionally, Svensson (1993) could show that the dry matter content of the manure has a big influence on NH_3 emissions from manure applied on a soil surface.

Emission factors for NH_3 losses from field applied manure are related to time of year when the spreading take place, type of manure (slurry, solid manure, urine) and spreading technique with or without incorporation in the Swedish inventories (Karlsson and Rodhe, 2002). Losses vary between 3 to 90% of the NH_4^+-N applied with manure. The lowest default value 3% are valid for band spread slurry in the late autumn, immediately incorporated and the highest 90% are for broad cast spread manure on leys in the summer time. Accordingly, hard soil and/or grass sward prevent the contact between manure and soil. On the contrary, high infiltration rate of the slurry and a porous soil lead to fast infiltration of the slurry into the soil and consequently low NH_3 losses (Rodhe *et al*., 2004; Rodhe and Etana, 2005). Also, placing the slurry in the bottom of vegetation, like winter wheat with higher heights than 20 to 30 cm, results in lower NH_3 emissions than spreading it in vegetation lower than 10 cm on a bare soil surface (Sommer *et al.*, 1997).

I.1.4 Other wastes and their effects on the environment

a) Trace elements

In many agricultural systems, and especially in livestock farms, there is a net input of trace elements because of relatively large inputs through atmospheric deposition and purchased feed, to which Cu, Zn and other essential elements are added (Öborn, 2001; Gustafson *et al.*, 2003). Trace metal contaminants in fertilizers and lime also contribute to the net input. Entirely arable systems, however, can show a net depletion of certain micronutrients (Öborn *et al.*, 2003).

Long-term accumulation of trace elements, such as Cd, Cu and Zn, can affect soil fertility and product quality and promote trace element leaching from arable land. Water run-off from arable soils could be a potential source of trace element pollutants to groundwater and surface water (Römkens *et al.*, 2002). In the past, trace element inputs via atmospheric depostion and mineral fertilizer (mainly Cd) caused a net accumulation in arable soils and crops (Andersson, 1992). Inputs in atmospheric deposition and mineral fertilizers are currently declining and instead it is possible that inputs of trace elements in purchased feed to livestock may now cause significant accumulation in the arable soils via manure application (Moolenaar and Lexmond, 1998). Bengtsson *et al.* (2003) measured the trace element leaching and surface run-off (Cd, Cu and Zn) for a five-year period at three arable fields representing the soil types Eutric Regosol, Thionic Gleysols and Dystric Cambisols in Northern Sweden. The authors found that the trace element outputs in leached water from the soil was quantitatively significant at field level, compared with other amounts of inputs (fertilizers, manure, deposition, pesticides) and outputs (harvested crop), but also that the variation was large between years and soil types.

b) Pesticides used in agriculture

The objective of the EU thematic strategy on the sustainable use of pesticides (COM (2006) 372 final) is to minimise the risks for health and environment from pesticides. Due to losses during handling and spreading of pesticides in agriculture, pesticides can be found in soil, water and air as well as in plants and organisms in ecosystems. In the EU Drinking Water Directive (98/83/EC) and the new Groundwater Directive (2006/118/EC), the quality standard of groundwater and water used as drinking water has been set to 0.1 $\mu g.L^{-1}$ for individual pesticides and 0.5 $\mu g.L^{-1}$ for the sum of individual pesticides. The Swedish Chemicals Agency has derived water quality standards (guideline values) for approximately 100 different pesticides (Kem, 2004). The guideline values give the maximum concentration of each pesticide estimated not to cause any negative effects on aquatic organisms. Pollutants from the use of pesticides can be found throughout the whole food chain. Among the Swedish human population, food is the major source of exposure to Persistent Organic Pollutants (POPs), such as chlorinated pesticides, polychlorinated biphenyls and DDT-compounds. In a study of mother's milk from primiparous women living in Uppsala county, Sweden, the concentrations of the chlorinated pesticides have decreased from 1996 to 2006. This is probably a consequence of reduced levels of many POPs in the environment and in food since the 1970's (Glynn *et al*., 2007).

Pesticides have been found on numerous occasions in Swedish waters during the 20-year period 1985-2005. Overall, the total incidence of concentrations above 0.5 $\mu g.L^{-1}$ has decreased. For surface water it is possible to discern a substantial reduction in pesticide concentrations, and also in the number of sites with high total detections of pesticides. There have been a number of national efforts made to reduce the levels of harmful substances in Swedish waters and it is likely that these have had an impact as regards surface waters. However, a similar improvement has not occurred in groundwater. The increase in the number of sites investigated makes it difficult to interpret the reduction in detections exceeding the water quality criteria. However, the lack of decrease in detection levels of the most commonly occurring substances in groundwater may indicate that the efforts made to date are insufficient. The results of 20 years of monitoring demonstrated clearly an overall longer time interval between the onset of risk mitigation measures (including banning of certain compounds) and a reduction in findings of pesticides in groundwater (Törnquist *et al.*, 2007). Concentrations of pesticides in water can influence aquatic organisms. One example is an experiment with brown trout (*Salmo trutta* L.) exposed to the pyrethroid pesticide cypermethrin, a disrupter of olfactory receptor function (Jaensson *et al*., 2007). Parr exposed to cypermethrin had significantly lower blood plasma levels than control males. Also, parr exposed to cypermethrin displayed fewer courting events, spent less time near the nesting females and had lower volumes of strippable milt.

c) Organic load (BOD) in waste and wastewater from intensive livestock operations

The water pollution potential of waste and wastewater from intensive livestock operations can be substantial. The major pollutant categories are Biological Oxygen Demand (BOD), N and phosphate. In summary, all these waste categories from agriculture are higher than the same contributions from the total resident population (some examples can be found from Portugal and Israel; National Strategy for Animal Production and Agrofood Effluents, 2007). The table below gives a summary on the water pollution potential of waste categories in a typical Mediterranean country (Table 1).

Table 1: Water pollution potential of waste categories from different origins in an average meditteranean country (National Strategy for Animal Production and Agrofood Effluents, 2007)

Sector	Inhabitant equivalent		
	(BOD_5)	(Nitrogen)	(Phosphorous)
Beef cattle	6,151,000	9,041,000	2,123,000
Piggery	3,167,000	3,327,000	4,315,000
Aviculture	2,000,000	3,157,000	3,680,000
Slaughter Houses	45,000	25,000	20,000
Olive oil	251,000	39,000	34,000
Milk industry	274,000	49,000	14,000
Wine	452,000	55,000	19,000
TOTAL	12,340,000	15,693,000	10,185,000

A similar picture has been achieved in Israel, where waste from livestock farming alone contributes 70% of the pollution potential of the total resident population. The FAO document “Livestock’s long Shadow” (FAO, 2006) gives a rather clear review on global water consumption and pollution by Livestock.

Of great significance, even though of lesser volume, are residues from wine and olive industries, as those wastes contain recalcitrant organic chemicals like phenols, tannins or terpenes, that are highly toxic and can interfere significantly with efforts of biological water purification approaches. In the often-small streams, characteristic of the Mediterranean landscape, release of such waste material can cause irreversible damage.

d) Other waste

A report from the working group in the RAMIRAN (Research Network on Recycling of Agricultural and Industrial Residues in Agriculture, since 1976) on other wastes than manure generated into the farm (Balsari, 2002) identified packages for crop herbicides and mineral fertilizer wastes produced for animal health treatment and used motor oil production. Balsari concluded that neither the national laws nor the Waste Framework European Directive (75/442/EEC) did cover completely the “other wastes” managements. At the time of the Directive, the most common waste disposal method was burning of both paper and plastic materials. The burning of the wastes is prohibited both in France and Italy and, in these countries, the used oil is supposed to be picked up by official and authorised collecting centres. Only results from Italy, France, Belgium and Great Britain were included in the report.

I.2 Aquaculture

In many areas in the world, aquaculture is a large and growing sector which releases effluents, in varying degrees, to the surrounding environment (Hall *et al.*, 1990a,b; Wu 1995; Basurco and Lovatelli 2003; Karakassis *et al.* 2005; FAO 2006). Major environmental impacts of aquaculture have been associated mainly with high-input high-output intensive systems. Fed aquaculture, such as net cage fish farming, releases nutrients in both particulate and dissolved forms, via the nets, to the surrounding waters and to the underlying sediments, regardless of whether the aquaculture is practiced in freshwater lakes, rivers, estuaries or coastal marine waters (Fig. 2). Similarly, land-based pond or tank aquaculture releases large loads of nutrients in flow-through or partially-closed recirculating production systems. In the case of semi-intensive shellfish culture, the farmed invertebrates also release organically-rich particulate faeces and pseudofaeces and excrete dissolved N and P into their surroundings. Macroalgae also release certain amount of tearing, breakage and “shedding” of thalli which contributes to the loading of particulate organics to the environment around such farms. It is only in the low-yield, extensive integrated systems that include various elements of biofiltration, or in the highly-intensified, high-yield integrated systems that we find minimal release of nutrients to the environment, and to date, these are mainly experimental or pilot-scale systems (Kautsky and Folke 1991; Schneider *et al*. 2005; and others).

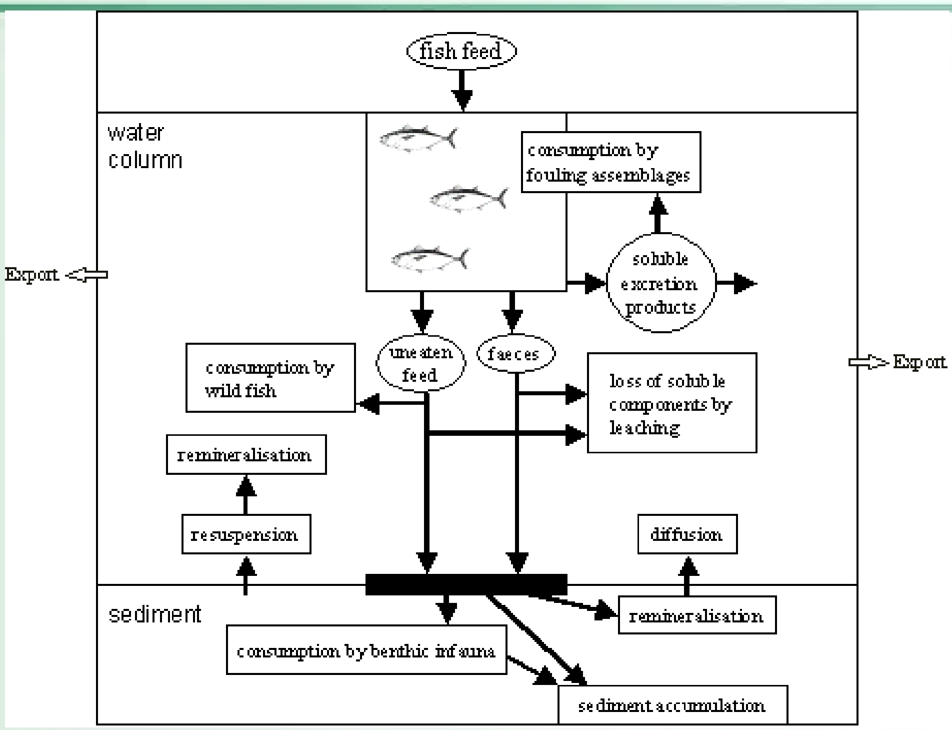


Figure 3: Organic matter pathways in the surrounding environment of fish cage farming (From http://www.sardi.sa.gov.au)

The large quantities of effluent wastes naturally lead to several important questions, including (but not exclusively): which compounds are released (effluent "quality"); what is the flux of material released (that is, effluent "quantity"); how far are the effluents dispersed; are there seasonal variations in the composition and fluxes of the effluents; what are the "near-field" vs "far-field" effects (that is, what are the scales of this problem?); which communities are affected by these effluents and to what extent are they affected; what can be done to eliminate or re-use these nutrients, etc.

I.2.1 Types of wastes

Finfish and shellfish farms discharge soluble wastes into the water column and particulate wastes which settle to the seabed. Different temporal and spatial scales of aquaculture effects have been considered (Silvert, 1992). Accordingly, 3 basic categories have been made: internal impacts (refers to the short-term (minutes to hours) effects of a farm on itself and the immediate surroundings, for example, oxygen depletion), local impacts (may affect neighbouring farms and wild stocks within a distance of ~1 km, for example, organic matter loading) and regional impacts (water body-scale effects with time spans ranging from a tidal cycle to a season or more, for example, eutrophication). The scales mentioned here are strongly dependent on the local settings such as current regime and degree of exposure (Tett, 2008). Further attraction of wild fish and sessile organisms can modify the sedimentation regime through consumption of the large effluent particles (Dempster and Sanchez, 2007). The small-size organic particles (How small is small?) may settle near or far from the farm, depending upon the local hydrodynamic regime (Sara *et al.*, 2006) and a small proportion of this organic material is composed of fine organic particles (for example, dust from fish feeds) that practically remains in suspension (Tsapakis *et al.*, 2006). The soluble wastes consist of organic or inorganic compounds that may be taken up by organisms, undergo chemical interactions in the water column or in association with surfaces, or simply persist in the water column if they are non-reactive (Black 1998).

a) Solid waste

When determining the amount of waste a system will generate, the amount of feed offered is an essential factor. In a properly managed farm rearing carnivorous finfish species (salmon, seabream, etc.), approximately 30% of the feed used will become particulate solid waste (faeces and waste feed; Lupatsch and Kissil, 1998; Roque d’Orbcastel and Blancheton, 2006; Roque d’Orbcastel *et al.*, 2008). The amount of waste is often greatest in the summer months when feeding rates are highest (Krom *et al.*, 1985; Miller and Semmens, 2002). Faecal matter constitutes more or less compact particulate material that may settle in the tank where the fish are held or in a sedimentation tank. The chemical composition (C, N, P and trace elements) and physical characteristics such as size, density, water content and impact resistance of feces depend on feed composition and fish species as well as the phase of development (Aquaetreat, 2007; Roque D’orbcastel and Blancheton, 2006). The N in the faeces generally occurs as peptides, amino acids and other organic degradation products of the ingested proteins. As P is found in all plant and animal feed ingredients, excess dietary P is excreted in both solid and dissolved forms. Nevertheless, a major portion is found in solid form, being more amenable than N to adsorption/sorption onto/on particulate (Holmer *et al.*, 2007). The P excreted with the particulate wastes is mainly indigestible material such as bones and other P-containing compounds originating in the fishmeal. When the particulate effluents settle, organic matter is oxidized (mineralized), consuming oxygen and releasing a variety of compounds, including various forms of N and P.

Water flow patterns in production units are important for waste management because a proper flow will minimize the fragmentation of fish faeces and allow for rapid settling and concentration of the settleable solids. This can be critical because a high percentage of nonfragmented faeces can be quickly captured which will greatly reduce the dissolved organic waste. A reduction in downstream pollution can be achieved by the rapid removal of solids in the settleable form before discharge to public waters. By settling out downstream, solid wastes cover benthic animals and reduce oxygen levels, which reduce the biodiversity of a stream (Miller and Semmens, 2002).

An increase in suspended solids will result in an increase of BOD (the quantity of oxygen utilized in the biochemical oxidation of organic matter). That is why a higher portion of settleable solids, quickly removed, will reduce the dissolved portion of waste from the farm. Generally, the smaller a particle is, the more leaching will take place. The majority (in numbers) of the solids (including surplus feed) produced in aquacultural operations are particles measuring 30 μm or less. Small particles also take longer for settling to occur

In the case of land-based aquaculture, elevated levels of waste are released during the harvest of fish and cleaning of tanks or ponds. In particular, the final 25% that drains from a pond normally contains the majority of the metabolic and pathogenic waste.

Dead animals, parts of dead (including shells, heads, guts, bones, etc.) and living animals (including fish scales and other "solid material"), exoskeletons and shells generated by aquaculture activities are also considered solid waste. In addition, any other wastes or particulate matter generated during normal farm operations, for example, removal of fouling organisms from nets, dropping of equipment, bags, etc. from the fish farm are also included in the broad category of particulate wastes.

b) Dissolved waste

Fish excrete soluble compounds (dissolved wastes) in many forms through the gills, kidneys, and with faeces, depending on the quality of the feed, the fish species and their phase of development (Midlen and Redding, 1998; Miller and Semmens, 2002; Roque D'orbcastel and Blancheton, 2006). Lipids and carbohydrate degradation produce CO_2 (representing about 50% of the carbon intake) and water.

For most fish species, between 50 and 70% of the N intake is excreted (Lupatsch and Kissil, 1998; Islam, 2005; Merino *et al.*, 2007; Roque d'Orbcastel and Blancheton, 2006). The main nitrogenous waste of protein degradation is ammonia, representing 80 to 90% of the soluble N excreted. Ammonia is a gas which dissolves in water to form ammonium ions (NH_4^+), and unionized ammonia (NH_3, concentration vary with the pH, temperature, and salinity) which is harmful or lethal to aquatic organisms. Its most toxic form is in the unionized state, but this form is seldom encountered in net cages and in land-based aquaculture systems. Urea is also a quantitatively important form of N excretion. Nitrate and nitrite excretion are negligible. A large proportion of the excreted ammonium is taken up by phytoplankton, macroalgae and bacteria, and this N then travels up the food web. When conditions are suitable (*that is,* large surface area to allow the growth of autotrophic bacteria), ammonia oxidizing bacteria take up the excreted ammonia and transform it into less toxic forms i.e. nitrite and nitrate (Michaud *et al.*, 2006; Crab *et al.*, 2007; Zamora-Castro *et al.*, 2007). These compounds may then, be utilized by plants and algae for growth.

P found in fish feeds is metabolized and the degradation products of phospholipids and other P-containing compounds are excreted in various dissolved forms. The main soluble P waste is orthophosphate (PO_4^{-3}), representing only about 20% of the fish P intake (Lupatsch and Kissil, 1998; Roque d'Orbcastel and Blancheton, 2006). There is a dogma which is in the process of being revised whereby P is the major limiting nutrient in freshwaters, whereas N is often the limiting nutrient for productivity in marine waters.

Nitrogen and phosphorus release from aquaculture activities may contribute to eutrophication by promoting growth of algae or plants. Watershed resource managers focus on reducing the amounts of phosphorus and nitrogen in a watershed when attempting to improve water quality (Miller and Semmens, 2002).

c) Other types of wastes

There may be other types of wastes from aquaculture in addition to the nutrients mentioned above (Midlen and Redding, 1998; Miller and Semmens, 2002). For example, fish can be considered as a waste following accidental fish escape in the event of broken cages or damaged nets. Reproduction products (gametes) constitute another kind of waste that is difficult to control, especially in the case of net cage rearing or long-line shellfish production (Dimitriou *et al.*, 2007). Additionally, dense stocking rate (overcrowding) may induce stress problems and increase susceptibility to diseases. Overcrowding may lead to poor water quality due to decreased oxygen level, high-accumulated metabolic products and excrement, rapid growth and transmission of noxious parasites, microorganisms and pathogens.

Exogenous materials such as veterinary drugs, antibiotics or anesthetics, may cause a negative environmental impact, although this impact is still not well-known (except concerning the development of resistant microbial communities) or documented (Miller and Semmens, 2002). Utilization of drugs has decreased noticeably during recent years because of increasingly stringent legislation and due to replacement by vaccines, which have little impact on the environment (Sánchez-Mata and Mora, 2000).

Other chemical products, such as cleaning agents and disinfectants, can also be significant sources of pollution; however, the tendency is to drastically reduce the use of all such chemicals (Midlen and Redding, 1998; Miller and Semmens, 2002). Sometimes, antifouling substances are used to decrease seaweed and other biofouling on nets and boats but the use of toxic antifouling products with heavy metals, for example, is now forbidden in many countries (Costello *et al.*, 2001).

I.2.2 Impacts on environment

As already mentioned, effluents released in the environment may be roughly divided into dissolved and particulate matter. This part of the review will then focus on two main impacted environments: pelagic and benthic. On the one hand, dissolved nutrients have direct effects on the water column, influencing pelagic changes. On the other hand, the majority of the literature dealing with environmental impacts of aquaculture has focused on particulate effluents which settle to the seafloor and may cause a variety of effects on benthic ecosystems; the complex of a community of organisms and its environment functioning as an ecological unit.

I.2.2.1 Impact on pelagic environment

a) Effects on water quality

In a comprehensive meta-analysis study, Sarà (2007) reviewed 340 studies to test whether aquaculture facilities had any effect on physical and chemical variables in the surrounding waters. It was surprising to find that dissolved oxygen (DO) concentrations were also unaffected by aquaculture, since most aquaculture systems are largely heterotrophic with high oxygen demand (OD) due to the large biomass of captive animals. On the other hand, water transparency, turbidity, and pH were substantially altered (especially in pond culture) by shrimp and fish farming.

Practically in all studies of the water column around net-cage fish farms, there was either no change or only a localized increase in water column nutrients (for example, phosphate, ammonium) or particulate organic matter. It is noteworthy that there is minimal transformation of ammonium to nitrite or nitrate in this environment. The absence of a "cloud" of nutrients in the water column around fish farms has been attributed to a fast nutrient dilution or dispersal (Pitta *et al*., 2006) and to a rapid transition up the food web to higher trophic levels (beginning with grazing), eventually leading to an increase in the abundance of wild fishes around fish farms (Machias *et al*., 2004, 2005) but also in the overall fisheries production in the area (Machias *et al*., 2006). The pulse of nutrients derived from fish farms could enhance production, but it is unusual to find large populations of bacteria, phytoplankton, cyanobacteria or protista around fish farms that can alter the characteristics of the water column (Pitta *et al*., 1999; Karakassis *et al.*, 2001; Pitta *et al*., 2005; Pitta *et al*., 2006). Indeed, most surveys indicate minor (if any) deterioration of water column characteristics at sites with sufficient water circulation and/or tidal flushing (Nordvarg and Johansson, 2002; Soto and Norambuena, 2004). In contrast, in some closed fish farms several authors reported increased turbidity around fish cages, between others effects, due to a high phytoplankton and bacterial development (Stirling and Dey, 1990; Pitta *et al*., 1999; Karakassis *et al*., 2001; Sakami *et al*., 2003).

In an attempt to quantify the effect of fish farms on Mediterranean sea water quality, Karakassis *et al.* (2005) compared nutrient inputs from this sector to other anthropogenic sources and concluded that at such large spatial (and temporal) scales, total aquaculture contribution may only amount to a 1% increase in nutrient pools, as compared to other human activities that might double or triple the nutrient pool.

It is possible to accurately calculate the amount of effluents that fish release to the environment, as nutritionists measure and monitor this variable on a regular basis for all intensively-farmed fish in carefully controlled tank experiments. It is another thing entirely to try and measure the nutrient flux released from fish cages or from mussel lines because a variety of uptake, dispersal and reaction processes occur at very short time and space scales. These processes make it extremely difficult to detect changes around aquaculture sites in all but the most sheltered and enclosed (generally unsuited for aquaculture) sites, when using standard oceanographic sampling approaches.

Lyngby (1990) claimed that measurement of dissolved nutrients generally provides little information about environmental quality because of the spatial and temporal variability associated with these measures (Wolanski *et al*., 2000; Karakassis *et al*., 2001). Because nutrient concentrations vary strongly over the day, it is necessary to sample frequently (this is quite costly) to capture these small-scale temporal fluctuations (Samocha and Lawrence, 1997; Karakassis *et al.*, 2001). Moreover, the increase in concentration above background levels is often so small that it is difficult to document by standard analytical techniques (Karakassis *et al.*, 2001, Dalsgaard and Krause-Jensen, 2006).

Nonetheless, the variables that are traditionally used to characterize water quality, at aquaculture, and other sites, include: concentration of ammonia (both ionised and unionised forms), nitrate, nitrite and soluble reactive P, and indirect measures of productivity, that is, DO, chlorophyll *a* content, turbidity and BOD, for land-based facilities). In order to interpret these data standard measures, including temperature and pH, are also recorded.

Depletion of DO can occur within and around finfish farms due to the respiratory activities of the farmed fish and microbial degradation of waste materials in seabed sediments. Excessive oxygen depletion in the water column could potentially stress or kill the fish and other animals (for example, epibiota), with sediment DO depletion resulting in the release of toxic by-products (for example, hydrogen sulphide) into the water, which can also have adverse effects on fish and other organisms (Ref.). Significant DO depletion at finfish farms has usually only occurred when cages are heavily stocked or where they are located in shallow sites with weak flushing (La Rosa *et al.*, 2002).

Case study in the Gulf of Eilat:

In contrast to the open ocean systems mentioned above, an extensive study found that even a relatively small aquaculture operation in the gulf of Eilat was found to be of major impact on the sensitive ecosystem of the gulf waters. The gulf harbors the northern most coral reefs and is characterized by crystal clear water and highest biodiversity under strict protection. However, the nutrient surpluses deposited, as well as escaping non native fish species, have recently been judged to be too heavy a burden on the gulfs ecosystem and the decision was taken to partly remove the operation from the gulf waters and changing the feeding regime (Evaluation of Pollution in the Gulf of Eilat; Report For the Ministries of Infrastructure, Environment and Agriculture. Ed. Atkinson, M. J., Birk, Y., and Rosenthal, H. Dec.10, 2001). Such considerations may be of general validity concerning fish farming in relatively closed parts of the sea where currents are insufficient to rapidly dilute surplus pollutants.

b) Effects on flora

The main concerns related to the effect of mariculture on water quality were that the discharge of nutrients from finfish farms could: a) lead to an increased occurrence of algal blooms, b) alter the ratios of nutrient elements in seawater thereby favoring the occurrence of toxic species over harmless algae, and c) make potentially toxic algae more toxic.

However, in a study prepared for the Scottish Executive Central Research Unit (2002) by local ecologists found no connection between the farm effluents and toxic algal blooms or algal physiology. Indeed, Tett and Edwards (2002) suggested that nutrient enrichment by fish farms would be insignificant unless the farm was located in an enclosed basin where water exchange was poor.. Furtheremore, dense stocks of bivalves (for example, mussels), may release large fluxes of ammonia into the surrounding waters (Maestrini *et al*., 1986; Dame, 1996). It has been suggested that elevated ammonia levels could stimulate phytoplankton production, possibly leading to more frequent algal blooms, including the domoic-acid-producing (toxic) diatom *Pseudo-nitzschia multiseries* (Bates, 1998; Bates *et al*., 1998). Whereas this is a possibility, the hypothesis has been challenged by conflicting evidence gathered in large research programs focusing on harmful algal blooms (Harrness, 2005). Despite the above-mentioned evidence, certain authors, for example, Assadi *et al.* (2007) claim that aquaculture activities are an important cause of coastal harmful algal blooms (HABs) and the debate on this matter continues. It is noteworthy that most of the literature on the topic of "aquaculture and harmful algal blooms" focuses on the risks and damage that HABs cause to aquaculture and not vice versa.

c) Effects on fauna

Zooplankton:

Because of their role as principal grazers and as prey for larger animals, zooplankton communities are a key component of coastal ecosystems. Any potential impacts from aquaculture on zooplankton might perturb the balance of phytoplankton dynamics, with subsequent risks of algal blooms. Surprisingly, however, there is remarkably little literature available on the impacts of marine aquaculture on zooplankton communities. In their 5 year study investigating the effects of sea lice medicines on sea loch ecosystems, Black *et al.* (2005) could find no correlation between fish farm activity and zooplankton dynamics in any of the systems they studied. Even though zooplankton are weakly motile, they are largely subject to water movement, and in areas of restricted exchange their populations may be more at risk from contaminants than in strongly tidal or well flushed systems. Even in completely enclosed systems such as freshwater lakes, the picture is not straightforward, with Fidalgo (1991) finding no significant difference between near-cage and distant station zooplankton communities, while Demir *et al*. (2001) detected larger zooplankton populations near trout cages in an Anatolian lake (Turkey!) compared with the distant station. Xie and Yang (2000) noted profound changes in copepod communities in Chinese lakes used for aquaculture, although they ascribed those changes to grazing by planktivorous fish. In contrast, Guo and Li (2003) observed higher numbers of rotifers, lower numbers of cladocerans, and no change in copepod numbers near freshwater fish cages in China.

In shrimp culture, the release of nutrients produced by the culture is known to have an effect on zooplankton communities downstream. McKinnon *et al.* (2002) and Trott *et al.* (2004) demonstrated that phytoplankton and bacterial populations, enhanced by the release of nutrients, were heavily grazed by zoo- and microzooplankton in the mangrove ecosystem of North Queensland, Australia.

Fish:

Given the well-known aggregation of wild fish to structures in aquatic environments, it is not surprising that wild fish are attracted to net-cage marine fish farms. Fish that aggregate around cages have been shown to have altered physiological condition, tissue fat content, and fatty acid composition compared to their wild counterparts (Dempster and Sanchez-Jerez 2007), mostly related to their consumption of uneaten feed which is not suited to the dietary requirements of the wild fish. Recent studies have suggested that finfish farming can increase regional fish biomass and promote the conservation of wild stocks, even beyond the immediate vicinity of the cages (Dempster *et al.*, 2004, 2006; Machias *et al*., 2004). In their study of fish farms along the Spanish Mediterranean coast, Dempster *et al.* (2002) found that while diversity measures varied between farms, individual abundance, biomass and species richness of wild fish around the cage structures were inversely correlated with the distance of the cages from shore; the same diversity measures were positively correlated with the size of the fish farms. The number and diversity of wild fish attracted to cage structures also seems to vary in relation to position in the water column. In their comparative study of fish farms, Dempster *et al.* (2005) found that Mediterranean fish cages in Spain overwhelmingly attracted wild fish at the same water depth as the cages, while in the Canary Islands, abundances and biomass were highest among bottom dwelling wild fish at one farm site, and highest at the surface at a second cage site.

The effects of accidental loss of farmed fishes include competition, predation and reproduction inhibition and disease transfer between caged and wild fish, and ecosystem effects (Moyle *et al*. 1986) as well as genetic interactions between farmed species and their wild conspecifics (Youngson *et al.* 2001). Introduction of species (accidental or deliberate), may alter or impoverish the existing communities and populations of the receiving ecosystems through inter-breeding, predation and competition for food, space, habitats, etc. For example, *Penaeus japonicus* (shrimps) from Japan but farmed in European lagoons have colonised vast areas of European marine areas. In India, the invasion of Tilapia and African catfish in the natural environment is destroying the original fauna of the system. Genetic pollution of indigenous stocks is possible. In Norway, the number of salmon escaping from fish farms exceeds the wild population, resulting in wild types diminished due to introgression of domestic stocks, loss of genetic adaptation to local conditions as well as potential susceptibility to diseases.

I.2.2.2 Impact on benthic environment

The emphasis on benthic (as opposed to pelagic) impacts is linked, among other things, to the fact that: a) there is an obvious impact directly under many farms whereas impacts are rarely observed in the water column, b) because benthic communities integrate the environmental stress over longer time periods than pelagic ones, benthos has become a standard monitoring (long-tradition) procedure for all types of marine pollution (beginning with organic enrichment; Pearson and Rosenberg, 1978).

The primary source of particulate matter from marine aquaculture is faeces and uneaten food. When organic input to the sea does not exceed the ability of the system to break it down aerobically, the rate of decomposition of organic matter is determined by a number of factors, among them carbon utilisation by benthic macrofauna, decomposition by heterotrophic bacteria, leading to straightforward oxidation to carbon dioxide and water. Many studies have indicated that the extent of altered benthic community structure and biomass is limited to less than 50 m distance from the fish cages (Kalantzi and Karakassis and references therein). However, there are signs that pelagic fish, benthic invertebrate and seagrass communities may be affected to a larger distance (Dimech *et al*., 2000; Machias *et al*., 2005; Pergent-Martini *et al*., 2006; Holmer *et al.*, 2003). When the threshold point is exceeded (*that is,* in shallow water sites, in areas with poor water exchange, or where overfeeding/excessive biomass occurs), sedimentary anoxia takes place. Near-bottom oxygen becomes depleted and aerobic processes shift to anaerobic ones (Brown *et al.*, 1987; Wildish *et al*., 1990). Hence, water depth and current velocity are critical factors determining patterns of sedimentation around cage sites (Weston, 1990; Pohle *et al*., 1994; Silvert, 1994; Henderson and Ross, 1995; Burd, 1997; Pohle and Frost, 1997; Brooks, 2001; Cromey *et al.*,

2002) and the spatial extent (distance) to which we may anticipate to find benthic effects. At one extreme, unsuitable sites that have very shallow water columns and long residence times will rapidly show benthic deterioration, while at the other extreme, deep-water sites with strong flushing may show no benthic impacts at all (Hartstein and Rowden, 2004; Klaoudatos *et al.*, 2006)!

a) Effects on sediment quality

Unlike the often ephemeral effect of aquaculture on water quality, its impacts on benthic ecosystems are more pronounced and often have negative effects on benthic communities. Various researchers have established that the organic enrichment of sediments (Karakassis *et al.*, 1998; Sarà *et al.*, 2004; Holmer *et al.*, 2005) enhances bacterial activity (La Rosa *et al.*, 2001; Vezzuli *et al.*, 2002), in particular anaerobic activity, which leads to reduced sediments (Holmer and Kristensen, 1992; Holmer and Frederiksen, 2007) and a decrease in diversity in meiofauna communities (Mazzola *et al*., 2000; Mirto *et al.*, 2002). Sarà *et al.*, (2004) observed that sediments around the cage were organic-enriched at about 1,000 m from cages. Benthic batecteria were closely related to organic enrichment and their density was three times higher in stations beneath the cages (Vezzullli *et al.*, 2002). Total sediment metabolism (measured CO_2 production across the sediment-water interface) was about 10 times higher during the farming periods than at unaffected control station (Holmer and Kristensen, 1992) and sulfate reduction, was directly correlated with the sedimentation of waste products from the examined fish farms (Holmer and Frederiksen, 2007).

Many aquaculture-impact studies have focused on the distinct fish-farm footprint marked by dark sediments (due to iron-sulfide deposition), bacterial mats and reduced macrofauna communities, which may be collectively referred to as “near-field” effects (for example, Tlusty et. al, 2005, Holmer et al., 2005). These “near-field” impacts have been modeled by Cromey *et al.* (2002) and Stucci *et al.* (2005) and the predictive models may be very useful tools for aquaculture site selection and for management of the coastal environment. Although it is generally accepted that the bulk of aquaculture impacts are concentrated in the immediate surroundings (within 50 m of the edge) of the farms, several sedimentologists have examined the distribution of fine particulate matter effluents and their “far-field” effects (Smith *et al.*, 2005; Milligan and Law, 2005). It appears that several elements associated with fish farming, for example, zinc and copper, may serve as indicators of trace metal enrichment (Dean *et al.*, 2007) at distances occasionally exceeding hundreds of meters from the farms (Yeats *et al.*, 2005). Robinson *et al.* (2005) studied the effect of increased macroalgal growth, related to aquaculture activity (>1 km away), on clam growth in New Bruswick, Canada, and concluded that this may be a case of negative far-field impact of one form of aquaculture on another.

Benthic communities beneath fish farms are often quite different from natural communities in unimpacted sediments. The sediments under fish cages have altered physical structure (changes in grain size distribution, texture, porosity, etc.) and different porewater chemistry (hypoxia, anoxia, pH, sulfides, porewater nutrient levels, etc.) and these affect the biological composition of the sediments (Costa- Pierce, 1996; Burd, 1997; Boesch *et al.*, 2001; Vezzulli *et al.*, 2002). During anaerobic decomposition, microbiological activity in the sediment switches to the bacterial degradation of organic matter using nitrate and sulphate as an electron acceptor, reducing them to compounds such as methane, hydrogen sulphide, ammonia and carbon dioxide. The characteristic black colour of these reduced sediments arises from the binding of iron to sulphide, with excess hydrogen sulphide released to the pore water after available iron becomes depleted (Jørgensen, 1982; Morris, 1983). Sulphate reduction has been estimated to account for 75% of sediment metabolism under marine fish cages compared to 50% in surrounding areas (Holmer and Kristensen, 1992). The process of nitrification/denitrification, by which bacteria oxidise ammonia to nitrates and nitrites, and reduce nitrates to nitrogen gas, becomes inhibited as nitrate levels increase, and may cease to function as a mechanism for removal of organic nitrogen directly under fish cages (Kaspar *et al.*, 1988).

b) Effects on benthic fauna

Meiofaunal communities:

Few studies (Duplisea and Hargrave, 1996; Angel *et al.*, 2000, Mazzola *et al.*, 2000; Mirto *et al.*, 2000; La Rosa *et al.*, 2001, Lampadariou *et al.*, 2005a) have examined the response of soft-sediment benthic meiofauna to aquaculture derived organic enrichment. The results have shown that changes in meiofaunal communities may serve as sensitive indicators to environmental pollution as a result of the small size of the organisms, direct benthic recruitment (sediment colonization) and short generation times.

Macrofaunal communities:

As a result of the major effects described above, the scientific and regulatory communities have focused most efforts on describing changes in macrofauna abundance and diversity associated with aquaculture wastes (Tsutsumi, 1995; Karakassis and Hatziyanni, 2000; Pearson and Black, 2001; Wildish *et al.*, 2001; Wildish and Pohle, 2005). The well studied distribution of benthic organisms along a gradient of fish farm effluent follows the pattern of organic enrichment effects reviewed by Pearson and Rosenberg (1978).

The benthic assemblages in the heavily enriched (often anoxic) sediments directly below the fish farms are generally most impacted, with low faunal diversity and biomass (*that is,* with small body size), but sometimes present in very large numbers (over 100,000 individual.m^{-2}; Nickell *et al.*, 1998; Pearson and Black, 2001). The early stages of disturbance to the benthic fauna from fish farm enrichment typically results in the dominance (in the range of 10^3-10^4 individuals.m^{-2}) by the opportunistic polychaete *Capitella sp.* (Nickell *et al.*, 1995; Nickell *et al.*, 1998; Karakassis *et al.*, 1999; Pereira *et al.*, 2004). In the most severely affected fish farming areas, up to 90% of the benthic macro- and megafaunal species previously present may be eliminated (Weston, 1990), as the number of species that are able to cope with extremely enriched conditions is very limited. Indeed, an overabundance of organic matter arriving on the sea bed may result in a physical overwhelming of the feeding apparatus (tentacles, cilia etc.) of many of the members of the benthic fauna. This resilient community (able to withstand altered ecological conditions without changing substantially) is extremely important in controlling oxidising conditions in sediments. Attempts to model and estimate sediment irrigation and benthic nutrient flux are completely unrealistic without considering the contribution of bioturbating organisms (Gust and Harrison, 1981). While microbial activity in sediments is important in determining the sedimentary response to organic enrichment, bacterial diversity and abundance have been shown to depend on bioturbatory activity (Aller and Yingst, 1978; Branch and Pringle, 1987; Dobbs and Guckert, 1988). It is now accepted that macrofaunal and megafaunal bioturbation considerably influences the biological, chemical, and physical nature of the sediments (Aller, 1978; Aller, 1982; Rhoads and Boyer, 1982; Nickell, 1992; Nickell *et al.*, 2003). Hansen *et al.* (1990) estimated that on an annual basis, ~ 40-50% of organic matter from marine fish farms was degraded in the presence of this macrofauna, compared to 11-15% when the macrofauna was absent. Thus in the absence of bioturbation by macrofauna and megafauna, aerobic sedimentary processes are restricted to the top few millimetres. High rates of diagenesis (organic breakdown) in marine sediments are therefore dependent on the activity of these burrowing organisms.

The area outside the zone of greatest impact, the “transitional zone” (Pearson and Rosenberg, 1978) is often characterized by increased diversity and high biomass as it is organically enriched (nutritious) yet not extreme with respect to sediment conditions (DO, H_2S). In an unimpacted benthic community, there is usually a wide range in animal size, increasing with distance from the source of enrichment (for example, fish cages).

The depth distribution of macrofauna subjected to organic enrichment has been described by Pearson (1987). While the majority of the macrofauna can be expected to exist in the surface layer of sediment all along the enrichment gradient, the maximum depth at which macrofauna penetrate increases markedly towards the source of enrichment. Farthest away from the enrichment source, Pearson (1987) found 2-3% of the fauna below 12 cm depth. Moving horizontally inwards towards the source of enrichment, less than 1% of the macrofauna were found below 12 cm at the edge of the enrichment, and greater than 5% of the fauna were present at depth at the source of enrichment.

In the same way that distance from the fish farm creates a gradient along which benthic communities become structured, so does distance in time. When a fish farm ceases operation and the surrounding environment begins to recover, the same changes in abundance, species richness, and biomass occur that Pearson and Rosenberg (1978) first observed in other areas of organic enrichment. A number of studies (Nickell *et al.*, 1995, 1998; Karakassis *et al.*, 1999; Pereira *et al.*, 2004) have examined the changes in benthic community succession following cessation of fish farming, both in northern and southern European waters, and have found similar relationships, including periodic collapses in the recovery process. Nickell *et al.* (1995, 1998) found biological recovery (compared to reference conditions) took up to 24 months in the most impacted areas at a salmon farm on the Scottish West Coast. In contrast, Karakassis *et al.* (1999) found that a sea bass and sea bream site in Cephalonia, Greece had not recovered fully after 23 months. However, the time for recovery relies on a complex process that depends on the hydrography, depth, nutrient status of the water body, additional (seasonal) inputs (to the initial load of organic matter), antibiotic residues, etc.

A classical approach to assess environmental impacts of human activities, based on these above benthic community characteristics (for example, individual abundance, species richness, biomass) is the use of biological indicators (bioindicators). Indeed, as benthic populations and communities are very responsive to anthropogenic environmental alterations (such as aquaculture), numerous biological indices have been established to assess these effects. A bioindicator is an anthropogenically-induced response in biomolecular, biochemical, or physiological parameters that has been causally linked to biological effects at one or more of the organism, population, community, or ecosystem levels of biological organization (McCarty and Munkittrick, 1996). The most popular ecological indices used to describe changes in macrobenthic communities include: number of species, biomass, Shannon-Wiener diversity, Pielou evenness, AMBI (Borja *et al.*, 2000) and Margalef index (Margalef, 1956). Values generally increase with distance from the aquaculture sites. The ratios of abundance over number of species, and of biomass (animal size) over abundance are also well known indicators of organic pollution (Pearson and Rosenberg, 1978) and have proven useful in impact studies around marine fish farms (Nickell *et al.*, 1995, 1998). Other indices exist on macrofauna, even if they are poorly used to deal with aquaculture effects. The first global index (for example, number which summarize lots of information) created was made for freshwater environments in France (called IBGN) and other synthesized ones were then created for marine environments. But as existing biotic indicators are mainly for seas, one of the current aims of the water framework directive (WFD; 2000/60/CE) is to determine the most appropriate index for transition waters such as lagoons, which exhibit special characteristics and communities. Even though in many cases, the relationship between abundances of indicator species and ecological factors is semi-quantitative, such approach may be sufficient for environmental management purposes and can thus be quite valuable (Stewart 2005).

By consuming waste feed and assimilating nutrients, wild fish aggregations have the potential to ameliorate seabed effects beneath fish farms (Felsing *et al*., 2004; Dempster *et al*., 2005). In studies from Western Australia (Felsing *et al*., 2004) and the Mediterranean (Vita *et al*., 2004), wild fish have been shown to reduce the amount of feed that reaches the seabed by as much as 60-80%, having thus a positive effect on nutrient discharges from aquaculture (Machias *et al*., 2004). Additionally, any feed that does reach the seabed may be quickly consumed by bottom feeding fish (Thetmeyer *et al*., 2003). However, a significant consumption of waste pellets also yields increased defecation by wild fish, with expected increases in dissolved organic C and NH_4^+ into the water column (Fernandez-Jover *et al*., 2007).

c) Effects on epibenthic communities

The effects of fish farming on epibenthos, both inter- and subtidal are much less well documented, and with conflicting results, with Angel and Spanier (1999) and Angel *et al.* (2002) reporting the settlement of diverse epifaunal communities next to fish cages in the Red Sea, whilst Boyra *et al.* (2004) reported a significant shift in species composition from filter feeding epifauna in areas free from fish farming in the Canary Islands, to pollution-tolerant algae on hard substrates near fish cages. In their study of the potential for using epifauna as biofilters to mitigate aquaculture nutrient release, Cook *et al*. (2006; in the EU FP5 project, *BIOFAQs*; http://cordis.europa.eu/data/PROJ_FP5/ACTIONeqDndSESSIONeq112482005919ndDOCeq268ndTBLeqEN_PROJ.htm) found enhanced epibenthic biomass at fish farm sites compared with controls in Scotland and the Red Sea. Changes in species richness, however, differed between areas, with a greater number of species being found on the fish farm structures than the controls in the Red Sea, fewer on the fish farm structures in Slovenia (Adriatic), and no difference in Scotland. Additionally, different taxa dominated the epibenthic communities in the different study areas, with the fish farm epibenthos dominated by tunicates in Scotland, bryozoa in the Adriatic, and algae in the Red Sea. Clearly more research needs to be done on this component of the marine ecosystem and its response to nutrient efflux from aquaculture.

d) Effects on benthic flora

In addition to benthic faunal responses to aquaculture wastes, marine macrophytes and seagrasses may also be affected.

On the one hand, the increased nutrients present in the water column as a result of fish farming activity may significantly increase the growth of macroalgae (Buschmann *et al.*, 1996; Troell *et al.*, 1999). Although little quantitative literature is available on the subject, the integrated open water culture of macroalgae and finfish relies on this assumption being valid. Some early work in the Baltic recorded enhanced growth and biomass of *Fucus vesiculosis* (Roennberg *et al.*, 1992) transplanted to near a fish farm, while Ruokolahti (1988) measured increased biomass and growth of the intertidal alga *Cladophora glomerata* near fish farms. Halling (2004) found increased growth rates of up to 40% (*that is*, 7% SGR.day^{-1}) of the red alga *Gracilaria chilensis* grown in conjunction with salmon over monoculture alone. At increasing distances from the fish cages (up to 300 m), stable isotope ratios confirmed the fish farm as the source of increased nutrients. The general conclusion for the impact of fish farming on macrophyte ecology would thus appear to be one of increased growth and biomass as a direct result of nutrients, both suspended and dissolved, at least for the few species studied.

On the other hand, the physical presence of cultivated shellfish or macroalgae on long-lines, or large finfish net cages can create large shaded areas on the underlying seafloor. In addition, suspended solids in the water column may reduce sunlight penetration into the water column, and these factors with altered physic-chemical structure of the sediment, may impact photosynthetic activity, thereby affecting benthic macrophytes. Holmer *et al.* (2003), in a review on the effects of fishfarm nutrients on *Posidonia oceanica*, indicated that detrimental effects from the particulates settling on the seagrass causing decreased phosynthetic ability, both by physically obscuring light and by enhancing epiphytic (microphytobenthos, filamentous macroalgae, etc.) growth. The decreased photosynthetic ability due to enhanced epiphytic growth as a result of nutrient enrichment may account for a decrease of ca 25% in seagrass density near fish farms (Cancemi *et al.*, 2004). The possibility exists that the enhanced epiphytic growth also encourages algal grazers, which have a deleterious effect on the seagrass (Holmer *et al.*, 2003). Indeed, an extensive epiphytic load may decrease seaweed productivity (Littler and Littler, 1999), growth (Buschmann and Gómez, 1993; Worm and Sommer, 2000) and survival (D'Antonio, 1985). The negative effect of fast-growing epiphytic algae has been proposed as one factor explaining the decline of perennial macroalgae in nutrient-enriched areas (Vogt and Schramm, 1991; Worm and Sommer, 2000) and as well affect the success of introduced seaweeds.

While there are 5 European seagrasses (*Zostera marina*, *Z. noltii*, *Posidonia oceanica*, *Cymodocea nodosa*, *Halophila stipulacea*), there is a lack of published information on the effects of fish farming on any species apart from *P. oceanica*. *Posidonia oceanica* seagrass meadows are considered “determining elements” in assessing the biological quality of the Mediterranean coastal zones (EU Directive 2000/60/EC, of October 23rd, 2000), and they are highly vulnerable to human activity, such as marine aquaculture (Delgado *et al.*, 1997; Pergent *et al.*, 1999; Ruiz *et al.*, 2001; Pergent-Martini *et al.*, 2006). Posidonia beds suffer large-scale losses in response to nutrient enrichment (Ruiz *et al.*, 2001; Cancemi *et al.*, 2003), under or near sea cages, the meadows of *Posidonia oceanica* die, and this might go on for several years even after the cessation of activities (Delgado *et al.*, 1999; Holmer *et al.*, 2003; Pergent-Martini *et al.*, 2006). A 50% reduction in vertical growth of *Posidonia oceanica* near cages was observed by Marba *et al.* (2006), although these authors did not correlate the decreased growth specifically with fish farm nutrients, ascribing the impact to general fish farm effects. Similarly, Ruiz *et al.* (2001) reported a 28% loss of seagrass meadow (with 53% overall degradation) in a 10 year period as a result of fish farming; although these authors also reported large variability in nutrient release, they attributed the impact to “the dispersion of dissolved nutrients and detritus from the farming activity”. One of the conclusions from the MEdVeg project (www.medveg.dk), on the interactions between fish farms and seagrass beds, was that the input of organic matter and nutrients may have a negative impact on *Posidonia oceanica* beds as far away as 200 m or more from the farms. Clearly, more work needs to be done to firmly establish the role of nutrient release from fish farming on the ecology of seagrasses.

I.2.3 Innovative measures to detect the impacts

a) Innovative measures of pelagic impacts

As stated above, although we know that fish farms release substantial amounts of dissolved effluents to their surroundings, it is difficult to quantify the effects of aquaculture on water quality as neither dissolved nutrient levels, nor algal abundances (or biomass proxies, for example, chlorophyll *a*) are elevated near the farms. This is related to the highly variable ("patchy") temporal patterns of nutrient and particle release from the farms, and requires the use of innovative approaches. The alternative to traditional oceanographic or limnological methods of water quality measurement which cannot detect patchy phenomena (Lyngby 1990) is to use approaches that can quantify fluxes, *that is*, provide integrated measures of nutrient release (Costanzo *et al*., 2001, 2005).

Bioassay methods have been demonstrated as an effective alternative to these traditional methods to evaluate fish farm impacts. Phytoplankton bioassay, recently developed by Dalsgaard and Krause-Jensen (2006), shows that although phytoplankton abundances are similar next to fish farms and at reference sites, the phytoplankton growth potential is much greater near the farms (higher flux of inorganic nutrients) than at the reference sites. Martí *et al*. (2007) demonstrated the usefulness of toxicity bioassays in describing the environmental impact of fish farming. The authors found that sea urchin larval toxicity was significantly correlated with sulphides, seasonally and with total ammonia-nitrogen in the fish farms studied. Then, larval toxicity bioassays could be used as indicators of ecological damage to benthic infaunal communities.

b) Innovative measures of benthic impacts

A lot of new indices based strictly on benthic bioindicators are still created in order to be fewer dependants on taxonomic knowledges or other constraints detected. Based on the species response to an environmental change, indices have been used in some cases in order to detect effects of marine aquaculture activities on benthic communities. AMBI index (Borja *et al*., 2000), based on the Glemarec and Hily (1981) species classification regarding their response to pollution, as well as the one proposed by Simboura and Zenetos (2002), the Norwegian Indicator Species Index (ISI) (Rygg *et al*., 2002) or the Benthic Quality index (BQI) (Blomsquist *et al.*, in press), all applying the very same principles, are good examples. Roberts *et al*. (1998) also proposed an index based on macrofauna species which accounts for the ratio of each species abundance in control vs. samples proceeding from stressed areas. This proposal is however semi-quantitative as well as site and pollution type specific. In the same way, the Benthic Response Index (Smith *et al*., 2001) is based upon the type of species in a sample (related to pollution tolerance), but its applicability is complex as it is calculated using a two-step process in which ordination analysis is employed to quantify a pollution gradient within a calibration data set. The AMBI index, for instance, which accounts for the presence of species indicating a given type of pollution as well as species indicating a non polluted situation, has been considered very useful in term of implementing the European Water Framework Directive (2000/60/EC) in coastal ecosystems and estuaries. In fact, although this index is very much based on the paradigm of Pearson and Rosenberg (1978), which emphasises the influence of organic matter enrichment on benthic communities, it has been shown useful for the assessment of other anthropogenic impacts such as physical alterations in the habitat, heavy metals inputs, etc.

It has been shown that reliable monitoring of benthic impacts can be achieved at a much lower cost using identification at higher taxonomic levels (Karakassis and Hatziyanni 2000), or various combinations of lower taxonomic resolution, larger sieve mesh size and smaller area samplers (Lampadariou *et al.* 2005b). The Biomass Fractionation Index (BFI), based on the ratio of the biomass of small body size (retained on 0.5 mm sieve but passing through 1 mm sieve) over total macrofaunal biomass, was found to be a quick method for detecting benthic impacts of fish farms and particularly useful in shortage of taxonomic expertise of any kind (Lampadariou *et al*., in press). The index was successfully applied at seven commercial fish farms in the eastern Mediterranean, and it was observed that BFI decreased with distance from fish cages.

Marine benthic macrophytes, in their turn, respond directly to the abiotic and biotic aquatic environments, and thus represent sensitive bioindicators regarding their changes (Orfanidis *et al*., 2001). Pergent-Martini *et al.* (2005) identified the descriptors of *Posidonia oceanica*, constituting the first step to allow the use of this species to assess the ecological status of the Mediterranean coastal zones. An index (POMI, *Posidonia oceanica* Multivariate index) was developed based on those physiological, morphological, and structural descriptors combined into a variable using a PCA (see Romero *et al.*, 2005).

Finally, sediment profiling imagery (SPI), an integrated analysis of the structure of the sediment and its biological activities, has also been used as a low cost and rapid surrogate of standard analysis for detecting benthic effects of fish farming (Karakassis *et al.*, 2002).

The above changes in benthic community structure (individual abundance, species richness and biomass) are generally recognized as common in all temperate areas, and their predictable nature has led to attempts to model these successional changes in response to varying organic (fish farm) inputs. One such model DEPOMOD has been validated for salmon aquaculture in Scotland (Cromey *et. al.*, 2002), and another for cod (Cromey *et al.*, 2007). These models accurately predict an indicator of benthic community disturbance (the Infaunal Trophic Index) in response to modelled deposition of solids from the farm; the salmon model DEPOMOD is in use by the Scottish regulator for setting consents. The effects of sea bass and sea bream farms on the benthic ecosystem have also been modelled in the Mediterranean using the MERAMOD model (Cromey *et al.*, 2004).

c) Innovative measures of bentho-pelagic impacts

Another innovative approach that is gaining popularity is the use of stable isotopes to detect spatial extent of fish farm effluents. Some authors have combinated this technique with the use of bioassays, like Jones *et al.* (2001) who pesented the first study using isotopical techniques in macroalgal bioassays to asses the extent of impact from shrimp effluent. Several studies have demonstrated the use of macrophytes as bioindicators of fish farm effects since they can provide information about the ecological impact of anthropogenic nutrient inputs (Udy and Dennison, 1997; Lyngby *et al.*, 1999; Lin *et al.*, 2007). Moreover, it has been demonstrated that primary producers are more affected by fish farm waste than other components of the food web taking up aquaculture-derived nutrients (Vizzini and Mazzola, 2004). Actually, many studies support the effectiveness of that combination to evaluate anthropogenic nitrogen from other sources (Cole *et al.*, 2005; Deutsch and Voss, 2006) as well as to evaluate the extent of nutrient release from fish farms (García *et al.*, in press). Plants exposed to fish farm effluents showed significantly higher $\delta^{15}N$ signatures than those collected at reference sites. This shift seems to be caused mainly by the higher $\delta^{15}N$ of the N-rich fish wastes rather than by changes in biochemical fractionation during N uptake

(Yamamuro *et al.*, 2003; Cohen and Fong, 2005) or by other factors such as light intensity or differences in species (Grice *et al*., 1996).

Stable isotopes have also been applied to trace the flow of nutrients from the farm, as a point source, to the associated food web to evaluate the extent of fish farms impacts. Lojen *et al.* (2005) used stable isotopes to trace the flow of effluents from a Red Sea fish farm to the surrounding community of fouling organisms, obtaining $\delta^{15}N$ values significantly differents in bioassays exposed at fish farm effluents than in reference sites. Whereas the signal provided by $\delta^{13}C$ was inconclusive, the $\delta^{15}N$ signature was much clearer and the authors reported that among the organisms tested sponges, tunicates and polychaetes showed greatest uptake of farm-derived N. Vizzini and Mazzola (2004, 2006) and Vizzini *et al.* (2007) measured $\delta^{15}N$ in consumers of the the vicinity of fish farms in Italy and found that contrary to expectations, the area affected by aquaculture waste (about 500 m) was larger than previously thought. Dolenec *et al.* (2007) measured $\delta^{15}N$ values in invertebrates founding that the organisms analyzed also reflected the enrichment of the environment in ^{15}N due to the presence of enriched effluents from aquaculture operations. Lin and Fong (2008) showed the usefulness of three bioindicators-growth, issue N content and N stable isotope signature ($\delta^{15}N$) in macroalgae to detect the aquaculture activities as source of nutrients.

Stable isotope approach has also been validated to detect the spatial scale of fish farm effluents in organic matter in the sediment (Ye *et al.* 1991; McGhie *et al.* 2000; Yamada *et al.* 2003). The study of Yamada et al. (2006) carried out applying technique of nitrogen stable isotopes in sedimentary organic matter detected extensions of fish farms effluents of 300 m from a farm in Japan.

On the other hand, seagrasses responses to organic matter inputs such as reductions of cover and density, decrease in shoot size, appearance of necrosis in leaves, reductions in vertical rhizome growth and change in epiphyte loads enable us to use them as bioindicator of marine environmental degradation in studies that monitor the effects of fish farming (Frankovich and Fourqurean, 1997; Dimech *et al.*, 2000; Cancemi *et al.*, 2003; Marbà *et al.*, 2006). Seagrasses also showed physiological changes that can be used as bioindicator such as increase in tissue nutrient content, alterations in free amino acid content and composition, decreases in carbon reserves, and variations in the nitrogen stable isotopic ratio and trace metals content, among others (Udy and Dennison, 1997; Pergent-Martini *et al.*, 1999; Invers *et al.*, 2004; Vizzini and Mazzola, 2004; Pérez *et al.*, 2007, Pérez *et al.*, in press). In general**,** physiological responses were reported to be better indicators of saturating nutrient supply to an environment than morphological changes since physiological responses can be used as early indicators of marine environmental degradation caused by aquaculture activities.

I.3 Fisheries

Assessing the scale of fisheries effects relative to other impacts can be difficult, because of confounding and interacting combinations with other anthropogenic effects (for example, pollution, habitat degradation, climate change) and natural variability of environmental factors.

Impacts from fisheries on the environment have been well described and reviewed (Dayton *et al.*, 1995; Goñi, 1998; Kaiser *et al.*, 2003; Gislason, 2003; Agardy, 2000). More specifically, capture fisheries impact target resources; reduce their abundance, spawning potential and, possibly, population parameters (growth, maturation, etc.). Fisheries modify age and size structure, sex ratio, genetics and species composition of the target resources, as well as of their associated and dependent species (Kenchington, 2003). When poorly controlled, fisheries develop excessive fishing capacity, leading to overfishing, with major ecosystem, social and economic consequences. According to Pauly and Christensen (1995), the overall impact of fisheries at a very large scale has been described as comparable, in aquatic systems, to that of agriculture on land in terms of the proportion of the system's primary productivity harvested by humans.

The alteration of the habitat by fishery activities may be physical (for example, through the "ploughing" effect of dredges and trawls), or biological (for example, through introduction of species; alteration of trophic cascades). Fishing may result in changes in productivity of resources (some positive and some negative) and affects associated species. Some aspects of fisheries can have significant and long-lasting effects, for example, destructive fishing techniques using dynamite or cyanides or inadequate fishing practices (for example, trawling in the wrong habitat); loss of fishing gear, possibly leading to ghost fishing; lack of selectivity, affecting associated and dependent species, resulting in wasteful discarding practices, juvenile mortality, added threat to endangered species, etc.

I.3.1 State of captures from fisheries (over-exploitation of natural resources)

More recent publications hold that excessive fishing has become the main destabilizing factor of ecosystems, directly, through removals and associated impacts, as well as indirectly, through the aggravation of eutrophication and subsequent oxygen depletion (Jackson *et al.*, 2001). Overfishing transforms an originally stable, mature and efficient ecosystem into one that is immature and stressed by various ways (*that is,* by targeting and reducing the abundance of high-value predators, fisheries deeply modify the trophic chain and the flows of biomass and energy across the ecosystem (Pauly, 1979); using unsuitable fishing gears that result in a high level of wasteful by-catch and destruction of egg bearing and juvenile fish (Vijayan, 2000); etc).

For instance in Kerala, India, several species including Ribbon fish, Bombay duck, tuna, seer fish, lizard fish, croakers, penaeid shrimps and Bull's eye were exploited above optimum level and thread fin breams were exploited at optimum levels whereas *Sandinella guttatus* was exploited below the optimum level. About 12% declines in total marine landings were observed in Orissa, India compared to the previous year. Among commercially important groups, fishery of oil sardine, Bombay duck, Croakers and Seer fishes, Ribbon fishes, Penaid prawns and Cephalopods recorded a decrease in the catches and there is a slight increase in lesser sardines.

Scientific studies have revealed that there is decline in the biomass of demersal resources assemblage. During the last few years, reduction in trawl fishing operations was observed because of declining catches and catch rates that reflects decreased resource availability, especially of high value demersal resources such the prawns and cephalopods (Srinath *et al.*, 2003). The significant increase in the fishing hours observed from 1985 seemed to level off in the recent years mainly because of decline in the catch per hour. This could also be attributed to the smaller mesh size of ring seines, which increases the probability of juveniles being caught (James *et.al*.1991; Deveraj *et al*.1997; Edwin and Hridayanathan, 1998). According to Pillai (2006), it is observed that there is overfishing due to increased fleet size and over capacity of fishing vessels.

Data available on fisheries production are based on sales from co-operative societies, which do not take into account for other important sources and uses of fish produced, such as illegal removal of fish by poachers, non-marketed production (subsistence consumption) and illegally marketed fish (Das *et al.*, 1996). Thus, the highly productive fisheries of Kerala are known to suffer from overexploitation. It was indicated that fishing effort by existing pole and line units have attained near optimum in India, and maximum increase in fishing effort is limited to 20% of the existing effort to attain maximum economic yield.

Over the past few decades, the average trophic level of fish landings has declined, indicating that capture fisheries are increasingly turning to small pelagic fish that are lower on the food chain (Garcia and Newton 1997; Pauly *et al.* 1998). Trites (2003) and Cury *et al.* (2003) have reported that removal of top predators such as mammals, tuna or sharks, may release an unusually large abundance of preys at lower levels with cascading and feedback effects on the food chain and species composition. For example, as most sharks and some batoid fishes (angel fishes) are predators located at or near the top of marine food webs, their depletion modifies the intricate trophic interactions of their ecosystems (Pauly and Murphy, 1982; Jackson *et al.*, 2001). The removal of predators through fishing in Kenyan reefs resulted in the expansion of sea urchin population, which apparently led to a decrease in live coral and a loss of topographic complexity, species diversity and fish biomass (McClanahan and Muthiga, 1988).

I.3.2 Nonselective fishing gear and consequences

Unsatisfied with the reports of the member states, a study group from the International Council for the Exploration of the Sea (ICES) on the effect of bottom trawling was convened in 1987 to collect information available since 1972. Based on this database, many member states initiated national and international studies on the effect of trawling on seabed and the benthic communities (Bergman and Hup, 1992).

a) Impact of trawling

Bottom trawling has been reported to be the most destructive type of fishing method prevalent in the world fishing sector that inflict drastic changes in the marine ecosystem by way of removal of fish and other benthic communities (De Groot, 1984, Bergman and Hup, 1992). During bottom trawling, large quantity of epifaunal and infaunal organisms are injured, removed and killed due to the passage of heavy otter boards and nets (Auster *et al.*, 1996). Many studies have been conducted globally in order to assess the long-term and short-term impacts of bottom trawling (Krost *et al.*, 1990, Auster and Langton, 1999; Joice and Kurup 2005, 2006). The mortality of benthic species associated with or preyed upon by target bottom fish resources resulting from the use of trawls can vary greatly, depending on how the gear is built or rigged (Moran and Stephenson, 2000).

Jones (1992) reported that the direct contact of trawling gear with substratum by means of ground rope, chains, bobbins, sweeps doors chaffing mats or parts of the net might results in scrapping, ploughing and sediment resuspension. Jennings *et al.* (2001) described trawling and dredging as the most destructive fishing practices, which cause innumerable direct and indirect changes in the ecosystem. Direct changes in the fish population and in the benthos can occur by the scrapping of trawl gear on the seabed (Reimann and Hoffman, 1991). The consequences of trawling include variation in the fish stock (Deep-sea trawls catch a quantity of undersized fish) and changes in the mortality (for example, quantity of young and eggs of demersal fishes in India are hauled up by trawl nets, particularly during the monsoon), recruitment/settlement, diversity and production of benthos (Pearson and Rosenberg, 1978). Caddy (1973) found that the passage of the dredge stirred up a load of suspended sediment, which reduces the visibility from 4-8 meters to less than 2 meters for 10-15 minutes, and can affect the oxygen budget and nutrient level (Caddy, 2000). The magnitude of effect depends on the depth of penetration of the gear in to the sediment (Bridger, 1970; De Groot, 1984) as well as on the type of benthic habitat and the type of the organism (Kaiser *et al.* 2005). Physical impacts on sediment due to bottom trawling have been observed by various technics such as direct observations, underwater cameras (Caddy, 1973) or Side-Scan sonar (Fonteyne *et al.*, 1998).

Coral reef ecosytems (Smith 1978) have suffered significantly from reef fishing for both ornamentals and food fish (Johannes and Riepen, 1995). Bottom trawling (for example, dragging of weighted nets across the seafloor) causes significant levels of bycatch and substantially disturbs seafloor ecosystems. There is also increasing concern about damage to cold water coral (*Lophelia pertusa*) from trawling in Norway, Ireland and Scotland, with up to 50% of these deep water reefs being affected in Norwegian waters (Fosså *et al.*, 2002; Grehan *et al.*, 2005; Hall-Spencer *et al.*, 2002). It is estimated that worldwide trawling disrupts an area as large as Congo, India, and Brazil combined (Watling and Norse, 1998).

b) Evaluation of by-catch and discards

The catching of non-target species is known as “by-catch”. The composition of the by-catch and the quantity depend on the gear, area of operation and season. The majority of bycatch is kept and sold because the species caught are marketable (providing a significant contribution to the revenues of some fishing operations). Alternatively, they can be used as feed for aquaculture species. Discards of by-catch are the portion of the total catch which is dumped or thrown into the sea. With some types of fishing, such as trawling, the amount of bycatch landed (and subsequently sold) can be several times the amount of target species landed. However, if by-catches are not thrown away, adaptation of boats appears necessary (for example, larger, stronger, with more capacity, etc.). Nevertheless, much bycatch known as “trash fish” (poor compared to the valuable catch of shrimp and marketable fish) is simply discarded into the sea for economic or regulatory reasons. According to the FAO’s report (Alverson *et al.*, 1994), it is estimated that 27 million tons or approximately 27% of the global catch are discarded annually. As non-target species tend to be neglected by conventional assessment and management, the risk is high that they be overfished with serious consequences on their reproductive capacity as well as on the food sources of the target and other species.

The discarded fish in the Western Mediterranean (Benidorm, SE Spain) comprised mainly of sardine, flatfish and horse-mackerel (Martínez-Abraína *et al.*, 2002). The average ratio of fish discarded over fish landed was ca. 65% and they estimated that trawling waste was probably enough to support a local gull population four times larger than that present during the study period. The by-catch also includes accidental capture of endangered species in fishing nets such as marine mammals, (for example; Read *et al.* 2006; Casale *et al.*, 2004).

Since 90% of the mechanised boats operating along the Kerala Coast are bottom trawlers, that is to say non-selective fishing gears, which haul up all the organisms dwelling at the sea bottom, destructive effect to the non-target organisms has been rising as a matter of great concern in India. The fishing pressure from water trawlers along the coastal waters of Kerala is reported to be very high compared to any other maritime State of India. Hence, the Department of Ocean Development with the Cochin University of Science and Technology, under the research project “Impact of Bottom Trawling in the Sea Bottom and its Living Communities along the Coastal Waters of Kerala”, have quantified discards from 375 bottom trawlers operating from six major harbours during the period April, 2002 to March, 2002, along the Western Coast of India, including non-edible species and species which are poisonous in nature, non-marketable and inferior quality species. Chandrapal (2005) concluded that around 2.4 millions tons of discards (accounting for 37.13% of finfish, 28.46% of crabs, 8.13% of stomatopods, 9.94% of gastropods, 1.96% of shrimp, 0.85% of jellyfish, 1.5% of cephalopods and 1.17% of soles) are thrown back into the sea from bottom trawlers operating along the Kerala waters annually due to non-edible nature, unpopular nature of species and size, low market value and lack of storage facilities, etc. The edible portion of the discards is worked out around 0.85 lakh tonnes per annum (lakh = 100,000 tonnes) because 94% of the bottom trawlers have cod-end mesh size of 18 mm and below instead of 35 mm imposed by the Government, which may bring about serious biodiversity degradation in the coastal waters. With the high survival rate of crabs, stomatopods, gastropods, etc., there is a fear that their proliferation may amount to the transformation of the mature ecosystem into the immature and inefficient ecosystem over a period of time.

In India, a study taken up to develop a methodology for the assessment of shrimp by-catch (led by the National Resources Institute (NRI), U.K. and funded by the Overseas Development Administration (ODA) and the Bay of Bengal Programme's post-harvest fisheries project) estimated the quantity of by-catch as 100,000-130,000 millions tons during the year 1988, because selling price of these varities of fish (mainly Sciaenidae, Leiognathidae, Nemipteridae, Clupeidae, Trichiuridae, Carangidae, Mullidae, Harpadontidae and Menidae) was less than 5 Roupies per kg. Another study on discards into the sea on the East-Coast of India indicated that the total trash fish available from the (about 3450) trawling boats is about 2 lakh metric tons per year, in the late 1990s (Chandrapal, 2005).

I.3.3 Pollution and biodiversity concerns

Substantial amounts of gear may be lost due to storms, entanglements on reefs and rocks, and other mishaps. Lost or discarded fishing equipment cause significant harm to wildlife. Indeed, fishing vessels generate significant quantity of garbage including fishing nets, monofilament lines, hooks, traps, and packing bands and containers for frozen bait. The nets are mostly made of polyethylenes and polypropylenes. The nets lost or discarded drift and washed ashore or sink. A number of factors like whether it is loose or bagged and the physical and chemical characteristics of the solids decide the fate of garbage after it is discarded (Swanson *et al*., 1994). Large, dense particles sink quickly and the small particles tend to disperse in the surface layer. Emulsified particles may remain in the water column for longer periods of time. Organic material may or may not sink; garbage and sewage-related discharges have been observed in windrows up to 5 kilometres long in the coastal ocean and often "wash ashore as waves of debris" (Swanson *et al*., 1994).

Only little scientific information available on the effect of debris on marine invertebrates and plants or marine habitats apart from observations that debris damages coral reefs, is ingested by squid (Araya, 1983; Machida, 1983), and may present a new habitat niche for encrusting marine species (Winston, 1982). Fowler (1987) linked the increased entanglement of northern fur seals (*Callorhinus ursinus*) to increase in fishing effort in the North Pacific and Bering Sea and opined that the factors collectively suggest that mortality of fur seals due to entanglement in marine debris contributes significantly to declining trends of the population on the Pribilof Islands. Chiappone *et al.* (2005) studied the impact of lost fishing gear on Florida Keys coral reef ecosystem and found that the lost hook-and-line fishing gear accounted for 87% of all debris encountered and mostly responsible for the documented impacts. Laist (1987) overviewed the effect of lost and discarded plastic debris in the marine environment on the biota and mentioned that the accumulating debris poses increasingly significant threats to marine mammals, seabirds, turtles, fish, and crustaceans. Lewison (2004) studied the impact of pelagic longlines on loggerhead and leatherback sea turtles. More than 200,000 loggerhead and 50,000 leatherback sea turtles were estimated to have been taken by the global pelagic longline fishery in the year 2000, based on bycatch data from more than 40 nations and from 13 international observer programs. In the past 20 years, the Pacific loggerhead and leatherback populations have declined by 80-95%. The authors calculated that the current bycatch levels for these turtles are unsustainable, and more efforts need to be made to better quantify the bycatch, and to find ways to reduce it. Pruter (1987) reviewed the literature on the sources, amounts and distribution of various types of plastics in the marine environment and found the major sources of these materials are from land, vessels and beachgoers. Green and Mattick (1977) discussed and reviewed the possible methods for the utilization or disposal of fishery solid wastes.

Hoss and Settle (1990) compiled a list based on existing literature and their own work of at least 20 fish species (larva, juvenile, and adults from benthic to pelagic habitats) reported to ingest plastics. Adults had ingested a wide variety of items, including rope, plastic pellets, packaging, sheeting, cups, cigar holders, a bottle, and colored fragments suggesting that the primary source of anthropogenic debris is marine vessel and fishing activity (Moore and Allen, 2000). Bart (1990) reported plastics in 12 percent of the yellow fin tuna and 3 percent of the blue fin tuna caught off the coast of Virginia. Higher percentages of plastics found in these and other pelagic species have been attributed to more frequent association of these fish with areas where debris concentrates, such as in drift lines. Derraik (2002) reviewed the deleterious effects of plastic debris on the marine environment and discussed the mitigation measures. He mentioned that marine animals are mostly affected through entanglement in and ingestion of plastic litter. The likely source of most debris is the multinational trawl fisheries of the North Pacific Ocean. Threats of debris have even been reported for whales, monk seals (*Monachus schauinslandi*), coral, and other wildlife in northwestern Hawaiian Islands (Donohue *et al.*, 2001; Henderson, 2001; Boland and Donohue, 2003; Boren *et al.*, 2006).

Sobsey *et al.* (2003) found that the potential sources of faecal coliform are the sanitary wastes from boat occupants discharged into the surrounding water.

Regarding the sustainability of fisheries industry in general, besides the effects on target species and the habitat degradation, it is also worth noting that fishing depends heavily on petroleum-derived fuels, consuming more than 1.2% of the global oil production (Tyedmers *et al.* 2005). However, its use of energy is more efficient than other food production systems (such as beef or egg production or salmon farming).

Introduction of species

Exotic invasive species are ranked as the second cause of biodiversity threat (Wilcove *et al*., 1998; Mack *et al*., 2000), leading to species extinctions, changes in hydrology and ecosystem function, among others. Due to the expansion of global transport and trade, the number of species that have entered new areas through human activity has increased by several orders of magnitude during the past centuries (di Castri, 1989) and this accelerating invasion of marine systems by non-indigenous species (NIS; Carlton, 1996) has become a subject of environmental concern (Carlton and Geller, 1993; Vermeij 1996; Ruiz *et al.*, 1997, 1999, 2000; Cranfield *et al.*, 1998; Orensanz *et al*., 2002). A large number of species, either intentionally or accidentally, are transported between biogeographical regions or introduced to new areas. Not only the ballast water and fouling on ship's hull are responsible for species introductions, but also the import of species for aquaculture and recreation. It remains thus difficult to incriminate one factor more than another in introduction of invasive species, since all these factors can be linked. For example, fishing vessels can contribute to the dissemination of species which have already been introduced in the habitat by other mean; hence, fishing gear can uproote invasive algae from a bottom and bring them in another place (for example, the Mediterranean *Caulerpa taxifolia*). Then, the ability of a species to establish is influenced by a number of factors such as climate, interactions with other species like competitors, predators, pathogens and other abiotic conditions of the new area. Exotic species that spread beyond the point of introduction and become regionally abundant are termed invasive species (Richardson *et al.*, 2000, Kolar and Lodge, 2001). Numerous marine non-indigenous species have been identified as stressors which affect native species, their community structure and function and ecosystems as well (D'Antonio and Vitousek, 1992; Baskin, 1998; Steffani, 2001; Orensanz *et al.*, 2002, Robinson and Griffiths, 2004) and vectors of negative changes affecting local economies and human health (Hallegraeff, 1998; Bryan, 1999; Harvell *et al.*, 1999).

The introduction of marine exotic species may result in severe ecological perturbations in native communities (Bax *et al.*, 2001; Piriz and Casas, 2001; Grosholz, 2002). In seaweed assemblages, competition for light and substrate can be intense, leading to the local exclusion or sharp decrease of native species (DeWreede, 1996). Dramatically reduced local biodiversity have been predicted in the presence of invasive kelp *Undaria pinnatifida* among many areas (Battershill *et al.*, 1998; Graciela Casas *et al.*, 2004).

II. Limiting the waste (before production)

II.1 Agriculture

II.1.1 Introduction

Tools to meet environmental targets for nutrient management in agriculture :

Improving the plant nutrient use efficiency (NUE) in farming systems requires attention to all stages in the production, especially when handling manure. Examples will be presented on the role of element balances for agriculture and spreading strategies on fields with focuing on diminishing N and P leaching losses to water.

At the national level, element balances have become implemented as tools to meet environmental targets for nutrient management in agriculture, either on a voluntary or a mandatory basis. Goodlass *et al.* (2003) identified no less than 50 input output accounting systems (IOAs) for nutrients in use in EU member states. Comparing them, the authors concluded that (1) if IOAs are to be used in a European-wide context a more uniform and coherent concept for a balance is needed ; (2) overall IOAs can offer a useful tool for voluntary improvement in agri-environmental performance and topics that are not strongly regulated ; (3) more studies are needed to ensure that farmers in reality change their behaviour, and (4) there is a need to develop reference values.

In the Netherlands, the N and P accounting system MINAS (MINerals Accounting System) was first voluntary but become later a key instrument in the implementation of the Nitrate Directive (Hanegraaf and Den Boer, 2003; Schröder *et al.*, 2003). Nutrient balances are also included in the Swedish action plan designed to deliver the target reduction of N losses to the marine environment from agriculture (SJV, 2000). The action plan includes multiple measures and the expected improvement arising from education and advisory activities that rely heavily on farm-level nutrient balances using the STANK model has been estimated to a 20% increase in NUE (Linder, 2001). The accounting system MINAS allows maximum N and P surpluses in the Netherlands, which decrease stepwise until a level has been achieved that satisfies agronomic and environmental needs, including the Nitrate Directive (Dir 91/676, 1991). At De Marke experimental dairy farm, the Netherland, a relationship was found between the farm N surplus and nitrate concentration in groundwater, although there are variations between years (Aarts *et al*., 2000). However, RIVM (2002) and

Hanegraaf and Den Boer (2003) have questioned whether the target N surpluses of MINAS can guarantee that the limit of 50 mg nitrate.l^{-1} will not be exceeded. The tenuous link between N surplus and N leaching in less intensive agricultural systems are illustrated by examples from three small Latvian watersheds. In a 6-year study by Jansons *et al.* (2003), there was only very weak relationships between N surplus and leaching losses at the scale of the water catchment area. The losses were instead more strongly correlated to the acreage of arable crops within a catchment, reflecting the impact of agricultural management related to crop production.

A better understanding of system inertia is also important to understand the spatial variability and temporal aspect of nutrient loss. This complexity is difficult to handle in the nutrient accounting balance system. Positive relationships between nutrient loss and the proportion of agricultural land under regular cultivation or with high stocking densities have been described (Johnes *et al.*, 1996). These broad relationships between N surplus and leaching have been used to develop the concepts of loss coefficients, which have been used widely, although mostly at the water catchment areas (Johnes *et al*., 1996). Despite the existence of this general relationship, there is less evidence to support specific relationships between balances or inputs of N (Bechmann *et al.*, 1998) and P (Edwards *et al.*, 2000) and nutrient losses. Fundamental differences between N and P behaviour in soils and in their mobility through the landscape govern therefore the time scales over which surpluses of N and P become important. Reductions in N and P surpluses alone will not be sufficient, and their relative importance for combating environmental problems must be assessed in relation to other forms of loss and land use management (Öborn *et al.*, 2003). Under the Nitrate-Sensitive Area (NSA) monitoring scheme, measures to reduce the surplus have been combined with improved management techniques such as better timing of manure inputs and establishment of over-winter cover crops (Lord *et al.*, 1999). Similarly, for P, controls over land use and manure management affecting soil erosion and run-off rates are more likely to have a short-term measurable impact on reducing P export to surface waters (Edwards and Withers, 1998).

II.1.2 Alternative farming methods to pesticides/herbicides

a) Allelopathy

The effect of synthetic pesticides on the environment has necessitated looking for alternate farming methods. Allelopathy is defined as a direct or indirect effect by one plant, including micro-organisms, on another through production of chemical compounds that escapes into the environment to influence the growth and development of neighbouring plants (Rice, 1974). The most active allelochemicals in rye residues, hydroxyamic acids, which occur as glycosides in the living plants, is known to have strong inhibitory action on germinating dicotyledons and monocotyledons weed seedlings (Barnes and Putnam, 1983). The unique relationship between plants and their pathogens suggests that micro-organisms may be a better source of future herbicides than allelochemicals produced by higher plants (Duke, 1986). Crop rotation and intercropping are alternate methods of biologically managing the weeds. Intercropping of *Desmodium uncinatum* and *D. intortum* are reported to reduce the infestation of *Striga hermonthica* due to a putative allelopathic mechanism (Khan *et al.*, 2001). The use of allelochemicals from the plants in bio-control of weeds has been reviewed by Rajendran and Gnanavel (Personal communication 2008a,b).

b) Biological weed control

Sustainable agricultural system involves a range of technological and management strategies to reduce costs, protect health and environmental quality, and enhance beneficial biological interactions and natural processes (National Research Council, 1989). Hence alternative weed management strategies for sustainable agriculture are gaining significance in the recent past. Biological weed control is the deliberate use of natural enemies (plant feeding and disease causing organisms) to reduce the densities of the weeds to economically or aesthetically tolerable limits. Currently, the most effective means of managing weeds are herbicides, which account for more than 60% of all pesticides used in crop production (Gianessi and Puffer, 1991). Repeated use of herbicides has been implicated in weed shift, depression in the rhizosphere microflora contamination of groundwater, soils, and food products which may threaten public health and safety. Bioherbicides (as biocontrol) strategy can be used to sustainably control weeds; using insects takes longer period of time (Harley *et al.*, 1996). Bioherbicide play a significant role in organic farming, an agricultural production system, which avoids or largely excludes the use of synthetic fertilizers, herbicides, pesticides, fungicides, growth regulators and livestock feed additives. Herbicide resistance among the weeds also poses a great problem. But, bioherbicides offer wide scope to combat this resistance problem. The potential for successful use of bioherbicides in managing herbicides-resistant biotypes was demonstrated where growth of an imazaquin-resistant common cockleber biotype from soybean field was suppressed with the mycoherbicides, *Alternaria helianthi* (Abbas and Barrentine, 1995). Bioherbicides have an important role in managing invasive weeds, defined as those alien plants spreading naturally in natural ecosystems and producing significant changes in terms of composition, structure or ecosystems process. The phytopathogenic bacterium *Pseudomonas syringe p.v. phaseolicola* was shown to suppress the growth of the invasive weed *Pueraria lobata* (Zidack and Backman, 1996). Integrated weed management assumes greater significance in sustainable agricultural systems. Application of selected deleterious rhizobacteria (DRB) during tillage may be effective in integrated weed management (Kremer and Kennedy, 1996).

c) Organic manures

Low budget technologies are needed to address the farmers need and to manage environmental problems. Organic farming using organic manures, panchagavya, dasagavya and Amritha karaisal are considered as low budget technologies. Amritha karaisal can be effectively used for controlling pests and increasing crop yield. It is made up of cow dung, cows urine, jaggery, field soil, millet variety and curd. The solution can be applied as a soil tonic as well as plant spray (Personal Communication, Dr A Murugan, 2008).

II.1.3 Animal feeding improvements

There is a low efficiency of conversion of dietary nutrients into purchased livestock (meat, milk, eggs) at farm level. Withers (1996) estimated that 55% of P applied as fertilizer is removed in crops but only 18% of dietary P is exported in consumable animal products. About 82% of fertilizer P ended up within the farm. To improve nutrient use efficiency on mixed farms, there is a need of integrative approaches that address livestock diet, grazing and manure management, as well as agronomic practices. In making the decisions to sell or to feed crop produced, the farmer is potentially influencing not only the growth and health of livestock, but also the quality and quantity of manure. This in turn affects future crop production and quality, and the need of purchased fertilizers (Watson *et al.*, 2005).

In livestock production systems, low nutrient utilization efficiencies are inevitable, since only a small proportion of nutrients ingested is actually retained. This varies with livestock species, but is approximately between 13 and 28% N for dairy cows, between 5 and

13% N for sheep, and between 4 and 10% N for beef cattle (Henzell and Ross, 1973). Actual values also differ with age; for example, in pigs the proportion of N retained has been calculated to vary from 18% in piglets to 47% in weaners, the corresponding values for P being 14% and 39%, respectively (Fernández *et al.*, 1999). Reproductive status is also important: more N and P is retained in sows during lactation than from dry sows. Matching dietary intake to the requirements of different growing stages could thus substantially reduce nutrients in excreta (Watson *et al.*, 2005). Ration balancing has the potential to ensure animal health and production requirements while minimizing adverse environmental impacts (Lynch and Caffrey, 1997; Rotz *et al.*, 1999; Valk *et al.*, 2000). Some examples: Reducing dietary P from 0.49% to 0.40 has been shown to reduce faecal P excretion by 23% without compromising milk production or reproductive performance (Wu *et al.*, 2000). In pigs, adding phytase to improve the availability of P or adjusting the dietary amino acid balance to match requirements are both options that are likely to have a major impact on reducing P and N excretion (Fernández *et al*., 1999). Reducing P intake of dairy cows on Wisconsin (USA) farms from current practice to recommended national levels reduced the number of farms with positive P balance by 67% (Powell *et al.*, 2002). Changing diet formulations affects not only total nutrient excretion, but also the availability of nutrients in excreta and their partitioning between dung and urine (Wohlt *et al.*, 1991; Kebreab e*t al.*, 2001).

Changing diet to reduce environmental impact has cost implications. Commercial feeds are often based on least-cost formulation, which oversupplies nutrients because cheaper raw materials often have a poorer balance of amino acids and lower digestibility. On-farm mixing of rations from arable crops produced on-farm as well as crop residues or outgrades returned from the vegetable production often form an important part of livestock diets. However, these feed components also need careful management and analysis of nutrient content and dietary value in order to improve NUE (Watson *et al.*, 2005). Rations for livestock are often formulated with large safety "margins" so that nutrients can exceed nutritional requirements by as much as 30-50%. While this may not affect animal performance, it can result in excess application of both major and trace elements to soils via manure. On mixed farms with a high degree of crop and livestock integration, this should be less likely than on specialist farms focused only on purchased feeds and animal products (Dou *et al.*, 2001; Watson *et al.*, 2005).

II.1.4 Measures to reduce ammonia emissions from stable, manure storage and spreading of fertilizers

a) The whole handling chain

In a project LIFE Ammonia, carried out 1999-2003, methods and techniques to reduce NH_3 emissions in milk production were studied and demonstrated on a farm in southwest of Sweden (Sannö *et al.*, 2003). The former manure-handling system for semi-solid manure was replaced by a slurry system. After rebuilding the tied barn, the dietary crude-protein concentration was lowered for the cows, the drainage of urine in the gutters was improved, the gutters were cooled by incoming water, and the exhaust ventilation air was filtered in a biofilter. Before rebuilding of the cowshed, all semi-solid manure was stored on a farm-yard manure pad and all urine in a separate pit. After rebuilding, all slurry was stored in a covered manure container. Semi-solid manure and urine were spread separately before rebuilding, and slurry was spread by band spreading or shallow injection after rebuilding. Totally, the ammonia losses from the farm were reduced by 48%, from 21.7 to 11.3 kg NH_3-N.cow^{-1}.yr^{-1}. If the effect of the biofilter was included, the losses were 7.3 kg NH_3-N.cow^{-1}.yr^{-1}, a reduction of 66%. Based on the project results, the following practical advices were given to dairy farmers:

- Analyse your home produced fodder
- Decrease the crude protein content in the feed
- Avoid high temperature in stable and manure
- Decrease emitting areas with manure in the stable
- Plan for sufficient storage capacity in order to avoid improper spreading occasions (windy, warm conditions)
- Cover the storage container
- Incorporate the manure immediately after spreading or with injector equipment. Band spreading in growing crop of certain height will also reduce NH_3 emissions.

b) Stable

In the Netherlands, Aarnink *et al.* (2007) listed the most relevant principles to reduce NH_3 from cow houses. They identifyed better feeding management, reduced emitting surfaces and adaptation of airflow along manure surfaces. Smits *et al.* (1995) showed the potential for reducing NH_3 emissions by reducing the rumen degraded protein surplus in animals. The NH_3 emission reduction of several solid floor designs was studied in detail by Swierstra *et al.* (1995) and V-shaped solid floors reduced the ammonia emissions with approximately 20-25%

compared with the emission level of a traditional cow hours with slatted floors and storge below. In pig houses, the fouled floor area could be reduced by keeping the temperature low. Huynh *et al*. (2004) showed that by using cooling systems, pen fouling during the summer can be prevented. The choice of bedding material could also influence NH_3 losses. For instance, peat added to the straw bed for young cattle resulted in significantly lower NH_3 emissions than with only straw (Jeppsson, 1999). Peat adsorbs ammoniacal N effectively and can prevent NH_3 losses (Witter and Kirchmann, 1999).

c) Storage

Ammonia losses can be sharply reduced if the air directly above the slurry store is prevented from circulating. A method that efficiently reduces NH_3 losses is to cover the slurry stores with, for instance, a roof, a floating plastic cover or a stable natural crust (Sommer *et al.*, 1993; Karlsson, 1996a; Smith *et al.*, 2007). If the slurry storage is filled underneath the cover, this can be kept intact even during filling, which reduces the risk of NH_3 emission (Muck *et al.*, 1984).

From storages with solid manure, especially if composting take place with high temperatures, NH_3 losses could be high (Karlsson, 1994). Peat included in the bedding material will also reduce NH_3 losses during storage (Jeppsson *et al.*, 1997; Karlsson, 1996a; Rogstrand *et al.*, 2004). In the old days, there were also roofs on solid manure storages, but with increasing numbers of animals and thereby big amount of manure, it is not common. However, it could be an effective measure to reduce ammonia losses from solid manure storages (Karlsson, 1996b). Additionally, a roof keeps rainwater away, which could prevent nutrient leakage from the manure pad if it has insufficient or lacking drainage.

d) Spreading

Good contact between soil and manure reduces the risk of NH_3 emission (Malgeryd, 1998). Results clearly show that the most effective way to reduce NH_3 emission after spreading is to inject or incorportate the manure into the soil. When applying slurry to a growing crop, placing the slurry in the canopy bottom in bands (bandspreading) gives a lower emission than broadcasting (Fig. 4). The reduction occurs because the crop canopy changes the microclimate near the soil surface, lower wind speed, temperature and radiation, and increased relative humidity (Thompson *et al*., 1990). Irrigation after spreading also reduces NH_3 emission (Malgeryd, 1996; Rodhe *et al*., 1996). Solid manure can give rise to substantially greater NH_3 emission than slurry when applied at the same rate under identical environmental conditions and should not generally be considered as a low-concentrated N fertilizer (Malgeryd 1996).

Figure 4: Bandspreading of slurry on ley

In a crop, special devices are required in order to achieve an efficient incorporation. For grassland, there are shallow injectors available that incorporate the slurry into the upper soil level to a depth of less than 0.1 m. The injectors are not designed to work for all soil conditions and, especially in dry and hard soils, the injectors do not penetrate to a sufficient depth (Smith *et al*., 2000; Rodhe and Etana, 2005) and reduction of NH_3 losses is consequently not achieved. However, in many cases, injection of slurry into the soil in grassland could be an efficient way to reduce NH_3 losses after spreading compared with surface bandspreading (Huijsmans *et al*., 2001; Misselbrook *et al*., 2002; Mattila and Joki-Tokola, 2003; Rodhe and Etana, 2005; Rodhe *et al*., 2006), Table 1.

Table 1: Reduction in NH_3 emission after shallow injection into ley compared with surface bandspreading

Soil description	Reduction in ammonia losses compared with band spreading, %		Reference
	Shallow injection, open slots	Shallow injection, closed slots	
Sand	20-50	75	Hansen et al., 2003
Sandy loam, silty	32 (0*-73);		Smith et al., 2000

clay loam, and clay			
Sandy, peat and clay	54	96	Huijsmans et al., 1997
Silty clay, silty loam, silty clay loam	0-48** (average)		Rodhe and Etana, 2005
Silty clay		100	Rodhe et al., 2006

* Low or no reduction at dry and hard soil conditions

** Sufficient injection technology

The reduced NH_3 emission found by Hansen *et al*. (2003) corresponded to a decrease in nitrogen losses of 3 to 7 kg $N.ha^{-1}$ with injection in open slots and 19 kg $N.ha^{-1}$ with injection in closed slots compared to band spreading. Smith *et al*. (2000) found an average loss of 11.1 kg $N.ha^{-1}$ after band spreading and 8.6 kg $N.ha^{-1}$ after shallow injection (10 experiments) into ley.

Figure 5: Shallow injection of slurry into ley

II.1.5 Measures to reduce losses from field application and fertilizers

a) Amounts of applied N, choice of crop and time of application

Several research has been carried out to quantify the magnitude of leaching following different management practices at field level, and to develop countermeasures against leaching (for example, Bergström, 1987; Macdonald *et al*., 1989; Djuurhus, 1992; Thomsen *et al*., 1993). The research results have been applied in recommendations through development of training and extension services and formulation of good fertilization practices. However, the fertilizer recommendation systems differ between European countries. For example, Rahn *et al.* (2001) reported very different strategies to derive N fertilizer recommendations for field vegetables in 15 European countries. In the context of nitrogen fertilisation, special attention should be paid to field vegetables, which often are intensively managed crop systems. Current nitrogen recommendations for intensively grown field vegetables aim at predicting economically optimum application rates of N fertilizers (Feller and Fink, 2002). Another approach is to use environmental fees. Since 1984, Sweden has used environmental fees to reduce the use of mineral N fertilizers. In January 2006, the N tax was SEK 1.80.kg N^{-1} (SJV, 2006).

An efficient way of reducing plant nutrient losses from arable land during the autumn and winter is to keep the land under vegetative cover (green land) during this period, particularly in areas with light soils and gentle climate (Aronsson, 2000). In Sweden, the rules state that in the very South parts, 60% of arable land shall be under vegetative cover during the autumn and winter. In the rest of southern Sweden, the requirement is 50%. There are also rules when certain crops must be sown and ploughed up in order for the area to be considered as being under vegetative cover during the autumn and winter (SJV, 2006).

Excessive N fertilizer applications of mineral N or animal manure undoubtedly increase leaching and a N application adapted to the needs of the crop is a key factor to decrease N leaching (for example, Bergström and Brink, 1986; Vinten *et al*., 1991). The timing is also important where autumn-applied manure on uncropped fields was found to be one of the most important sources of large nitrogen leaching loads (Djuurhus, 1992; Torstensson *et al*., 1992).

When animal manure is applied, it can be more difficult to estimate the most suitable mineral N addition. There is also a considerable uncertainty about how much of the NH_4^+-N in applied manure becomes available to the crop, when volatile losses and possible N immobilization after spreading are taken into acccount (Van Faassen and Van Dijk, 1987; Jackson and Smith, 1997). Spring application before sowing is recommended for manure, where NH_4^+-N is the major part of N, as it, on average, leads to a significant yield increase (Torstensson, 1998). When manure is applied in the autumn, it should preferably be spread in early autumn and on fields with an actively growing crop with high N consumption, such as leys or grass catch crops (Lindén *et al*., 1993).

Incorporated ley, green manure residues or catch crops can be a valuable source of crop available N, but also a serious source of leaching (Torstensson, 1998). The temporal distribution and total amount of crop available N released are difficult to predict with precision. The subsequent release of mineral N depends on the residue N and C content, the amount of residues incorporated and temperature and moisture conditions in the soil (Jansson and Persson, 1982; Thorup-Kristensen, 1994; Janssen, 1996).

b) Measures to reduce P losses

A great number of studies have been conducted on the relationship between soil P status and P losses into water. When the soil P values increase beyond agronomically optimum ranges, there is a reasonable consistens pattern whereby P losses increase significantly (Sims *et al.*, 2000). However, P losses have large spatial and temporal variations and can be influenced by several factors interacting with each other. It is therefore important to consider site-specific factors to be able to find measures to reduce P losses (Djodjic, 2001; Börling, 2003).

The development of risk assessment tools and Decision Support Systems can be valuable for an overview, processing and understanding of P problem-related issues. Hydrologic and nutrient models working in a GIS environment may improve the understanding of temporal variations and processes of importance for P behaviour in soil (Djodjic, 2001).

When adding P fertilizer, it should be incorporated into the soil rather than surface applied. Application of tillage practices that are more efficient in P incorporation and mixing into the soil are preferable, which enhances P sorption to the soil particles. The practice of plowing down surface applied P fertilizers should be closely examined due to the risk for ponded conditions above the plow pan in the top soil. Phosphorus applications of mineral fertilizers or manure should be avoided at sites and during occasions when P transport (surface run-off or preferential flow) is likely to happen. The aim should always be that no more P is added than the crop can use. High rate applications of store (basic) P fertilizer should be avoided as much as possible, especially in areas sensitive to leaching (Djodjic, 2001; Börling, 2003).

c) Reducing land degradation

Besides doing environmental damage to groundwater and surface water bodies, leak-age of nutrients, particularly of N, from overused soils as a result of soil organic matter loss is the driving force in land degradation and desertification not only in arid regions, but also in the Mediterranean and temperate climates. Degraded, infertile dryland with strongly reduced plant growth has low soil organic C and low nitrate contents, with the major missing factor being absorbed NH_3. In this case, incorporation of organic matter, low in N and high in C as for straw, will restore the N cycle and result in intermediate storage of N both as organic N and ammonia (Kanal, 1995; Maeder *et al.*, 2002; Garnier *et al.*, 2003; Pimentel *et al.*, 2005; Kramer *et al.*, 2006; Lal, 2006). Nitrate leaching can also be significantly reduced by no-till agriculture (Power and Peterson, 1998).

II.1.6 Soil tillage effects on the risks of nitrogen leaching

Another important measure for reducing N leaching from arable soils in cold-temperate, humid regions is avoidance of early autumn soil tillage in order to minimize N mineralisation during autumn and winter. Tillage such as ploughing is preferably made in late autumn or in spring before sowing of the next crop (Aronsson, 2000). An efficient way to reduce the accumulation of leachable N during autumn, following annual crops, is to delay tilling activities as much as possible until late autumn or spring, leaving the soil undisturbed during autumn (Vinten *et al*., 1991; Torstensson, 1998). In order to reduce N leaching during the period from October to March in Sweden, farmers can receive financial support if they choose to till their fields in the spring instead of in the autumn. Participation has been high and in 2005 about 91,000 hectares were subject to spring tillage (SJV, 2006).

II.1.7 Other waste

a) Trace elements

Since 1984, Sweden has used environmental fees to reduce the input of Cd to arable fields from mineral fertilizers containing P. The tax on Cd in mineral fertilizers containing P was in January 2006 SEK 30.g Cd^{-1} that exceeds 5 g Cd.ton P^{-1} (SJV, 2006).

To sustain long-term soil quality, monitoring tools, such as soil surveys and element balances, are required for trace elements (for example, Cd, Zn and Cu) (Keller and Schulin, 2003). Soil monitoring gives the current soil status, but is often not sufficiently sensitive to detect the slow rates of accumulation or depletion that occurs. These can readily be detected by soil surface balances (for example, Andersson, 1992; Keller and Schulin, 2003; Bengtsson *et al.*, 2003). There is generally no direct link between the impact of trace elements and their soil surface balance, but the latter makes it possible to evaluate the direction of change in a system, *that is,* a net accumulation or a net depletion in the soil over time (Öborn *et al.*, 2003).

b) Precision farming

Precision farming is an innovative technology that has been introduced by the Tamil Nadu Agricultural University (TNAU) in India to increase crop yield. This technology is meeting with large success in Tamil Nadu province. A precision farmer, has earned more than Rs. 5 millions in 11 months from his brinjal crop grown at a cost of 120 cents. The farmer harvested 170 tons rather than 60 tons in conventional farming. The fertilizers were used wisely (Continuing success of Precision farming in Tamil Nadu, 2008).

II.1.8 Extension services and information

a) Decrease of N and P losses

Since 1995, extension services and information are part of the Swedish Environment and Rural Development Plan. This plan includes regional targets for the activities. The county plans offer training, both to farmers (individual or groups). Individual discussions may give the farmer knowledge about environment friendly solutions for the handling of manure and other plant nutrients, based on the enterprise's situation and needs. When whole groups are gathered, county administrative boards and other operators may provide information and demonstrations about the best use of manure and mineral fertilizers in order to reduce the risk of plant nutrient losses. Gatherings like this are also good opportunities for exchanging valuable experiences (SJV, 2006). In areas identified as particularly vulnerable to plant nutrient losses, a project called "Focus on Nutrients" (www.greppa.nu) has been introduced within the county plans. This project provides knowledge and tools to farmers, helping them to reduce losses of N and P in a cost-efficient manner. For instance, extension services advocate farmers to reduce the rate of fertilizers when too much fertilizers are used, to move the spreading of manure to times when plant nutrient losses are minimal, and to adapt animal feeding better to need. The first individual extension service visit includes a presentation of the environmental objectives, and the farmer's environmental accounts are studied. During this first visit, the farmer and the advisor together draw up a plan for the extension services that may be relevant for that particular farm in the next three years. The idea is that the advisor from "Focus on Nutrients" also will return to the farms to follow up on changes. During the second visit, a plant nutrient balance is drawn up for the farm. Such plant nutrient balance is then repeated, in order to further improve nutrient handling and to monitor developments (SJV, 2006).

b) Decrease of occurrence of pesticides in water

In Sweden, some measures have been taken to minimise the risk to health and the environment during the 20-year period for which water quality data are currently available. Such measures include the Swedish national risk reduction programmes and an information campaign called “Safe Pesticide Use” that was launched in 1997. This campaign focused on safe filling and cleaning of agricultural sprayers and safety distances as regards surface runoff and wind drift (Törnquist *et al*., 2007). Comparing the quantities of pesticides sold to agriculture and horticulture in 2005 with the average quantities sold during the early 1980’s shows that there has been a 63% reduction in the 20-year period (Kem, 2006). The present risk reduction programmes includes intensive sampling from four small rivers draining catchments of 8-16 km^2, dominated by arable land. Water samples are time integrated and analysed for approximately 80 substances representing a majority of the pesticides applied in the area (Adielsson *et al*., 2007). Results from the beginning of the 1990s revealed concentrations of up to 200 $\mu g.L^{-1}$ for single pesticides, sometimes with high concentrations as a result of accidental spillages when filling or cleaning the spraying equipment on surfaces with drainage directly connected to the stream. Investigations also demonstrated very high concentrations (up to 2000 $\mu g.L^{-1}$) in run-off water entering surface water inlet wells on farmyards close to areas where filling of sprayers had taken place and, also, where the farmyard had been treated with herbicides to keep it free of weeds (Kreuger, 1998). During recent years concentrations in stream water have decreased by more than 90%, even though the amounts of pesticides uses in the area have not decreased. The reduction in concentration was a result of information campaigns directed to farmers in the area and political measures. Today most samples have a total concentration of pesticides below 2 $\mu g.L^{-1}$ (Adielsson *et al.*, 2007).

II.2 Aquaculture

Phosphours and N in farm wastes are of great concern due to their role in enhanced eutrophication, oxygen depletion, and turbidity in receiving waters. Therefore, it is essential that aquaculturists understand how to minimize P and N losses through proper fish farm design and management.

Thus, improving the feed quality and quantity in the both intensive and semi-intensive aquaculture would greatly optimise the system functioning and help to limit emission of wastes due to input surplus.

Food and feeding are the engine of growth and production, and their good management is as important as the design of the diet. The cost of feed is usually the greatest operating cost in aquaculture. The improvement of feeding is not only priority because of the cost of the feed, but also because, it is crucial for environmentally sustainable aquaculture.

II.2.1 Improvement of intrinsic feed quality

a) Elemental composition of feed

The crude biochemical composition of a body of a fish is variable, but can typically be: 20% protein, 1-2% lipid, 2% ash, 70% water, 15% C, 3% N, and 0.5% P (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. In intensive fish farms, the average pellet feed composition is: 40% protein, 17% fat, 19% carbohydrate, 14% ash, 7% water, 50% C, 6% N and 1% P (% dry wt.; Table 2). The ranges for the components are, however, quite broad. Carbohydrates and lipids are included in formulated foods to increase the protein retention efficiency and reduce N and P waste production (Adron *et al.* 1976, Ruhonen *et al.* 1999, Hillestad *et al.* 2001) which are considered as potential pollutants from aquafeed. Feed and fertilizers represent about 60-80% of the total cost of aquaculture production.

Table 2: Examples of the percent composition of fish food used in different aquaculture systems (% dry wt.).

Species	Food type	Protein	Fat	Carbo-hydrate	Ash	Moisture	N	P	References
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)
Gilthead sea bream	exp	39.9	10.4	35.5	10.9	10.1			Fountoulaki *et al.* (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)
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)
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)
White sea bream	exp	64.1	18.5		16.2	11.4			Sá *et al.* (2007)

The first step in the reduction of N and P wastes in aquaculture is the use of optimized feed formulations (for example, to find the most adequate ratio of elements, according to the model of Redfield's). Fish feed with low N and P content entails a reduction in excreted waste (Islam, 2005). It appeared easier to mitigate P than N content in fish feed. Feed-related P wastes can be minimized by using feed which is highly available to the fish yet have low water solubility. Fish meal, used in most fish feeds, contains bone which is a highly concentrated source of P, but it is not efficiently digested by fish. Recent advances in fish meal processing technology enable the removal of bone (Bell and Waagbø, 2008). On the other hand, soybean meal and other plant ingredients contain phytin, a plant P storage molecule. However, P in phytin is chelated and is released to the environment by the action of phytase and at low pH. Additionally, mixing water with the feed will enhance the action of phytase. Therefore, phytin-P is poorly digested by fish and other animals with simple stomachs (Hardy, 1999). Then, to increase the bioavailability of P in feeds, it is recommendable to increase the level of phytase in the feed (Papatryphon *et al.*, 1999; Baker *et al.*, 2001). To effectively control the excess N in the fish farm effluents, reducing the amount of N introduced into the system is the only solution since most N is excreted in dissolved form. Excreted N comes from several sources, including undigested and unabsorbed dietary protein, sloughed intestinal cells, amino acids absorbed in greater amounts than fish can utilize, and degraded metabolic products. Therefore, feed protein quality and quantity are important factors to consider in controlling effluent N, although some loss of N is unavoidable due to protein turnover in the fish and because enzymes that break down proteins are always active in fish (Jahan *et al.*, 2003; Schneider *et al*., 2004). For the production of feeds producing less solid waste, it is necessary to eliminate poorly digestible ingredients (such as whole grain or grain by-products used as binders and fillers in the feed formulae) and to use highly digestible ingredients with good binding properties (Schneider *et al.*, 2004). Further reduction of solid waste can then be achieved through careful selection of the ingredients to improve apparent digestibility and the nutrient balance of the feed. Feed that is high in lipids (fats) relative to proteins can reduce nitrogen excretion (Schneider *et al.*, 2004).

The formulation of feed is in a continuous development and where trash fish and wet feeds were used in the early periods of fish farming, this practice is almost absent now. Most fish farms thus use dry pellets, and the feeding efficiency in aquaculture can be evaluated from observing the food conversion ratio (FCR) between farms and species. FCR is most commonly defined as the amount of dry food consumed per wet fish biomass produced (Hall *et al.* 1990a,b). Accordingly, the FCR reflects the digestibility and quality of the feed, and thereby the efficiency of feeding. The feed quality is determined by the composition and the availability of the nutrients, *that is*, the digestibility. As parts of the food are not easily digested, the fish produce a certain amount of waste material (among others: N and P), and as 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. However, abiotic factors may influence metabolic pathways differently according to the species. For example, temperature has been tested on channel catfish (Andrews and Stickney, 1972), and salinity on juveniles spotted grunter, *Pomadasys commersonnii* (Deacon and Hecht, 1999), in order to improve FCR, but the results also showed that food intake depended on genetic strain (Silverstein *et al.*, 1999). Fernandez *et al.* (1998) studied the digestibility of pellet food in gilthead sea bream (*Sparus aurata*) and found that the apparent digestibility of elements were (in decending order): N > C > dry matter > P, with N, C and dry matter digestibility correlated. There is a large range in FCR values for cultured fish (FCR_{FISH}), based on individual fish in tank experiments, extending from as low as 0.64 for Red Pacu to 6.75 for Indian major carp (Table 3). FCR_{FISH} are generally based on experimental studies in tanks with individual fish, whereas FCR´s for farms (FCR_{FARM}) are higher and more variable (Table 4). The reason for this discrepancy is that direct food losses, as well as fish mortalities are included in the FCR_{FARM}. In addition, other factors like increased respiration due to stress (for example, temperature or feeding activity), may also contribute to elevate the feed conversion ratio (Islam, 2005, Mente *et al.*, 2006). FCR_{FARM} varies significantly between species with the fattening of tuna on frozen fish attaining extremely high FCR (25.6), whereas rainbow trout and salmon are close to values of 1. The relatively new species in sea cages, such as bream, bass and cod have relatively high FCR compared to well-established species such as salmon and rainbow trout (Holmer *et al.*, 2007).

Table 3: FCR_{FISH} (wet weight fish produced per weight feed consumed) for individual fish reported for different aquaculture fish species.

Species	Region	Food type	FCR_{FISH}	Reference
Asian sea bass	Asia	Exp pellet	1.03-1.33	Tantikitti *et al.* (2005)
Asian sea bass	Asia	Trash fish	2.86	Tantikitti *et al.* (2005)
Cuneate drum	Asia	Exp. pellet	1.05-1.4	Wang *et al.* (2006)
Indian major carp	Asia		4.55-6.75	Singh and Balange (2005)
Japanese sea bass	Asia	Exp. pellet	0.99-1.08	Xue *et al.* (2006)
Nile tilapia	Africa	Exp. pellet	1.43-2.53	Sweilum *et al.* (2005)
Red Pacu	North America	Exp. pellet	0.64	Palacios *et al.* (2006)
Red sea bream	Asia	Exp. pellet	1.14	Sarker *et al.* (2005)
Salmon	Norway	Exp. pellet	0.78-0.86	Hevroy *et al.* (2004)

Table 4: FCR_{FARM} for different combinations of fish species and food types (kg dry food supplied per kg wet weight fish produced).

Species	Region	Food type	FCR_{FARM}	Reference
Areolated grouper	Asia	Trash feed	6.52	Leung *et al.* (1999)
Atlantic blue fin tuna	Europe	Frozen fish	25.6	Aguado *et al.* (2004)
Barramundi snapper	Australia	Commercial food	1.23	Islam (2005)
Cod	Europe	Pellet	1.5	Gillibrandt *et al.* (2002)
Cod	North America	Commercial pellet	1.4-1.5	Chambers and Howell (2006)
Gilthead seabream	Europe	Commercial food	1.79	Lupatsch and Kissil (1998)
Gilthead seabream and sea bass	Europe	Pellet	1.60-2.39	Holmer and Frederiksen (2007)
Haddock	Europe	Pellet	1.5	Gillibrandt *et al.* (2002)
Haddock	North America	Commercial pellet	1.3-2.4	Chambers and Howell (2005)
Halibut	Europe	Pellet	1.3	Gillibrandt *et al.* (2002)
Mandarin, bream, catfish	Asia	Forage fish, formulated diet	2.29	Guo and Li (2003)
Rainbow trout	North America	Commercial pellet	1.14-1.28	Bureau *et al.* (2003)
Rainbow trout	Europe	Commercial pellet	1.75-1.86	Hall *et al.* (1990a)
Rainbow trout	Europe	Commercial pellet	1.83	Foy and Russel (1991)
Salmon	Europe	Pellet	1.17	Gillibrandt *et al.* (2002)
Silver perch	Australia	Exp. and commercial pellet	1.9-2.2	Allan *et al.* (2000)
Silver perch	Australia	Commercial food	2.45	Gooley *et al.* (2000)
Turbot	Europe	Pellet	1.3	Gillibrandt *et al.* (2002)

A compilation of data shows, that by fine tuning the composition of the food as described above, and increasing the availability of both energy and nutrients, it is possible to reduce FCR_{FISH} to very low values, like for example, 0.64 as reported for the Red Pacu (Table 3), whereas the use of trash fish is quite inefficient and lead to high FCR. For example, Asian sea bass had a FCR_{FISH} of 1.03-1.33 on pellets and 2.86 on trash fish. Also time of development plays a role for a decline in FCR. This can be observed from the progression of the FCR in salmon/trout farming during the past 3 decades due to improved management of the farms through optimizing feed, feeding practices etc. It can thus be expected that the FCR for the relatively new species in culture, such as sea bream and sea bass, will decrease, which has already been observed (Islam 2005). It is, however, questionable if they can become as low as for salmon as the optimization for salmon has been attained through increasing substitution of fish meal and oil with vegetable components. Salmons can tolerate high degrees of substitution, probably because they can feed both as carnivores and herbivores in their natural environment and at various life stages, whereas it is currently investigated how well carnivore species like cod and bream can grow on substituted feed formulas (Bell and Waagbø 2008). In this way, supporting and, where possible, participating in research directed towards reducing the percentage of fishmeal in cultured fish can help relieve the pressure on wild fisheries which comprise a large proportion of fishmeal.

The positive effects of improved management practices are clearly seen from a compilation of N and P release in the period 1987-2001 in the Danish marine rainbow trout production, where a reduction of 54% and 64% per produced tons of fish, respectively, was documented (Anon 2002, Fig 6). This has been explained by improved quality of the feed used along with a reduction in FCR. Similar trends in nutrient release can be expected for other intensive productions such as salmon and milk fish, and possibly also for sea bass and sea bream (Islam 2005, Table 3). It was shown in an intensive gilthead seabream farm, fed with extruded bream diets, that 180 kg of solids, 13 kg P, and 105.4 kg N were released to the environment through excretion and by uneaten feed for every 1000 kg of fish produced (Alvarado, 1997). Therefore, FCR assumes greater significance and needs to be addressed in order to have environment friendly aquafeeds. Hence, clear understanding of the essentiality of the feed and feeding management would help to negate the impact on the environment.

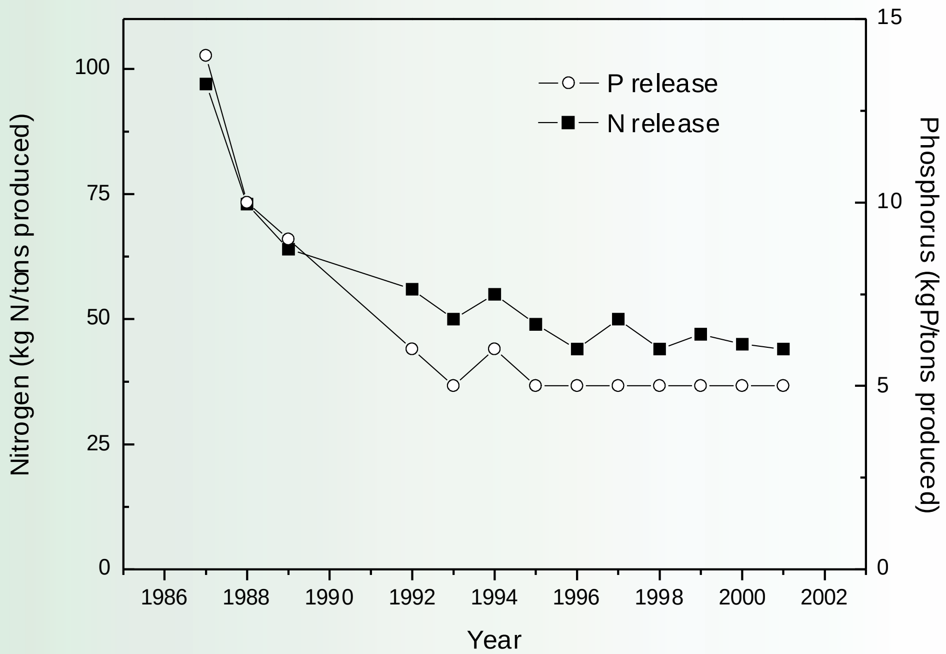


Figure 6: Change in release of N and P per tons of fish produced (Rainbow trout) in Danish marinefish farming during 1987 to 2001. The values are average of 25-30 fish farms (after Anon 2002).

b) Physical quality of feed

Other important way to minimize N and P waste is the type feed selection. Characteristics considered in feed selection include pellet formation, size, digestibility and palatability. Selection of the type of feed, in terms of physical attributes, can greatly influence the amount of waste produced at a facility and the resulting environmental impacts.

Three processing techniques currently dominate the production of pelleted diets in aquaculture and their use often, but not always, facilitate improvement in the raw product; grinding, steam conditioning and extrusion. These techniques inevitably affect both the physical and chemical characteristics of feed (Hilton *et al*., 1981): including, among others, water stability and durability, pellet hardness, nutrient availability (Tan, 1991) and digestibility (Hardy, 1989; Hui-Meng, 1989). But, while processing techniques such as those described can improve the nutritional and physical qualities of diet ingredients, heating processes can also have detrimental effects. Heat-sensitive vitamins and nutrients can be lost at elevated temperatures (Slinger *et al*., 1979; Kiang, 1989; Springate, 1991) and overheating can reduce the available essential amino acids (Evans and Butts, 1951; Carpenter and Booth, 1973; Viola *et al*., 1983).

Hilton *et al*. (1981) studied the effect of extrusion processing and steam pelleting on pellet durability, pellet water absorption and the physiological response of rainbow trout reared upon these diets. Extruded pellets were observed to be more durable, had superior water stability and absorbed more water than steam pellets. Trout reared on extruded pellets had prolonged gastric emptying in comparison with those reared on steam pellets. Trout reared on extruded pellets also showed significantly lower weight gain but higher feed efficiency than those reared on steam pellets. Booth *et al*. (2000) confirmed these results and detected that consumption of floating extruded pellets by fish was reduced. Then, the use of extruded pellets has many advantadges although the high cost of extruded feed deters farmers from using it. However, when the reduction in the cost of waste management is considered, along with the reduced feed conversion ratios, high energy extruded feeds have been proven to be more economical than regular pelleted feed. If taxes were added, according to the principle of “polluter-payer”, farmers would be obliged to use this more environmentally friendly form of feed.

Floating and dry pellet feeds allow the operator to monitor fish feeding activity and have greater stability than some sinking pellets, which enables the pellet to remain intact longer.

The size of feed pellets and the rate at which they are delivered may also affect the amount of feed an individual fish can ingest over a period of time. Pellets of sub-optimal size or pellets that are delivered at a high rate may cause wastage, as fish may be unable to catch large numbers of pellets before they sink through the net pen (Bailey *et al*., 2003). Stradmeyer *et al.* (1988) found that adult salmon showed a more immediate response to larger pellets but that these were more likely to be rejected than pellets of a shorter length. Smith *et al.* (1995) also found that pellets slightly smaller than the "normal" commercial size were eaten at the fastest rate, thereby indicating that the salmon are perhaps adjusting their feeding behaviour to compensate for a smaller feed pellet size. Some authors have suggested that for a range of fish species, the optimal feed size appears to be 25–50% of the mouth width (Wankowski, 1979; Wankowski and Tabachek, 1988). It appears that fish can adjust to reasonable deviations from the optimal pellet size with no significantly negative effects on growth. Therefore, feed "dust" and particles too small to be consumed by the fish should be screened out prior to feeding.

The freshness of the feed can also influence in the quantity of waste derived from fish farms. Excessive handling results in a greater percentage of fines that are generally not eaten by the fish. Regular deliveries and a good inventory system will keep the feed fresh. This is especially important in the hot summer months when the storage life of feed is reduced. Care should be taken to avoid feeding old feed. Many key ingredients such as vitamins are unstable beyond the time designated by the manufacturer. Fish will not fully utilize the feed if it is deficient in nutrients due to break down over time which will increase the amount of waste produced by fish. Most manufacturers recommend storing feeds in cool, dry conditions to maximize their shelf life. If feed is not purchased directly from the manufacturer, the purchaser should verify with the retailer that the feed had been stored properly.

To conclude, in most semi-intensive systems, supplemental feeding is based on fish biomass in the system, seldom taking care of environmental factors and natural foods, which are known to influence food consumption and fish growth. They are dispensed in powdered form, leading to significant wastage and difficult share among animals, particularly in small-scale rural operations (De Silva, 1993). A method that has been adopted in cyprinid polyculture, is using feeding bags. These bags are suspended at a number of locations with the perforated bottom touching the water surface (Tacon and De Silva, 1997) to increase the feeding efficiency. In Andhra Pradesh in India, farmers use perforated polythene bags suspended in wooden panels at many places in the pond. The feed is eaten by the fish in two or three hours, resulting in minimum wastage and helps the farmers to apply medication effectively through the feed (Veerina *et al*., 1993). In Bangladesh, supplemental feeds in carp farming are provided in both wet dough balls and in powdered dry mixture form. Miaje *et al.*

(1999) demonstrated that for Indian major carps and Java barb in polyculture, the supplemental feed comprising of mustard oil cake, rice bran and wheat bran in the form of pellet and dough, is more suitable than powdered form.

II.2.2 Feeding efficiency: control and physical improvement

Improvement of feeding can be done regarding methods of feed presentation, feeding rate and frequency. Besides choosing a high energy extruded feed for greater assimilation, waste management efforts will be most effective if focused on the quick removal of solids (Midlen and Redding, 1998; Miller and Semmens, 2002). Primary treatment, or solid waste removal, should be done as soon as possible to reduce waste fragmentation, which causes release of nutrients. Excessive waste accumulation has been shown to cause disease in fish farms (Gowen and Bradbury, 1987).

a) Feeding rate and frequency

Feeding regimes should reflect the feeding habit of the species under wild conditions, but most of recommendations of the feed manufacturers are hypothetical and set irrespective of dietary composition, pond fertilization rate, natural food availability, stocking density and standing crop (Tacon, 1993). Nevertheless, benefits from a greater feeding frequency have been well documented for shrimp, and reported to lead to reductions of leaching and feed loss, and to an improvement of growth and feed efficiency (Villalon, 1991; Sumundra, 1992). Wang *et al.* (1998) have demonstrated that: “Daily feeding patterns also changed in response to feeding frequency, indicating that when fish are fed at a particular frequency, knowledge of feeding pattern is necessary so that appropriate food amounts can be supplied at each provisioning. Based on the results, these authors suggested that an optimal feeding frequency should be determined not only on the basis of growth and food conversion efficiency, but also according to influences of daily feeding pattern and the desire to achieve size uniformity”. Feeding schedules should be designed to account for fish behaviour. The optimum feeding rate and frequency of presentation must, therefore, be determined for individual feeds and farms by carefully monitoring feed consumption, growth and feed efficiency over several growing seasons (Tacon, 1993).

b) Feed presentation

Through many supplementary feeding techniques are being practiced in different parts of the world for example, ranging from manual feeding by putting floating feed items into a floating or fixed-surface bamboo frame to simple demand feeders; in many cases, hard data are lacking on the efficiency of these feeding techniques (Tacon and De Silva, 1997).

Hand feeding allows the operator to better monitor the behaviour of fish and more quickly detect health problems and stress factors. However, hand feeding is a very subjective feeding technology (Paspatis *et al.*, 1999).

Demand or self-feeders have been shown to be useful since the fish themselves accurately control the feeding level by activating a trigger in the water (Alanärä and Brännäs, 1996). Furthermore, each feed can be recorded as a discrete event, providing information about the feeding rhythms of fish that is also of practical interest, as feeding schedules out of phase with natural rhythm can affect growth (Boujard *et al.*, 1995). Demand feeders help to ensure fish eat when they are hungry and can reduce feed wastage. Moreover, fish are able to regulate with a great accuracy their feed intake when fed either by hand or by self-feeders (Boujard and Médale, 1994).

Automated feeders are less expensive for larger operations and can be set to dispense feed more evenly over the entire water surface. However, automatic feeders are not as effective as self-feeders are, although in this method also, many differences are reported between the different studies (Paspatis and Boujard, 1996; Valente and Fauconneau, 2001).

c) Control of effective feeding

There are several factors that may influence the demand feeding activity of a fish like the ability of the fish to regulate such a feeding system, temperature, photoperiod, reward level, dietary energy content, stocking density, and social dominance (Landless, 1976; Boujard and Leatherland, 1992; Alanärä, 1994; Boujard and Médale, 1994; Alanärä and Brännäs, 1996). All of them may thereby influence the ability to optimise growth and feed conversion rates.

On the other hand, some methods such as improvement of video analysis, acoustic trackers, intelligent automated feeding system, etc., have been developed to stop feeding before polluting the environment:

- ❖ Underwater cameras are commonly used for detection of uneaten food pellets on marine farms. Parsonage and Petrell (2003) described in detail the development of the machine-vision system to detect pellets and pellet wastage event. When food wastage is detected, the computer sends a control signal which can be used to trigger an alarm light, sounding device or control feeder.

- ❖ Acoustic device can be used to control feed delivery in fish farm indicating the moment when feed supply should be stopped to avoid wastage (Tsukuda *et al.*, 2000; Mallkeh *et al.*, 2003). The intensity and duration of fish emitted high frequency (> 6 kHz) sounds during the feeding could be related to feeding activity and consequently to fish appetite (Lagardère and Mallkeh, 2000; Lagardère *et al.*, 2004).

The use of the intelligent feeders may be also a good idea for improving the feeding management. Available are different types of intelligent, automated, self feeding system such as BTP (bite-and-pull trigger) that is characterized by low feed waste when reward level is set correctly because it prevents accidental activation by fish, wind or wave. Another available type is restricted demand feeding, which restricts the feeding activity to a few hours per day (Paspatis *et al.*, 2000; Rubio *et al.*, 2004; Noble *et al.*, 2007). The *photoelectric sensor* (added to feeder) can determine the gathering behaviour and the decision to stop feeding can be determined accordingly before the water becomes polluted (Chang *et al.*, 2005). Finally, *infrared sensor* developed by Rubio *et al*, (2003) improves nocturnal feeding (an infrared pellet detector linked to on-line recording with a microcomputer). They suggest that to reduce feed waste, nocturnal feeding should be restricted.

II.2.3 Conclusion

Finally, the management of feed quality and quantity can be an important way to maximize feeding efficiencies and reduce waste production. Even with perfectly formulated feeds, high nutrient effluents will occurs if fish are overfed. The following factors should be considered in optimizing feeding regimes and techniques so as to reduce potential environmental impacts:

- ❖ Adherence to manufacturer's guidelines and feed charts based on fish size and water temperature for recommended feeding rates;
- ❖ Evaluation of different feeder types and feeding techniques.
- ❖ Avoiding the use of mechanical feeders that produce fines, or considering the use of on-site re-pelleting technologies (sieving the pellets through a vibrating screen and then re-pelleting the collected dust and particles);

- Feeding smaller amounts more often to prevent overfeeding;
- Using technologies such as video surveillance or hydro-acoustics that can detect when feed has reached the bottom.

Farming practices, including husbandry, also contributes significantly to the FCR of the different farms (for example, feeding regime; Coloso *et al.*, 2002) and also environmental setting is considered to be important, such as current regime and wind and wave exposure of the farms, although literature values are not easily assessable (Cromey *et al.*, 2002).

II.3 Fisheries

II.3.1 Methods of avoiding capture of unwanted species: selective fishing tackles

Discards represent a significant part of the world marine catches. In recent years, a substantial reduction in discards has been detected. The major reasons are: (a) a reduction in unwanted bycatch and (b) increased utilization of catches.

Bycatch reduction is largely a result of the use of more selective fishing gears, a variety of discard management measures and improved enforcement of them, and also a decline in the fishing effort exerted by some fisheries. Another important factor is that fishery managers, stakeholders, users and in general, all the society, are showing more progressive attitudes towards the need to solve discarding problems in relation to the environment and the fishing resources. The best utilization of the discards is related to the improving food processing technologies and the expanding market opportunities for what it has been consider as “low value” catch. Thus, the protein necessities of the developing countries have resulted in an increased retention of bycatch for human or animal consumption. Reduction in discards is more likely to arise from increased utilization of bycatch, rather than reduction of actual biomass reduction.

Summarising, discards are considered to constitute waste, or infra-use of fishery resources. Large numbers of studies have drawn attention to the need to monitor and reduce discards and unwanted bycatch, in order to assess: (1) the impact of discards on marine resources and environment, and (2) to promote technologies and/or other methods of avoiding, reducing or improving their utilisation.

External methodologies to avoide catch of unwanted species are defined in this section as the regulation or management strategies targeting the specific issue of reducing bycatch. In general, there are major differences between reasons for reducing discards between fisheries but more important between the tropical and temperate fisheries. In general, tropical fisheries are located in coasts and waters of developing countries with high demand for protein (low and high value catch), either for human consumption or animal feed. In social and economic terms, the total commercial biomass extracted may be more important than the specific target species landed. In these cases, the underlying fishery management objective is to maximize the catch, despites the species composition.

On the other hand, the total biomass harvested in the temperate water fisheries is likely to be reduced as a result of the introduction of selectivity gear devices, Bycatch Reduction Devices (BRDs) and other measures. For example, overfishing of whitefish and the higher price of, for instant other species (shrimp) encourages fishers increasingly to target the higher priced ones. From the biological point of view, the complicate predator-prey relationships between crustacean and finfish and the mix or multispecies nature of these fisheries further complicate their management (for example, NAFO area, North Sea, Western Waters…).

In order to reduce the by-catch or incidental fishing of animals, several measures have been initiated. While Melvin Edward (1999) detailed the tools to reduce seabird by-catch (caught in hooks of the liongline nets), Garcia-Caudillo *et al.* (2000) studied the performance and efficiency of Square Mesh/Extended Funnel BRD in the bottom trawl shrimp fishery of the Gulf of California and inferred that the total by-catch was decreased by 40% in comparison to the control trawl. Indeed, BRD are measures implemented to reduce the by-catch from fishing activities. Different types of BRD have already been tested worldwide, for example, in the Gulf of Mexico (see http://www.sefsc.noaa.gov/sedar/download/SEDAR7_DW38.pdf?id=DOCUMENT) or in South-Atlantic
(see: http://www.safmc.net/Portals/6/Library/FMP/Shrimp/ShrimpFMPBycatch.pdf).

Criales-Hernandez *et al.* (2006) studied the impact of by-catch reduction devices on ecosystem through simulation and opined that the results are encouraging both in terms of protection of selected functional groups and in socio-economic terms. Hannah and Jones (2007) reported the reduction of fish by-catch by between 66% and 88% from historical (pre-

BRD) levels in trawl vessels using by-catch reduction devices (BRDs) for fishing for ocean shrimp (*Pandalus jordani*), in California.

Case study: Acoustic Alarm or Pingers (Cox et al., 2007)

Large-scale field experiment with acoustic alarm or pinger, a measure to avoid by-catch of dolphins and porpoises in gill nets, was conducted in 1994 to determine its efficiency in reducing the bycatch of harbor porpoises (*Phocoena phocoena*) in the Gulf of Maine and the by-catch rate was reduced by 92% (Kraus *et al.*, 1997). Further trails in in drift gill-net fisheries in California in 1996 and 1997 indicated 85% reduction in the bycatch of short-beaked common dolphins (*Delphinus delphis*) and California sea lions (*Zalophus californianus* (Barlow and Cameron, 2003). Experiments with pingers in Washington (Gearin *et al*., 2000), the Bay of Fundy (Trippel *et al.*, 1999), the North Sea (Larsen, 1999), and Argentina (Bordino *et al.*, 2002) reported similar reductions in the bycatch (approximately 70–90%) of various small cetaceans.

Subsequently, the International Whaling Commission (IWC) recommended pingers as a means to reduce bycatch of harbor porpoises and other species (IWC, 2001). Furthermore, the European Union is in the process of implementing the use of acoustic alarms in gill-net fisheries (European Union 2004). In the post implementation scenario, though the use of pingers resulted in reduction of by-catch, on occasions their use showed increased by-catch. The possible reason for the increase and subsequent decrease in bycatch may be linked to maintenance of pingers (Caretta *et al.*, 2005).

Case study – sea bird by-catch reduction

Large number of sea birds is killed by gill nets, longlines and trawls. Most attention regarding seabird bycatch has been directed toward longline gear. Longline bycatch of long-lived albatrosses and petrels occurs globally (Lewison *et al.*, 2005). The baited hooks in the long lines attract the seabirds before it sinks. The birds get drowned with the lines (Brothers *et al.*, 1999). Additionally, offal discards coinciding with gear deployment attract birds to the boat leading to even higher numbers of hooked birds (Sullivan, 2004). A suite of measures based on fishing practices and bird behavior to reduce seabird bycatch by longline fisheries has been developed and experimented. These measures include bird-scaring streamer lines, bluedyed bait, line shooting, line weighting, night setting, side setting, and underwater line setting. These measures have been tested in experiments in the North Pacific, Atlantic and Southern oceans (Cox *et al.*, 2007). The available evidence suggests that bycatch avoidance measures have reduced bycatch significantly in both Alaska and South Georgia. But, it is unclear which measures have been most effective as different measures were used.

Internal methods to avoid catch of unwanted species are here defined as (1) tactics and strategies, special manoeuvres to avoid certain species or habitats where they are abundant and (2) the inclusion of technological devices in the fishing gear which is the generalised method for avoiding unwanted species. Gear technology and selectivity devices are briefly enumerated here as a wide range of them are continuously being developed. We selected the most representative gear and selectivity devices introduced in numerous fisheries to reduce discards are:

- Longlines: hook selectivity; there are restrictions on wire traces and minimum lengths of longline gudgeons to reduce unwanted shark bycatch or to increase survival rates. Additionally, some avoidance practices area implemented: night setting; appropriate deck lighting to reduce bird attraction; disposal of offal; use of streamers, weights and line shooters for underwater setting; long lines for sharks using indigenous hooks developed as a low energy resource-specific alternative to energy intensive, and less selective fishing methods such as trawling. Gear systems have improved the capture fishery production from the inland open water resources significantly over the years.
- Biodegradable escape panels in pots (Alaska) to prevent ghost fishing.
- Halibut excluder devices in pot fisheries (Alaska).
- Fish identification by means of electronic devices to avoiding some species in trawls.
- Use of multiple rig trawls to reduce cod bycatch (Denmark).
- Flexible grids built into trawl nets to pass through rollers (approved for Norwegian waters).
- Turtle excluder devices (TEDs) in many industrial shrimp fisheries which appears to have little impact on the level of discards. As an example, penaeid shrimp fisheries (in which TEDs use is mandatory) account for over a 75% of weighted discard rate (range 0–79%).
- BRDs, particularly in the Gulf of Mexico and Australian trawl fisheries and in the Argentine hake and shrimp fisheries.
- Use of square mesh panels in Nephrops fisheries (France).
- Regulation of soak times for gillnets.

As for some of the external measures, gear devices described in the above section are not necessarily the limiting factor in discard and by-catch reduction. Although, the cost of introducing gear modifications are, possibly, the single most important constraint, maybe it is more important to evaluate the real impact of this devices in the fishing resources as some of them may not to have major positive impacts on discard levels.

III. Reducing or converting the waste

III.1 Agriculture

III.1.1 Introduction

After the harvest of crops, such as sugar beet and vegetables, large amounts of N often remain in the soil. This N includes residual soil mineral N and N present in crop residues. Both sources of N may affect groundwater quality through nitrate leaching and air quality through NOx emission. Residual soil mineral N levels after application of the recommended rates of N fertilizer to Brussels sprouts, white cabbage and onions are low to moderate (20–75 kg $N.ha^{-1}$). Application of the recommended rates to other field vegetables, however, may leave large amounts of residual soil mineral N, especially after crops that are harvested before maturing, for example, spinach, where residual soil mineral nitrogen may even exceed a value of 200 kg $N.ha^{-1}$. Obviously, large amounts of nitrate will then be at risk of leaching and denitrification during the subsequent winter. Crop residues of spinach and celeriac contain 25–60 kg $N.ha^{-1}$, cauliflower residues 80–120 kg $N.ha^{-1}$, and white cabbage and Brussels sprout residues as much as 150–250 kg $N.ha^{-1}$. If the residues are decomposed before winter, N from the decomposed plant material may leach or denitrify during the subsequent winter period. If the residues are decomposed after winter, N losses are less likely because the released N can be used by the following crop that starts growing in spring. Realistic estimates of N losses through leaching and denitrification after harvest of field vegetables were generated with a simulation model. It was calculated that leaching losses may exceed 200 kg $N.ha^{-1}$ after spinach or leeks, but denitrification was low. Losses after Brussels sprouts and cabbage were much lower (Neeteson *et. al.*, 1999; Feller and Fink, 2002). To reduce N losses from crop residues it has been suggested to collect the residues and compost it together with amendments that are rich in C. However, this approach can be relatively costly. A promising alternative is to change the crop rotation. After the main crop an additional crop, the so called “catch crop”, is planted to prevent N losses from crop residues and mineralised N in soil.

In addition to environmental measures previously seen, the project “Focus on Nutrients” aims to encourage the introduction of, for instance the growing of catch crops and the establishment of wetlands.

III.1.2 Reduction of N leaching using catch crop cultivation

A catch crop can be under-sown in the main crop, simultaneously with, or just after the sowing of this crop. When the main crop is harvested, the catch crop has an established root system ready to take up N from the soil during late summer and autumn. N that otherwise could have been leached is then taken up and incorporated into plant material. The catch crop is then ploughed-in as late as possible in autumn or in spring. In many studies world-wide, it has been shown that the use of perennial ryegrass (*Lolium perenne* L.) as a catch crop is an effective measure to reduce N leaching in spring cereal crop production (Matinez and Guiraud, 1990; Nygaard Sørensen, 1991; Gladwin and Beckwith, 1992; Thomsen *et al*., 1993; Lewan, 1994; Francis *et al*., 1995; Davies *et al*., 1996; Møller Hansen and Djurhuus, 1997; Shepherd, 1999; Thomsen and Christensen, 1999).

The capacity of a crop to take up N is closely related to the time period available for N uptake (Christian *et al*., 1992; Francis *et al*., 1995). For example, in southernmost part of Sweden, the climate usually permits growth of a catch crop until November, sometimes even longer. In regions with shorter autumns, the use of catch crops is of more restricted value. In a field experiment in west central Sweden the N uptake in ryegrass catch crops during several years was only about half of that found in a field trial in southernmost Sweden (Lindén *et al*., 1999).

The effect of a catch crop on N leaching shows large variation between years and places, depending on differing precipitation and drainage conditions, variations in amounts of N in the soil available for leaching and how successful the establishment of the catch crop was. The use of catch crops has reduced N leaching by 50% or more in several studies (Martinez and Guiraud, 1990; Nygaard Sørensen, 1991; Gladwin and Beckwith, 1992; Thomsen *et al*., 1993; Lewan, 1994; Francis *et al*., 1995; Davies *et al*., 1996; Møller Hansen and Djurhuus, 1997; Shepherd, 1999; Thomsen and Christensen, 1999). A rough estimation for conditions in southern Sweden with spring cereal crops was that postponing tillage (stubble cultivation or mouldboard ploughing) from early autumn until spring and growing undersown perennial ryegrass as a catch crop reduced N leaching, on average by 50-70%. About half to two thirds of the reduced leaching seemed to be due to the N uptake of the catch crop, and the rest to delayed tillage (Aronsson, 2000). In a cropping system with catch crops, the N leaching was measured after fertilizing with pig slurry or mineral N. The reduction of N leaching was similar when normal rates of pig slurry and mineral N were used. When double rates of pig slurry were applied, more N became available than the catch crop could take up. Thereby, the reduction in N leaching was only about 30%, compared with 60% in plots with normal rates of N fertilizer (Torstensson and Aronsson, 1999; Thomsen *et al.*, 1993; Thomsen and Christensen, 1999).

A catch crop preserves in organic form N that otherwise might have been leached or denitrified as inorganic N. Several studies have shown that incorporation of plant material from a catch crop affects N release in soil mainly during the first year and to a minor extent during subsequent years. About 10-40% of organic N in catch crops was estimated to be released during the first year after incorporation (Jensen, 1991; Thomsen, 1993; Thorup-Kristensen, 1994; Thomsen and Christensen, 1999; Aronsson, 2000). The main part of organic N in catch crops contribute to a small long-term accumulation of soil organic N (Aronsson, 2000).

Since 1996, there is financial support in Sweden for reducing plant nutrient losses from agriculture. In order to reduce N leaching during the period from October to March, support is granted for the sowing of catch crops. In year 2005 about 180,000 hectares were sown to catch crops (SJV, 2006).

III.1.3 Agroforestry, buffer strips, wetlands and ponds

Trees, thanks to their deep and extensive root systems are able to recover leaking nitrate from great depth and finally redeposit it as organic litter on the surface, so that any kind of agroforestry approaches can improve or restore the N cycle: planting wind brakes, hedges, etc. can all reduce leakage of surplus nitrate into the ground water, and will also reduce surface runoff of phosphate, nitrate and organic matter into water bodies (Rhoades, 1996; Jackson *et al.*, 1999; Lehmann *et al.*, 1999).

Since 1996, Sweden has various forms of financial support, partly financed by the EU (SJV, 2006), for buffer strips, wetlands and ponds. The purpose of buffers strips is to reduce the erosion of plant nutrients, primarily P, from arable land to water. In Sweden the buffer strip shall be sown with grass and be at least 6 meters wide counted from the watercourse. In 2005 there were 9,000 hectare buffer strips (SJV, 2006). Wetlands and ponds may act as N and P traps, and are important for the reduction of the negative effects associated with plant nutrient leaching. They may also be significant for biodiversity in the landscape. In the end of 2005 about 3,300 hectares of wetlands had been constructed (SJV, 2006).

In the literature, various type of constructed wetland (CW) can be used to treat aquaculture wastewater or to recirculate the water (Lin *et al.*, 2002a,b, 2005; Tilley *et al.*, 2002; Schulz *et al.*, 2003). CW represent a natural treatment system based on biological symbiosis between macrophytes and microorganisms (for example, bacteria, fungi, algae), and their interaction with the soil chemistry (Schulz *et al.*, 2003). Thus, biotic and abiotic processes occur as purification mechanisms in such systems. CW have already largely proved to be a self-organization technology suitable for treating water with relatively low charge of pollutants (such as aquaculture wastewater; Lin *et al.*, 2005). In the "Manual on effluent treatment aquaculture" from AQUAETREAT Project, G (2007): Profitt described the role of natural wetlands up to the CW. In summary, the CW that are of two types (Surface flow and Sub-surface flow) have the capability to retain the suspended solids (SS), and the plants to use the dissolved nutrients reducing nitrogen as NH_3 and phosphate.

The achieving control of water quality through pH, DO, SS, organic matter and N removal and balance (that is, nitrification performance allow regulation of total NH_3-N and nitrite-N concentrations, leading to a possible recirculation of water), make wetland a good candidate in waste reduction and conversion. Efficiency of such media depends mainly on the hydraulic loading rate operated in the wetland compartment, which should not be too rapid in order to let sufficient time for either biotic (that is, microbial mineralization and transformation, and uptake of nutrients by vegetation) and abiotic processes (*that is*, chemical precipitation, sedimentation and substrate absorption) to occur (Lin *et al.*, 2005). While treatment efficiency of physical techniques have been tested in several studies, biological processes have been shown to vary widely according to different parameters such as this hydraulic rate (recirculating ratio), the nutrient input, and the feeding management of the system. In a study involving CW with different residence time of aquaculture effluents, Schulz *et al.* (2003) demonstrated that while the reduction of SS was similar for every flow rate situation (between 95.8% and 97.3%), total N and P removal varied proportionally with the hydraulic retention/residence time (HRT). Indeed, with a decreasing HRT (from 7.5 h corresponding to a 1L flow.min^{-1}; to 1.5 h corresponding to 5 L.min^{-1}), removal rate decrease from 20.6 to 41.8% for N and from 49 to 68.5% for P.

III.1.4 Nutrient removal by algae and bacteria based systems

The increase in intensive animal production during the last decades has resulted in an excess of animal waste. All transformation processes of humid agricultural wastes result in significant amounts of nutrient rich wastewater that have to be treated before its release or reuse (Warburton *et al.*, 2002). One solution to reduce the environmental problems that derive from animal waste N and P is their fixation as biomass and utilization of the resulting product as fertilizer, *that is*, avoiding NH_3 emission. Suitable microorganisms for cultivation in animal waste are algae (microalgae – *Chlorella* or *Scenedesmus* - or cyanobacteria such as *Spirulina*), because they are able to grow photo-autotrophically, and mixotrophically or heterotrophycally in some cases, at low C/N ratio found in the waste. Outdoor ponds and photobioreactors are used for the cultivation of algae, sunlight being the energy source (Fallowfield and Garrett, 1985; Oswald, 1988; Richmond, 1992; Chaumont, 1993). However, in outdoor open systems it is difficult to regulate pH and temperature values and in many regions of the world, climatic conditions do not allow the use of algal ponds.

More research work is needed to develop cost-effective and efficient nutrient removal systems in the case of animal wastewater and other high-strength organic wastewater. The challenge is to combine some basic knowledge related to growth kinetics, low-cost harvesting strategies, and algal strains with unique characteristics, among other issues, in order to develop integrated recycling systems. Furthermore, technical and economical assessments of such systems should prove their viability within a particular environment (Olguín, 2003).

Baumgarten *et al.* (1999) made a study of fixing nitrogen in swine waste as biomass, using the alga *Chlorella* sp. and bacteria naturally living in liquid manure. The algal and bacteria were grown in continuous cultures (undiluted liquid manure) to achieve reduction of NH_4^+ and total organic carbon (TOC) contents. For continuous cultivation, a photobioreactor was used. The results showed that a steady state was not achieved owing to a change in the composition of the bacterial population. NH_4^+ was totally removed, but NO_2^- (up to 100 mM) was transiently released. NO_3^- was not detected. These effects might be explained by the presence of heterotrophic nitrifiers, which are able to oxidize NH_4^+ to NO_2^- and to reduce NO_2^- to gaseous compounds.

III.1.5 Treatment of manure

Manure treatment could be a part of sustainable agriculture, when the circumstances are problematic (Burton and Turner, 2003). For instance, where there is a local surplus of manure relative to available suitable land for spreading, the surpluses is needed to be transported out of the local area or when there are some forms of specific disease risks. The editors Burton and Turner (2003) have together with co-authors made a comprehensive state-of-the-art in treatment strategies for sustainable agriculture in Europe. A range of manure handling techniques are presented that can tackle many of the problems identified and also offer some benefits. These include aeration (for example, for odour, water pollution and air pollution abatement), anaerobic digestion and lagooning (for example, for biogas production, odour abatement, reduction in BOD), separation (for example, for easier handling, reduction in BOD) and composting (for example, for formation of a fertilizer product). In order to reduce NH_3 losses, additives could be used. One option is to acidify liquid manure by adding mineral or organic acids (for example, lactic acid, nitric acid), which has proved to reduce NH_3 emissions from storage (Hörnig *et al.*, 1997). However the high amounts of acid and its high costs remain a drawback. Also, addition of zeolites or peat could reduce NH_3 emissions (Witter and Kirchmann, 1999). For more information, the reader is advised to read Burton and Turner (2003).

One example of biological resource recovery is a 4,000 head US pig farm, which is producing biogas from all its animal waste in a covered lagoon. The nutrient rich wastewater is used among others for production of hydroponic greenhouse tomatoes, whereby the exhaust heat from the biogas generator serves for heating the greenhouse (Hobbs *et al.*, 2002). The average biogas production at Barham Farms is 20 $m^3.hr^{-1}$ in the winter and 40 $m^3.hr^{-1}$ in the summer. The Barham Farms digester produces an average of 560 and 990 kWh of electricity per day in winter and summer respectively. Such a system not only prevents significant water pollution, but also has a strong potential for reducing agricultural greenhouse gas emissions under production of renewable energy.

III.1.6 Reduced losses of pesticides

Pesticides are often found as pollutants in surface and groundwater. An observation from many countries is that point sources of pesticides are one of the most dominating reasons for pesticide pollution of soils, creeks, streams and lakes, groundwater and local water supplies (Torstensson, 2000). This pollution affects of course more directly to the farmers because their own water supplies are being contaminated. In Sweden, a study of how the farmers handled the pesticides was carried out and a type of biobed suitable for the conditions prevailing in Sweden was designed (Torstensson and Castillo, 1997).

Bio-beds at filling place

A biobed is an excavation in the ground about 0.6 m depth, filled with materials that both bind and biologically degrade the pesticides spilled accidentally on the bed while filling the spraying equipment. The size of a biobed depends on what are the conditions at the farm. The material in the biobed is a mixture of organic materials that can bind the pesticides, keep the right humidity and be substrate for growth of the right microorganisms that can degrade the chemicals. In Swedish biobeds, a mixture of straw, mould peat and topsoil is used. A grass layer above the biobed is also used to help to keep the right humidity and also plays pedagogic purpose because the appearance of dead grass spots shows the occurrence of spills. The first biobeds were built in 1993 and nowadays it is estimated that more than one thousand biobeds are used all over Sweden. The fast implementation of this technique has been done through a successful information campaign, the high awareness of the Swedish farmers on keeping a clean environment, and the low cost of the biobed (Torstensson, 2000). The biobed system is so effective and cheap that it has wakened up the attention of many other European countries, i.e., Denmark, Finland, Norway, France and Great Britain. The system is also of interest in developing countries because it does not require too much maintenance or skilful expertise. Biobeds are being introduced now in countries such as Peru and Guatemala (Pers. Comm., Castillo, 2008).

III.1.7 Other industries

a) Olive oil production

Mediterranean countries alone produce 97% of the total olive oil production, while the European Union (EU) countries produce 80-84%. The biggest olive-oil producer is Spain, followed by Italy, Greece and Turkey. Olive mill wastewater (OMW) constitutes a serious environmental problem in olive-oil producing countries. The organic load (COD) of the wastewater is large, as high as 220 g COD.L^{-1} and BOD values of up to 100 g.L^{-1}. OMW contains high concentrations of recalcitrant compounds such as simple phenols, lignin and tannin, which give OMW its characteristic dark color (Niaounakis and Halvadakis, 2004).

The organic load of olive mill waste has been successfully transformed into biogas, under dilution and co-digestion with other organic wastes. In order to maximize fermentation efficiency, removal of phenolic compounds that inhibit microbial activity, is recommended. As those compounds are powerful antioxidants, methods for separation, purification and marketing of these molecules are under investigation (Capasso *et al.*, 1994; Visioli *et al.*, 1995; Visioli *et al.*, 1999; Gizgis *et al.*, 2006; Paraskeva and Diamadopoulos, 2006). More research work is needed to develop cost-effective and efficient agro-industrial waste cyclic system (Fig. 7) in order to develop integrated recycling systems.

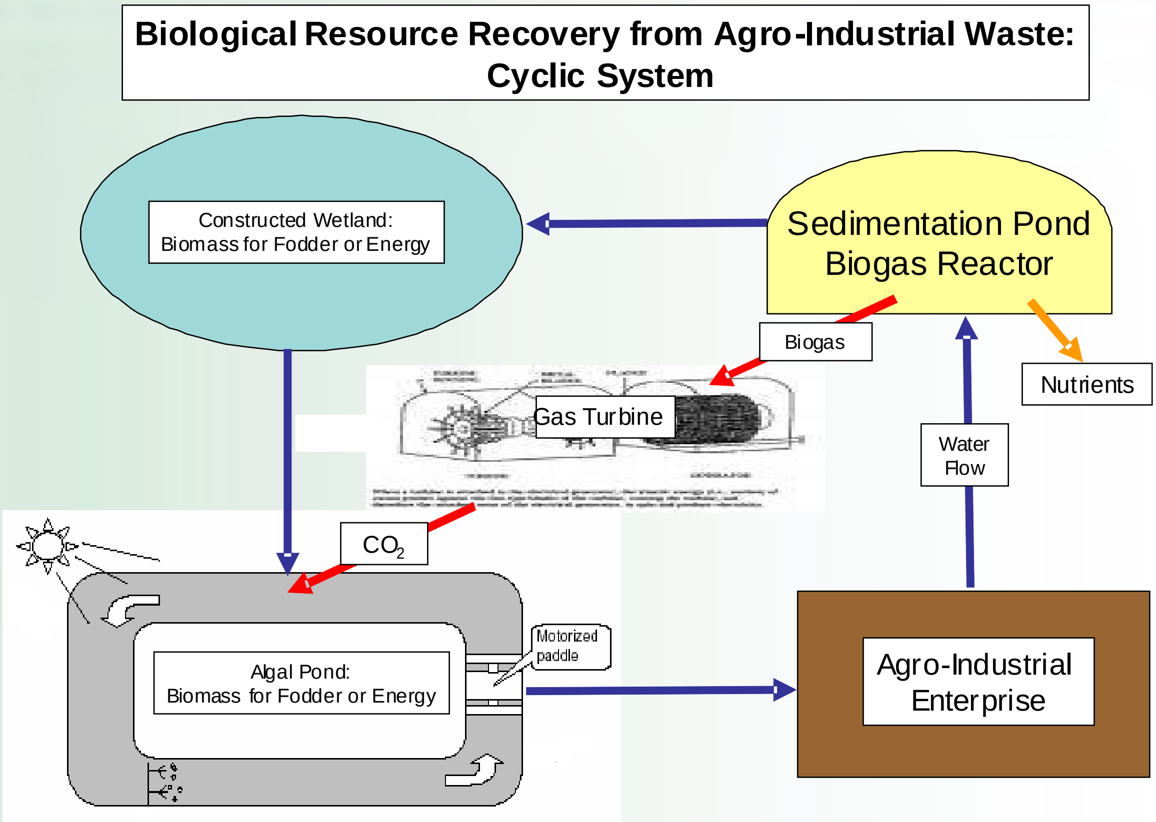


Figure 7. Biological resource recovery from agro-industrial waste (Leu, 2008).

b) Aqua-agriculture based systems

Aquaculture in drylands, for example, Israel and Spain, using low quality ground water, has developed into a successful industry, whereby agricultural and aquacultural approaches are integrated for maximal productivity and profit (Schneider *et al.*, 2005). Due to high evaporation and increasing salinity of the water, recirculating systems are not applicable in drylands. Instead, high solar radiation photosynthetic organisms are the optimal solution for nutrient recovery and biomass production (Oron, 1994).

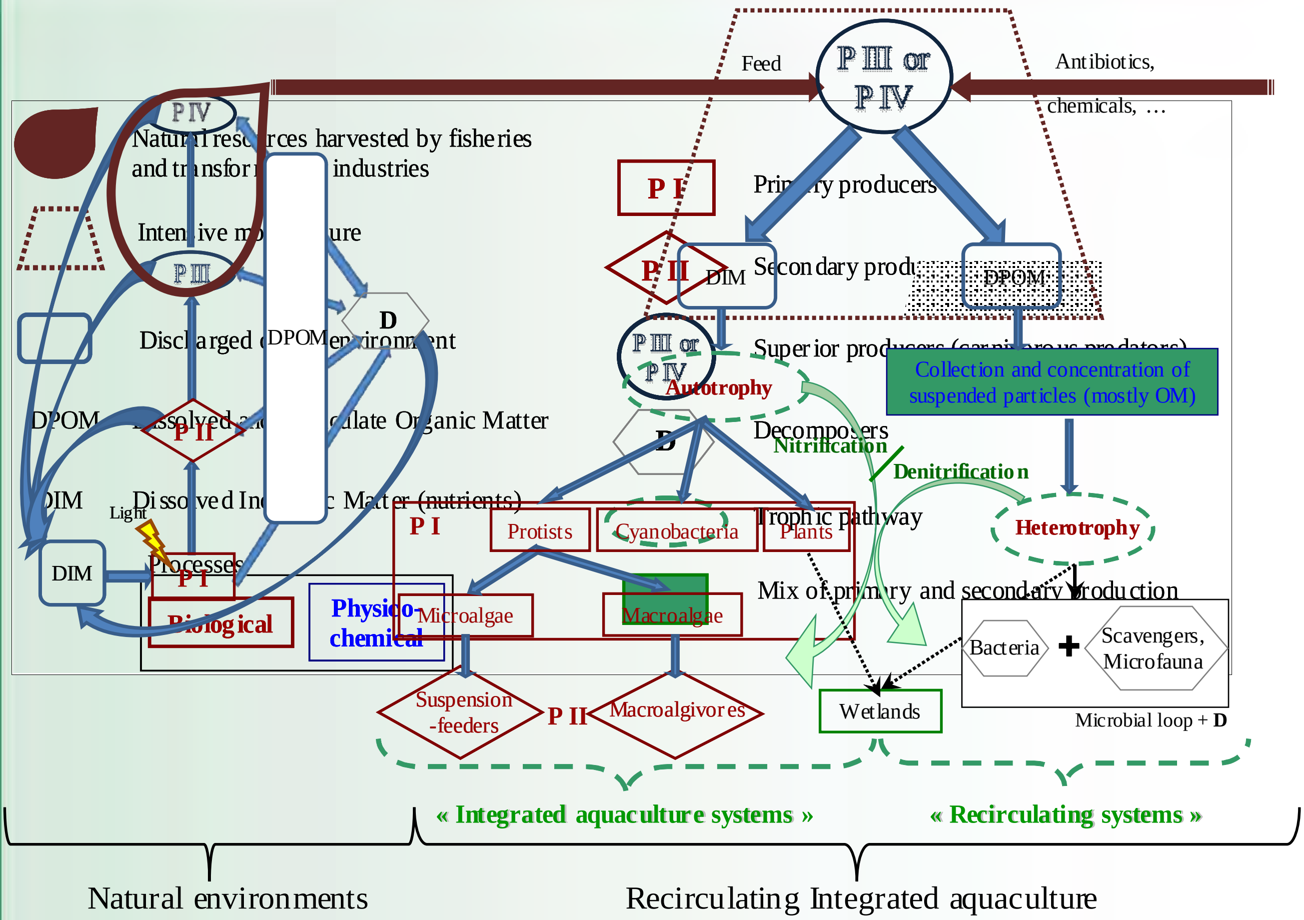
Dependent on local conditions, opposite approaches are used to derive maximal profits from scarce water resources under increasing environmental sustainability. In Australia's Murray-Darling basin, intensive irrigation agriculture leads to raising groundwater levels (Westcot, 1988; Khan *et al.*, 2008). In order to avoid irreversible damage to soils, the groundwater level is lowered by pumping into evaporation ponds, creating ample opportunities for aquaculture enterprises. A wide range of fish, crustaceans and seaweeds are being cultivated in integrated systems before the saline water is finally discharged into evaporation ponds (Flowers and Hutchinson, 2002; Fielder *et al.*, 2001). In this case, aquaculture wastewater is deposited in evaporation ponds, resulting in a net lowering of the ground water table under exploitation of the water for profitable economic activities.

In Israel, the situation in the Negev desert is different. In wide areas of the Negev there is no water table near the soil surface allowing use of marginal waters for irrigation of a variety of dryland crops under appropriate management guidelines (Shuval, 1987; Shalevet, 1994; Oster, 1994). Moderately saline groundwater is thus pumped from great depth and used for irrigation of agricultural crops like olives, dates, Jojoba etc. (Wiesman *et al.*, 2004). The obvious lack of water resources in the Negev induced efforts to maximize water exploitation by integrated aquaculture-agriculture systems for maximizing profits.

III.2 Aquaculture

The commercial success of fish farms in the near future will depend increasingly on their success in either minimizing the production of waste or utilizing waste as inputs to other production processes (Midlen and Redding, 1998). Using the conclusions of the previous parts of this review, it is clear that two main kinds of wastes are produced (*that is,* dissolved nutrients and particulate organic material) by intensive or semi-intensive aquaculture whatever the habitat (fresh or marine water) or the implantation of the system (landbased, ponds, or cages in open systems).

Therefore, when synthesizing the treatment of modern aquaculture waters and the mitigation of the environmental impacts of aquaculture, two main practical approaches are emerging: bacterial dissimilation into gases (due to the microbial loop) and plant assimilation into biomass (Neori *et al.*, 2004; Schneider *et al.*, 2005). This functional scheme mimicks effectively natural ecosystem functioning and the treatment or the valorization of the wastes is, of course, based on ecological processes that are partially or totally under control in aquaculture conditions (for example, temperature and pH surveys). Pure monocultures of fish or shrimp, that is, high producers, are associated with heterotrophs or autotrophs to either control water quality or valorize the waste or both (Fig 8).



Legend:

Figure 8: Ecological approach of future prospectives for a sustainable aquaculture.

To reduce or to convert, particulate and dissolved organic wastes, the heterotrophic metabolisms are favoured, via the microbial loop (*that is*, bacteria and microfauna that feed on bacteria), plus meiofauna and scavengers detrivorous organisms (for example, fungi, worms, sea cucumber, some fish as tilapia, catfish, milkfish, mullet, or crustaceans such as crab and shrimp). Eventually, dead animals or even concentrated sludge can be composted and so on, prior to heterotrophic use in aquaculture context (fish meal) or in agriculture (spreading as ferterlizer or soil amendment). Actually, this part relies on the saprophage trophic web from an ecological point of view.

To reduce or to convert dissolved mineral matter (that is, nutrients such as NH_4^+, NOx, PO_4^{3-}, etc.), autotrophic metabolism is required, *that is*, chimolitho-autrotrophic bacteria involved in nitrification and/or photo-autotrophs (prokaryotes as cyanobacteria, protists as micro- and macroalgae, or higher plants as *Salicornia*). Note that denitrification is a heterotrophic process that converts simultaneously dissolved mineral N and dissolved organic C (Hamlin *et al.*, 2008), thus it is, of interest for the two types of waste.

Combination of these latter processes (heterotrophic and/or autotrophic) is often found in case studies applications such as the usually used terms “recirculating systems” or “integrated aquaculture.”

Chapter III could then be divided into two main categories: physical and biological methods.

Note, some particular compounds of interest could be used from aquaculture waste but would not be presented in detail in this review since they produce a lot of waste themselves (with lower values than the original wastes) which have to be treated anyway or used marginally. These include foams and fats from aquaculture water treatment plant, shells, by-products (such as guts, heads and bones), skin (for example, from salmon), etc.

Note also that to control the wastes, we need also to control the water fluxes, situations that are encountered in continental or marine land based aquaculture. Special mention to marine open cages will be given where appropriate.

III.2.1 Physical treatment

The management of solids, through feed design, feeding management and flow regulation, in addition to the eventual removal of solids using separation and sludge treatment technology, becomes increasingly important as aquaculture systems intensify. For recycle systems, in which reused water must be of an adequate quality to maintain the culture organisms in a healthy and fast-growing condition, it is especially important to remove waste products. A build-up of solids in an aquaculture system can lead to a decline in culture water quality that will increase the stress on the culture organisms (Rosenthal *et al.*, 1982; Klontz *et al.*, 1985; Braaten *et al.*, 1986).

Studies concerning water quality requirements of different species have led to the development of tank designs in which solids transport and separation are important design criteria (Boersen and Westers, 1986; Watten and Johnson, 1990; Wagner, 1993). Various devices have been proposed or used for moving particles, with various efficiencies, including: the juxtapositioning of inflow and outflows; vanes; or screens. There are two primary methods for reducing solids in fish farm effluent:

- clarification – uses gravitational sedimentation and flotation systems of differing complexity

- mechanical filtration – uses filtration meshes dimensioned to trap solids.

a) Pre-treatment systems

The three main advantages to concentrating solids prior to clarification processes are an improved culture environment, increased treatment efficiency and a reduction in required treatment capacity (Cripps and Bergheim, 2000).

Technology used to pre-concentrate solid wastes prior to treatment can be classified into equipment or procedures that produce an intermittent plug, or continuous flow, of high solids content waste. Wastes that have accumulated are scrubbed and then flushed out of the tank by increasing the inflow rate and lowering the water level. Rather than allowing this plug to exit the farm in the main effluent, it may be stored in separate holding facilities to allow treatment processes sufficient time to function at a lower hydraulic load than if treatment of the primary flow was required. A preferential alternative to flushing is the use of a combined concentrator and separate waste solids outlet, as reviewed by Cripps and Bergheim (1995).

A commercially available particle concentrator system consist of a combination of a Norwegian Ecofish tank separator system and a US recycle system (Twarowska *et al.*, 1997). This system combined both a specially designed particle trap that separated excess feed pellets from faecal wastes so that feeding could be more closely monitored, and collectors for sludge and dead fish removal.

Finally, other physical pre-treatment can be achieved by, for example, chemical treatment with flocculant and coagulant, textile bags and also dehydration in special tanks or ponds (by natural or thermal drying).

b) Clarification systems

Clarification or gravitational sedimentation and flotation use the force of gravity to separate particles from a fluid. Difference in density between the particles and the fluid causes the particles to travel downward (sedimentation) or upward (flotation) in a quiescent or slowly moving liquid.

The specific gravity of fish faeces is slightly higher than that of water and therefore the rate of their sedimentation is low. Sedimentation rate depends on the characteristics of the material being settled (including their size), and on the velocity and turbulence of the water in which the particles are suspended. In aquaculture, a favourable settlement speed is considered to be 1 $cm.s^{-1}$. Most unused feed and faeces are separable by sedimentation.

Sedimentation of SS is made more difficult by degradation of the feed or faeces "pellet" as it travels from the fish through the fish-holding area to the sedimentation basin (cf. Quality of the feed, part II). Water turbulence, among other parameters, created by the speed of water flow and the swimming action of the fish, causes the faeces to stay in suspension and progressively abraded and broken down into smaller particle sizes. Very small particles become "non-settling solids". This degradation of faeces into smaller particles, when combined with time exposure in the water, leads to a portion of the nutrients contained in the solids to dissolve (cf. II.2.1.2). Fish farm design should therefore aim to trap and remove SS as early as possible after being deposited by the fish, to reduce this degradation process. Sedimentation is critically linked to the flow rate of water through the sedimentation area.

Sedimentation can be achieved by the following methods and structures:

- ❖ Simple sedimentation using a large area (ponds or basins) with long residence time of water (sedimentation basins may be appropriate for secondary de-watering or thickening)
- ❖ Channels, with or without physical barriers
- ❖ Quiescent zones and trapping of solids within a raceway
- ❖ Lamellar settlement tanks
- ❖ Swirl separation

Additionally, bio-sedimentation could be done by suspension-feeders, which can use detrital organic matter for their energy requirements (Shpigel and Blaylock, 1991; Lefebvre *et al.*, 2001). Floating bead filters (FBFs) are expandable granular filters that display a bioclarification behavior similar to sand filters. They function as a physical filtration device (or clarifier) by removing solids, while simultaneously encouraging the growth of bacteria (both heterotrophic and nitrifying bacteria co-exist in the filter; see the biological processes in the next chapter) that remove dissolved wastes from the water through biofiltration processes (Malone *et al.*, 1993). FBFs capture solids through four identifiable mechanisms, which include straining, settling, interception, and adsorption. With the exception of adsorption, the solids capture mechanisms are physical in nature and are common to all types of granular media filters (Malone and Beecher, 2000). In a single flow through cycle, FBFs can remove completely SS of > 50 µm and remove 40–50% of fine solids (<10 µm). In a multi-pass recirculating mode, bead filters provide complete clarification (Ahmed, 1996).

c) Mechanical fitration systems

Mechanical filters remove solids from water using physical barriers through which the solids particles cannot pass. This is usually achieved with a packed medium such as sand or with a mesh. Mechanical filters will remove both settling solids and those that will not settle due to their small particle size and/or low density.

Before selecting or designing mechanical filters, it is important to know:

- Type of solids to be filtered
- Concentration of solids within the effluent ($mg.L^{-1}$)
- Mesh or filter-media size (µm)
- Flow capacity of the filter ($l.s^{-1}$)
- Energy requirements to operate the filter (head loss in m)
- Pressure filters, gravity filters, drum filters, and disc and belt filters are different options to achieve these tasks.

The efficiency of microsieves at removing solids will depend on the condition in which the solids arrive at the filter. In particular, the degree to which the faeces have been degraded in travelling from the fish to the filter will be critical to the filter performance. Different configurations of tanks, ponds or raceways result in varying self-cleaning properties. It is a guiding principle for filtration efficiency that particles are removed as quickly and with as little turbulence as possible.

Optimal conditions must be achieved for each of the following key efficiency factors in order to achieve the best filtration result:

- Design of fish tanks or ponds
- Mechanical degradation of particles in the system
- Feed quality and feed management
- Surplus feed considerations

Several studies (Liltved, 1988; Liltved and Hansen, 1990; Bergheim *et al.*, 1991; Ulgenes, 1992a,b; Ulgenes and Eikebrokk, 1992; Bergheim *et al.*, 1993a,b; Twarowska *et al.*, 1997) have tested the treatment efficiencies of a commercially available filters (such as Hydrotech drum filter) and microscreens (such as the Unik disc microscreen). Efficiency was found to vary proportionally with the waste effluent concentration. For example, during extensive tests conducted by the Norwegian Hydrotechnical Laboratory, the treatment efficiency of a 60-mm pore size drum screen varied considerably within the ranges SS (67–97%), P (21–86%) and N (4–89%). Twarowska *et al.* (1997), however, achieved lower solids removal rates of 36.5% using the same type of 60-mm pore size screen. Ulgenes (1992b) testing 250- and 120-mm pore screens together achieved a wide range of SS removal efficiencies of 16–94%, whilst Bergheim *et al.* (1991) achieved an average 40% suspended dry matter (SDM) removal using 350- and 60-mm pore size screens.

Clearly, then, the efficiency of such screens is dependent on the characteristics of the effluent and pore size of the screens, and on pre-treatment techniques applied. The low removal efficiency could be attributed to the low particle concentration in the effluent and the large pore sizes of screens employed. These results indicate that screen pore size should be chosen to suit the application and that the choice should be based on the characteristics of the wastewater to be treated. On the other hand, the capacity of a drum screen is proportional to its length and its diameter, while the capacity of a disc screen is limited by the diameter (Wheaton, 1977). In practice however, at high flow rates, such as those in aquaculture applications, several disc or drum units are operated in parallel (Cripps and Bergheim, 2000).

d) Special case: Lift-up technology for open systems

LiftUp® is a technology originally designed to facilitate collection of dead fish in net pens (Ervik *et al*. 1994). LiftUp® consists of a conical pen bottom on which large solids such as dead or moribund fish and large feed pellets are collected and directed to a centralized point (www.liftup.no). Air is injected in a flexible tube located in the center of the machine thereby creating an airlift. Material collected on the bottom is lifted up and out of the pen.

This technology reduces the amount of waste falling to the sea floor and thus either increases the holding capacity of a farm site or reduces wastes at a site that is causing unacceptable benthic impacts. However, there are concerns of resuspending waste feed and faeces, and discharging it to the water column with the potential to degrade water quality (Buryniuk *et al.* 2006; Heinig *et al.* 2006). This is not merely an environmental concern, but also one of fish health; wastes reintroduced into the water column also could result in exposing healthy fish to pathogens associated with dead or moribund fish.

Consistently lower states of organic enrichment have been observed under the LiftUp® cages compared to the non-LiftUp® cages based on both benthic infauna and sediment chemistry results (Heinig *et al.*, 2006). Dissolved and particulate material resulting from LiftUp® operation surface discharge did not raise environmental concerns due to its brevity (<100 seconds), very small area (5m x 10m oval), and intermittent frequency (1-3 times per week); however, surface discharge did raise concern over spread of disease during presence of infection or parasites. LiftUp® type technology offers some potential environmental benefits for specific selection of site-types such as sheltered locations, micro-tidal and low current areas. Use of LiftUp® type technology is, however, not possible under subfreezing conditions due to the risk of physical damage, and in areas shallower than 20 m at low tide due to the physical extension of the LiftUp systems, or at high energy sites due to physical forcing on the system.

e) Concentration of the sludge

Solids captured by the processes already described will be a residual of fine particles in semi-liquid material, known as sludge. Sludge consists mainly of water and for efficient and economic handling, particularly where the material has to be stored or transported, concentration of the sludge to increase the solid content and to reduce the water content, will be required.

Concentration of SS will vary between farms, and at every stage of the effluent stream within one farm, depending on the activity on the site on a particular day. Generally, untreated fish farm effluents contain between 5 $mg.L^{-1}$ and 80 $mg.L^{-1}$ SS. The waste coming from the backwashing of a drum filter can contain around 2000 $mg.L^{-1}$ SS.

Further sludge concentration can be achieved through second filtration and settlement tank based on the same principle as above. The combination of a second filter and a settlement tank can be expected to achieve further concentration of the sludge up to approximately 7% or 70 $g.L^{-1}$ of SS (i.e. 93% water) (Twarowska *et al.*, 1997)

f) Conclusion

Solids control stages such as feed management, pre-treatment, primary separation, secondary solids handling and disposal may comprise an integrated solids management system (Alanara *et al.*, 1994; Summerfelt, 1998). The capital and running costs for effluent treatment that are considered acceptable are dependent on the size of the farm and the potential profit margin (Muir, 1982). In a review of some effluent treatment systems used in European salmonid production, Bergheim *et al.* (1998) found that the treatment costs amounted to 2–10% of the total production costs.

III.2.2 Saprophagous pathways for organic material

Despite applied filtration methods, however, small suspended solids still tend to accumulate in aquaculture systems and have to be removed in order to ensure good water quality. This is also the case for dissolved organic material. Bacteria can be used to remove these dissolved and particulate organic wastes through biofiltration processes (Malone and Beecher, 2000). Additionnaly, two other important processes occur in bacterial filters which are of interest regarding the fate of dissolved inorganic material: nitrification and denitrification (Van Rijn, 1996) (see bacterial filters below).

Moreover, in order to optimize the organic matter treatment, a third factor can be used through incorporating the waste into decomposer biomass. Indeed, bacteria and sludge (after their concentration; even if biological degradation of aquaculture solids has received little attention to date, Van Rijn, 1996) can be fed to microfauna, fungi, worms, sea cucumbers, crustacean and some detrivorous fish (such as tilapia, milkfish, catfish, mullet, …), and thus convert it into detrivorous biomass, which can be removed and sometimes valuable. For example, in a sea bass farms, *Nereis diversicolor* culture can retains nutrients with average value of 0.06% feed N (Schneider *et al.*, 2005).

a) Bacterial filters

Bacterial biofilters are the most common biological treatment system for fish farming effluents. It relies on the metabolic processes of heterotrophic and autotrophic microbial communities to remove dissolved organic compounds from the water and convert dissolved toxic inorganic N (for certain fish) to less toxic forms (Midlen and Redding, 1998). Through a series of oxido-reduction processes, pollutants are broken down into harmless gaseous N_2 and CO_2 by microbial populations and thus allow for an effective and significant aquaculture water recirculation (Gutierrez-Wing and Malone, 2006). However, the technology is not simple since a basic bacterial biofiltration system for fishpond water consists of several devices, high level of mechanization and consumption of external energy, for example, for the cleaning process, pumping and recirculation of biologically active sludge, active aeration, and active backwashing. Recirculating water from a fish pond system with a single nitrifying bacterial filter accumulates nitrate and sludge that need to be disposed of. In some case, using the most advanced technology, the nitrate and organic sludge are treated by an additional anoxic denitrifying bacterial biofilter arrangement. Furthermore, while bacterial biofilter technologies are suitable for relatively small intensive land-based cultures of lucrative organisms, there is no information available as to how such technologies can be integrated into large-scale low-cost fish net pens and semi-intensive shrimp ponds (Neori *et al.*, 2004).

Four important processes involved in biological effluent treatment are:

- Respiration, the conversion of organic material to CO_2 under oxic conditions
- Nitrification or oxidation of ammonia (ammonified and excreted ammonia) to nitrate as an oxygen-demanding process occurred in two steps involving microbial species (for example, *Nitrosomonas sp.* for the nitrite formation step; *Nitrobacter sp*. for the nitrate step). The ammonia removal rate vary according to the filter substrata for the nitrifying bacteria, but is a function of the initial ammonia water concentration (i.e. follows classic Michaelis-Menten kinetics), and it generally varies between 0.25 and 0.5 gNH_4^+-N.m-².day^{-1} (Malone *et al.*, 1993). It is noteworthy that in an experimental aquaculture system consisting of a seabream fish pond and a seaweed biofilter pond, the greatest nitrification activity was associated with bacterial biofilms in the seaweed pond (Dvir *et al.*, 1999).
- Denitrification: Although the bacterial filters are meant to be aerobic, most systems harbour anoxic zones which develop mainly in areas where organic matter accumulates. Anoxic conditions in biofilters result in the development of denitrifying populations. Heterotrophic bacteria, using organic degradation products as C and energy source, with nitrate as an electron acceptor, dissimilate nitrate via nitrite, nitric oxide and nitrous oxide to gaseous elemental nitrogen, which is subsequently released into the atmosphere (Van Rijn, 1996; Schulz *et al.*, 2003). Even thought nitrate does not constitute a very toxic component for the culture, the accumulation of this element can be harmful for some species. Denitrification is then essential in some culture systems to counter-balance effects induced by the nitrification such as acidification of the water, accumulation of nitrate; but this process can be inhibited by limitation of organic matter (Van Rijn and Rivera, 1990). Nevertheless, denitrification can also be an autotrophic process, when inorganic compounds are used by bacteria (for example,

hydrogen, reduced sulfur, manganese, iron) as electron donors to remove NO_3 (Van Rijn *et al.*, 2006). Advantages of such a system are less biofouling and avoidance of C contamination after filtration, but few studies have been done on this process. Denitrifiers are also able to remove excess P in their environment thanks to the metabolic processes under either aerobic or anaerobic conditions (Barak *et al.*, 2003). While nitrification results in an increased C uptake and consequently an increased nutrient uptake, denitrification allows a the reduction of nitrate to N_2 and the oxidation of organic substrate to CO_2 and water.

- Consumption and storage of soluble P, mainly in freshwater treatments.

In biofilters, bacteria attached to the surfaces provided by the filter media, produce a mucous film which supports quite a diverse microbial community including bacteria, fungi, protozoa and larger invertebrates. Purification occurs via a number of processes which include: fine and colloidal solids removal, direct ingestion of trapped or flocculated solids by protozoa and macro-invertebrates and, assimilation and/or transformation of soluble organic compounds and ammonia by microorganisms (Midlen and Redding, 1998). Technically, different basic configurations have been described within this biofilters design and management: trickling filters; submerged filters; rotating media filters; moving bed filters; fluidized bed filters; low-density media filters (Midlen and Redding, 1998; Sindilariu, 2007). The methods are highly effective and can be easily dimensioned (Sindilariu, 2007). Then, the treated wastewaters which were not used for recirculation purposes might be applied to crop- or grassland irrigation or sprinkling (http://www.united-tech.com; Sindilariu, 2007).

b) Others scavengers and detrivorous

Cultivation of invertebrate species (worms, molluscs, echinoderms or crustaceans) with important economic value could also be done on organic matter produced as wastes. Preliminary results related to this application indicate that on-farm reuse of sludge for aquaculture purposes, has a potential to produce high quality live feed (Aquaetreat, 2007).

Ahlgren (1998) placed the sea cucumber *Parastichopus californicus* inside salmon net pens to reduce the fouling of the nets and reported that they grew consuming detritus, thereby reducing the net particulate organic matter (POM) discharge to the environment. However this POM discharge reduction was not quantified in that study.

Nile tilapia and silver carp are reported to selectively retain some particulate matter according to their size, in aquaculture pond water (Turker *et al.*, 2003). While silver carp are more efficient feeding on larger phytoplankton, rarely consuming particles <10µm in diameter, Nile tilapia are known to filter preferably 1µm bacteria and 5 µm diameter phytoplankton.

Porter *et al.* (1996) and Katz *et al.* (2002), in two separate experiments, placed grey mullets in benthic enclosures under a commercial seabream cage farm in the Gulf of Aqaba, and observed their growth (in spite of no additional feed voluntarily introduced in the system). This bioturbation species, by consuming fish farm detritus, improved sediment conditions (reduced organic matter, increased DO, reduced hydrogen sulfides and enhanced the macrofauna community) within the enclosures as compared to control plots. However, given that mullets do not retain appreciable nitrogen from shrimp farm effluent (<5%) (Erler *et al.*, 2004), benefits to effluent treatment by these fish appear limited among certain environments compared with benefits obtained if using other filters. While Erler *et al*. (2004) reported a negative effect on the water quality using grey mullet as secondary crop, the introduction of banana shrimp, periodically grazing bacteria on artificial substrate (which were used to treat shrimp farm effluents) lead to an increased microbial growth rate and better nitrate assimilation (223 ± 54 $mgN.m^{-2}.day^{-1}$ instead of 126 ± 36 $mgN.m^{-2}.day^{-1}$ in the control) in the system.

Milanese *et al.* (2003) have demonstrated that sponges were very good candidate in bacterial biofiltration, with bacterial (*E. coli*) retention varying between 6 and 7.4 x 10^6 $cell.hr^{-1}$ for $1cm^3$ of living sponge (*Chondrilla nucula*), according to the clearance rate and initial bacterial concentration. The production of sponge biomass for commercial extraction of useful metabolites make this secondary crop valuable, and a system designed on fish or shrimp production – bacterial filtration – sponge production can be seen as integrated system.

c) Other treatment or valorisation of solid waste

Fungi

Growth of *Schizochytrium* on waste seawater collected from sludge-thickening processes has been investigated (Aquaetreat, 2007). This organism produces high levels of unsaturated fatty acids from glycerol- or glucose-supplemented wastewaters. Then, sterile effluent seawater from a turbot fish farm, supplemented with glucose or glycerol, has a potential use as a base for fermentation to remove organics and phosphorus, potentially producing high-value products either for fish feed or for other commercial uses such as oils. Such organisms contain about 20% oil within the cell.

Composting

Composting fish waste is a relatively new, practical and an environmentally sound alternative to disposing of fish waste (Carney *et al.*, 2000; Gill, 2000; Irish Sea Fisheries Board, 2003; Miller and Semmens, 2002). Composting is a controlled process in which microorganisms break down organic materials under aerobic conditions. The composted material must contain the correct proportions of N (in seafood wastes) for population growth, and C (from, for example, dry leaves, corn stalks, straw, bark, or sawdust) as food for the microorganisms. By-products rich in N used for composting should thus, be mixed with agriculture waste that are rich in C to achieve equilibrated C/N ratios.

If the raw material is clean and use simple tools on a small scale, composting can generate minor revenue for the farm (Miller and Semmens, 2002). Fish carcasses, which are high in N, should be mixed with a material high in C such as wood chips or sawdust for the production of mulch. Indeed, a few essential elements needed for successful composting are: a moisture content of 40-60%, porosity of 35-50%, pH should be 6.5-8.0, proper temperature phases and duration (mesophilic up to 40°C, and thermophilic 54–65ºC), a C:N ratio of 25-35:1, a particle size of 5-20µm and a proper curing time. Composting requires an oxygen concentration of more than 5%. Generally, if these parameters are maintained and depending on the quality of the input, quality compost can be obtained in two to four months, but the process needs regular attention. Temperature is a key process control factor and should be monitored closely. Pathogens and parasites can be controlled by maintaining the temperature above 55ºC (during the thermophilic phase). Any one of these factors can delay the process and each carbon and nitrogen source has different qualities which can impact the composting process.

Composting technology is commercially available and in use, but its further application is limited mainly by environmental aspects and process economics. For example, composting is an energy-consuming process, requiring 50-75 kWh of electricity per ton of organic waste input and the value of compost as fertilizers is very low, thus balance between benefits and draw backs, mainly sustainability, should be carefully evaluated if composting is to be considered in organic waste management.

Silage

According to the FAO report on aquatic waste treatment (Gill, 2000), fish silage is produced by acidification of fish waste using organic acids such as formic acid which is added at a rate of about 3.5% ($w.w^{-1}$) or mineral acids such as sulphuric acid which is added at slightly lower levels. A third method sometimes used in tropical climates involves the addition of simple sugars such as molasses and a lactic acid bacterial culture which generates lactic acid through the natural breakdown of the sugar. The use of acid is necessary to inhibit spoilage bacteria which could produce off odours, flavours such as trimethylamine or NH_3 and/or toxins such as histamine if left to ferment at neutral pH. Fish and shrimp silage is highly nutritious and is traditionally fed as a protein supplement to swine, mink and poultry. It consists of autolyzed fish offal and is normally manufactured by the addition of fresh fish viscera which contain the necessary enzymes for autolytic breakdown. The liquefied product has a pleasant "malty" odour and is often blended with dry feed ingredients to form a semi-moist diet. Silage has also been used successfully as a low cost ingredient in aquaculture diets (Hole and Oines, 1991). In fact, shrimp silage has been used as a source of pigment as well as nutrition for farmed salmon. Fermented fish silage produced by the addition of lactic acid bacteria and a carbohydrate source has been produced from offal from tilapia, shrimp and salmon and subsequently used in aquaculture diets. One advantage of this process over the traditional organic acid processes is that there is a substantial saving in operating costs provided an inexpensive source of carbohydrate such as molasses. Another potential advantage of using silage rather than meal in aquaculture diets is the fact that most of the silage processes used to date (with a few notable exceptions) do not involve heat denaturation of the proteins. One exception is a process in which silage is produced in the traditional manner and subsequently transported to a thermal processing facility where the silage is heated in a two-stage process to eliminate pathogen transfer. Another exception is the mixing of silage with other dry feed ingredients and then processing by thermoplastic extrusion to produce feed pellets which are heated under pressure and then expand when exiting the extruder producing air voids and thus a lower density. This latter process also results in the evaporation of water which is a requirement for product stability since silage normally contains 65-80% moisture before mixing with dry ingredients (Gill, 2000).

Anaerobic digestion for biogas and heat production

Anaerobic digestion has long been used for the stabilization of different wastewater sludge. It consists of a series of microbiological processes that convert organic compounds to methane, carbon dioxide and simpler organic compounds, in the absence of oxygen (Zehnder 1989) It involves the decomposition of organic waste by bacterial processes with the release of an energy rich biogas and the production of a nutrient rich digestate. Anaerobic processes are of interest in waste management to transform materials (low-value wastes) into simple compounds with an economical value (valuable products).

There is potential for sludge from aquaculture to be used in biogas generation if it can be thickened sufficiently to allow its economic transportation to anaerobic digestion plants (Aquaetreat, 2007; Project FAIR CT 98 9110). However, it is unlikely that sufficient organic matter would be generated on small to medium individual farms to have an efficient on-farm system. Considering freshwater fish sludge, biogas yields higher than 200 $L.kg^{-1}$ organic matter have been measured, which is considered the lower limit for the economical feasibility (Aquaetreat, 2007; Project FAIR CT 98 9110). There is some doubt related to how suitable marine fish sludge would be for biogas generation, since high sodium levels(> 3 g/L) may inhibit generation of methane (Ahring et al., 1991) An alternative would be to use pre-treatment technologies (chemical extraction or precipitation) or codigest with suitable and complimentary waste.

III.2.3 Herbivory pathways for dissolved inorganic nutrient

a) Chimolithotrophs (nitrifying bacteria)

By integrating finfish (or crustacean) culture with phototrophic and/or heterotrophic components, a portion of the effluent nutrients within the culture system may be retained (and eventually harvested). Although they are not one of the harvested crops within integrated systems, bacteria play a major role in nutrient cycling in aquaculture production and settlement ponds, yet the microbial dynamics and interactions and microbial impact on aquaculture system biogeochemistry is still not clear (Schneider *et al.*, 2005). Bacteria in aquaculture systems may be found both in the plankton, as free bacteria, or associated with particles and on surfaces (benthos). Benthic bacteria aggregates in biofilms on net cage and fish pond/tank surfaces and these are involved in organic effluent degradation processes. In order for the bacterial biofilm to continuously process nutrients, its surface must be broken and its thickness regulated by such processes as macrofauna grazing. Erler *et al.* (2004) demonstrated that in shrimp farm effluent treatment ponds, if benthic bacterial biomass is not regulated by scavengers' grazing, then nutrient assimilation is not optimised. It appears that macrofaunal feeding activity can increase both the specific growth rate of bacteria and their nutrient processing activity. More research is needed to further our understanding of aquaculture.

b) Photoautotroph (cyanobacteria, algae and plants)

By photosynthetic processes, plants and algae (either micro-, macro- or cyanobacteria) use the excess of nutrients generated from the fish wastes (particularly C, N and P) and the CO_2produced by fish and bacteria for their growth as food source, to convert it into biomass and oxygen (for their growth). The operation recreates in the culture system a mini-ecosystem, wherein, if properly balanced, plant autotrophy counters fish (or shrimp) and microbial heterotrophy, not only with respect to nutrients but also with respect to oxygen, pH and CO_2. Algal culture or plant culture in general, is a type of “extractive” aquaculture (Chopin *et al.*, 2001), since these organisms add to the assimilative capacity of the environment for nutrients, and lead to a process known as phytoremediation or phytotreatment (Suresh and Ravishankar, 2004; Lu and Li, 2006; Silva Castro *et al.*, 2006). Some algae can also have antibacterial properties, and thus, be more benefit for the main culture (Wang, 2003; Bansemir *et al.*, 2006).

Microalgae

Research on microalgae treatment systems, although not a new approach has been neglected (Wang, 2003) and only a few studies on marine integrated systems exist (Lefebvre *et al.*, 1996; Hussenot *et al.*, 1998). The fishponds behave as hypertrophic systems with high primary productivity, but phytoplankton biomass appears unstable with frequent blooms and crashes (Krom *et al.*, 1989; Lefebvre *et al*., 2004). On diurnal basis, phytoplankton or seaweeds may super-oxygenise the water (by huge photosynthetic activity), which could become toxic for other cultivated species (as fish) if they are reared in the same tank (Erez *et al*., 1990; Gordin *et al.*, 1990; Neori *et al.*, 1996). In such cases, algal density or biomass must be controlled to maintain desirable water quality conditions and to safeguard the cultured species (Lefebvre *et al.*, 2004; Bartoli *et al.*, 2005). Nutrient removal efficiency depends largely on the species used, on the nutrient ratio and the culture regime to treat aquaculture effluents. For example, Borges *et al.* (2005) have demonstrated high ammonia and NO_2-N removal (between 80 and 100%; higher than the range of 60 to 80% obtained with conventional bacterial filters) as well as NO_3 (40 to 100%) and P (20 to 99%) removal from selected diatom species to treat both seabass and turbot effluents. However, using *Tetraselmis suecica* in batch regime resulted in a not very efficient removal of P (maximum removal of 63%) compared with the two other species tested (94-100% of P removal). Then, according to the different removal values of the study (see table 5), the species *Tetraselmis sp.* appeared as the best option to fish effluent purification. The design, construction and operation (for example, dilution rate, silicate addition to rear diatoms, etc.) of microalgal wastewater treatment systems have been influenced by the need for adequate mixing to maintain efficient treatment and by the difficulty of separating the microscopic algal biomass from the treated effluent efficiently and economically to complete the process. Although high levels of treatment may be achieved using these types of microalgal culture (between 74% and 100% of ammonia removal and 97% to 100% P removal for the Diatom *Phaeodactylum tricornutum*; Craggs *et al.*, 1997; Borges *et al*., 2005; see table 5), the coast of technical supports severely restrict their economic use (Craggs *et al*., 1997).

Table 5: Efficiency of microalgae (and filter-feeders) treat aquaculture effluents

References	**System and/or Location (Flow-through experiments)**	**Main cultured species**	**Exctractive and valuable species**	**NH_4-N / TAN-N removal (%)**	**NO_3-N removal (%)**	**NO_2-N removal (%)**	**PO_4-P removal (%)**
Borges *et al.* 2005	Experimental recirculating system (with 75% daily water re-use rate)	Seabass (Dicentrarchus labrax) with a stocking density of 3kg/m^3.	Prasinophyta (*Tetraselmis suecica*)	98.20	90.91	69.71	20.57
	Private Portuguese commercial flow-through farm	Turbot (Scophthalmus maximus) with a stocking density of 20-30kg/m^3.	Prasinophyta (*Tetraselmis suecica*)	85.55 - 92.89	40.91 - 90.91		51.85 - 62.96
			Prasinophyta (*Tetraselmis sp.*)	100	66.61 - 100		94 - 97.47
			Bacillariophyta (*Phaeodactylum tricornutum*)	97.47 - 99.13	42.20 - 91.82		100
	Private Portuguese commercial recirculating/tidal dependent farm	Turbot (Scophthalmus maximus) with a stocking density of 20-30kg/m^3.	Prasinophyta (*Tetraselmis sp.*)	81.61 - 89.04	100 %	97.84 - 98.81	97.24 - 99.36
			Bacillariophyta (*Phaeodactylum tricornutum*)	73.59 - 96.87	96.48 - 100	97.74 - 98.75	97.23 - 99.38
Brune *et al.* 2003		Fish (*Ictalurus punctatus*)	Microalgae (*Scenedesmus* and other green algae) / Bacteria / Fish (*Oreochromis niloticus*)	38* of N retention by algal production			~30*
Craggs *et al.* 1996 not in Ref list	Flow-through land-based raceways	Seawater diluted with primary sewage effluent (1:1)	Cyanobacterium (*Oscillatoria sp.*)	100±0.4		51.7±10.7	99.4±0.8
			Diatom (*Phaeodactylum tricornutum*)	100±0.2		82.9±7.1	100±0.3
Craggs *et al.* 1997	Outdoor continuous culture	Oysters or clams	Phytoplankton *Skeletonema costatum*				
Hussenot *et al.* 1998	Land based fish farm(with lagoon and wetland)	Fish farm of turbot (*Psetta maxima*) or seabass (*Dicentrarchus*	Microalgae (diatoms *S. costatum*) and Oysters (*Crassostrea*	67 of DIN removal			46.60

		labrax)	*gigas*) [Maximum removal of the microalgae by oysters: between 92 and 100%]		
Wang 2003	Land-based recirculating system	Shrimp (*Penaeus vannamei*)	Microalgae (*Chaetoceros sp.*) and Oysters (*Crassostrea virginica*) [Conversion Oyster:Microalgae = 2:1 (1kg shrimp ⇒ 0,8 kg algae ⇒ 0,4 kg oyster in dry weight)]	~50* of DIN removal	~53*
Bardach 1986		Shrimp	Tilapia (grown in cages) and oysters		
Hopkins *et al.* 1993	Ponds in South Carolina	Shrimp	Clams and Oysters	Minimal decrease, non significant compared to monocultures, but production remained high	
Jones and Iwama 1991		Fish (Chinook salmon: *Oncorhynchus tshawytscha*)	Oyster (*Crassostrea gigas*)		
Lefebvre *et al.* 2004	Land-based fish farm/Outdoor continuous culture (concrete tanks)	Fish (Seabass)	Microalgae (phytoplankton population dominated by *S. costatum*)	Between 19 and 29 of total N (~100% of TAN)	64 to 89 % of total P
Osorio *et al.* 1993		Shrimp	Oysters		
Qian *et al.* 2001	Lab experiments (tanks)	Fish (mangrove snapper *Lutjanus russeli* and sea perch *Abudefduf septemfasciatus*)	Molluscs (abalone *Haliotis diversicolor*; scallop *Chlamys noblis*; green mussel *Perna viridis*)		
Stirling and Okumus, 1995	Open water environement (typical fjordic sea loch system)	Fish (salmon cages)	Mussels (*Mytilus edulis*) [Mussel growth rate significantly (2 to 5x) higher]		
Taylor *et al.* 1992		Fish (salmon)	Mussels		
Troell and Norberg, 1998	Coastal open waters	Fish cage	Mussels (25g with a density of 0.5 individuals.L^{-1})	between 60 and 80% of the seston removed according to the initial concentration (4 or 8 mg.L^{-1})	
Whitmarsh *et al.* 2006	Coastal open-water (Europe)	Fish (salmon)	Mussels		

* Estimation from Schneider *et al.*, 2005

If easily concentrated, microalgal biomass can constitute a saleable product. For example, the cyanobacteria *Spirulina* is largely used as nutritional additive because of its high protein content (Becker, 2006).

Macroalgae

The macroalgae serve as efficient nutrient “traps” in integrated aquaculture systems (Tables 6, 7 and 8). However, the appropriate choice of the “extractive species” depends on its physiology, the aim of the culture (biomass production or phytotreatment) and environmental parameters or culture conditions such as stocking density, depth, light intensity, water temperature, water residence time and nutrient concentrations on growth (*that is*, turnover rate) and nutrient removal efficiency (Gallant, 1993; Chopin *et al.*, 1996, 1999; Buschmann *et al.*, 2001; Porrello *et al.*, 2003a, b; Pereira *et al.*, 2006; Wang *et al.*, 2007). In intensive fish aquaculture, where most of the nitrogen is released as ammonium (toxic to fish), *Ulva* and *Porphyra* have highly efficient uptake and assimilation rates (Harlin *et al.*, 1978; Neori *et al.*, 1996; Carmona *et al.*, 2001), but these species impact natural nitrification processes in the culture systems, since these processes vary with the pH which decrease with photosynthetic CO_2 uptake (Krom *et al.*, 1995).

Compared to microalgae, the chemical composition and biomass of cultured seaweed are easier to quantify and control (Neori *et al.*, 1998). But recently, it has been observed that yield, growth and N uptake efficiency of different seaweed species were highly seasonal and subject to crash of biomass as well as to microalgal production (Troell *et al.*, 1999; Evans and Langdon, 2000; Matos *et al.*, 2006). In order to optimize the overall biofiltration efficiency, a compromise between apparently conflicting aims such as water flow, biomass production, nutrient uptake or reduction efficiency, that is to say quality and quantity, is necessary (Chopin *et al.*, 2001; Troell *et al.*, 2003; Hernandez *et al.*, 2006). While high nutrient uptake rates are achieved by supplying seaweed culture with high nutrient concentrations (which also maximise algal protein content), the highest nutrient removal rates are achieved when seaweeds are nutrient-starved, thus exhibiting low protein content (Buschmann *et al.*, 1994). Thus, biomass value of the algae is optimized at higher nutrient concentrations, which means high availability but weak relative filtration. Thefore, to solve this conflict, a compromise has to be done, taking several species in order to achieve both optimisations of treatment and of saleable biomass (*that is*, inspired from natural phytoplanktonic species successions according to the environmental condition).

Macroalgal cultivation in integrated systems requires careful attention to some management issues. In ponds or tanks with high algal biomass, oxygen levels often exceed saturation during daylight hours (especially in conditions of high irradiance levels) due to intensive photosynthetic activity and this may lead to severe physiological stress among the cultivated species such as fish suffering from "gas-bubble disease." Since the experiment of Bartoli *et al.* (2005) was not very conclusive, reporting a low N uptake by *Ulva* (see table 6) because of an excessive biomass (dry weight between 200 and 500 g.m^{-2} of a mat), a model was used to determine the real potential of *Ulva* treatment (N uptake potential) if the biomass of the algae is regularly harvested. The model reported that *Ulva* could reduce the dissolved inorganic N (DIN) loading from the fish farm by about 50%. The lack of management as, for example, a periodic harvesting of the surplus biomass (excess of biomass limits light penetration and alters water microcirculation), results in a collapse of the production, setting to zero the phytotreatment action (Bartoli *et al.*, 2005).

Table 6: Efficiency of macroalgae in treating aquaculture effluents.

References	System and/or Location	Main cultured species	Exctractive and valuable species	NH_4-N / TAN-N removal (%)	PO_4-P removal (%)
Buschmann *et al.* 1994, 1996; Medina *et al.* 1993	Tank (Chile)	Fish (Pacific salmon: *Oncorhynchus kisutch* and Rainbow trout: *O. mykiss*)	*Gracilaria chilensis*	70 - 95	
Chow *et al.* 2001	Tank	Fish	Oyster / Sea urchins / *Gracilaria*	100	
Enander and Hasselstrom 1994		Shrimp (*Penaeus monodon*)	*Gracilaria sp.* and Mussels (*Mytilus edulis*)	72% for total N (81% TAN and 19% NO_3^-)	83% (phosphate) and 61(total P)
Haglung and Pedersen 1993	Tank	Fish (trout)	*Gracilaria*	~90	Maintained at low level
Harlin *et al.* 1978	Aquaria	Fish	*Gracilaria* and *Ulva*	32 -112	
Jimenez del Rio *et al.* 1996	Tank	Fish (seabream)	*Ulva*	19 - 97	
Jones *et al.* 2001	Lab study	Shrimp effluent	Oyster / *Gracilaria*	2 - 76	
Kang *et al.* 2007	Laboratory experiments	Initial ammonium concentration (150 - 300 µM)	*Codium fragile*	86.3 ±2.1 - 99.5±2.6 (according to the irradiance, temperature and initial ammonia concentration)	
Krom *et al.* 1995	Land-based fish farm (Tank)	Fish (seabream)	*Ulva lactuca*	17 – 39 % of TAN (5 % of denitrification by bacteria) and 34 – 49 % of DIN	9 - 21
Langton *et al.* 1977	Tank	Tapes	*Hypnea*	70	
Pagand *et al.* 2000	Tank / Raceway	Fish (seabream)	*Ulva*	30 - 90	

Porello *et al.* 2003; Bartoli *et al.* 2005	Land-based fish farm (Tanks/Ponds)	Seabream (*Sparus aurata)* and seabass (*Dicentrarchus labrax*)	*Ulva rigida*	~5 – 36 % of DIN [Denitrification ~1.5 % by algae and ~4.9% by associated bacteria]	15
Schneider *et al.* 2005		Fish (*Oreochromis niloticus*)	*Lemna minor*	57 % of N	
Troell *et al.* 1997	Open water cage cultures	Fish (salmon)	*Gracilaria*	5 - 6.5	27 % of dissolved P
Troell *et al.* 1999	Tank and open sea-ropes	Fish (Salmon)	*Gracilaria* [40% higher growth rate in co-culture than monoculture]	50 (in winter) to 95% (in spring)	
Vandermeulen and Gordin 1990	Tank	Fish (seabream)	*Ulva*	85	
Wang *et al.* 2007	Land-based recirculating tanks	Sea cucumber (*Apostichopus japonicus*)	*Ulva pertusa*	68 ($0.459 gN.m^{-2}.day^{-1}$)	26

Table 7: Studies related to the EU project SEAPURA (2001-2004), with N and P removal efficiency of each experiment.

References	System and/or Location	Main cultured species	Exctractive and valuable species	NH_4-N removal / TAN-N removal (%)	PO_4-P removal (%)
Martinez-Aragon *et al*. 2002 (Phosphate); Hernandez *et al*. 2002 (Ammonium)	Land-based fish farm (Tanks)	Seabass (*Dicentrarchus labrax*)	Macroalgae (*Ulva rotundata*)	80.1- 88.2 (according to water flow and physiologic condition of algae before treatment)	60.7 - 96.2 (according to water flow and physiologic condition of algae before treatment)
			Macroalgae (*Enteromorpha intestinalis*)	81.1 - 99.9 (according to water flow and physiologic condition of algae before treatment)	85.3 - 99.6 (according to water flow and physiologic condition of algae before treatment)
			Macroalgae (*Gracilaria "gracilis"* → *longissima*)	61 - 90.3 (according to water flow and physiologic condition of algae before treatment)	71.4 - 98.0 (according to water flow and physiologic condition of algae before treatment)
Hernandez *et al*. 2006	Land-based fish farm (outdoor tanks) MICRO-SCALE	Fish (*Sparus aurata*)	Macroalgae (*Gracilaria longissima*)	62.20	93.20
				19.1±4.07 (with 44 % of ammonia uptake oxidized by nitrifying bacteria)	3.20
	Land-based fish farm (outdoor tanks) MESO-SCALE			56 - 176	3 - 25
	Land-based fish farm (outdoor tanks) MACRO-SCALE				
Bansemir *et al*. 2006	Land-based facilities experiment	Different fish pathogenic bacteria	26 Species of seaweed tested for their antibacterial activity		
Carmona *et al*. 2006	Land-based facilities experiment	Fish	Macroalgae (different species of *Porphyra*)	70 - 100	35 - 91
Kraemer *et al*. 2004	Land-based fish farm	Fish	Macroalgae (different species of *Porphyra*)	55- 90 according to irradiance, temperature and species	
Mata *et al*. 2006; Schuenhoff *et al*. 2006	Land-based facilities experiment	Fish (*Sparus aurata*)	Macroalgae (*Asparagopsis armata*)	Different TAN removal according to light and temperature: 14.5 $gTAN.m^{-2}.day^{-1}$ (double than *Ulva*)	

References	System and/or Location	Main cultured species	Exctractive and valuable species	NH_4-N removal / TAN-N removal (%)	PO_4-P removal (%)
Matos *et al*. 2006	Land-based fish farm	Turbot (*Scophthalmus maximus*) and seabass (*Dicentrarchus labrax*) farm	Macroalgae (Palmaria palmata)	41±17.26	
			Macroalgae (Gracilaria bursa pastoris)	63.8±24.62 % and 76.7±22.13 % according to stocking density and water flux	
			Macroalgae (Chondrus crispus)	41.3±17.32	
			Cascade of the 3 species	83.50	
Metaxa *et al*. 2006	Different land-based facilities experiment (RAS/Algal pond)			25	9
Pereira *et al*. 2006			Macroalgae (*Porphyra dioica*)	Maximum N removal = 150 $\mu mol.m^{-2}.s^{-1}$	
Schuenhoff *et al*. 2003	Semi-recirculating tanks	Fish (*Sparus aurata*)	Macroalgae (3-stage *Ulva lactuca*), Abalones and sea urchins	30 % of water nutrient	

Table 8: Studies from an Israeli pilot marine farm *SeaOr*, with N removal efficiency of each experiment.

References	System and/or Location	Main cultured species	Exctractive and valuable species	POM removal (%)	NH_4-N / TAN-N removal (%)
Cohen and Neori 1991	Tank	Fish (seabream)	Macroalgae (*Ulva*)		90
Shpigel and Blaylock 1991		Fish	Microalgae (phytoplankton) and Oysters (*Crassostrea gigas*)	34 – 40 % of oyster filtration (with flow rate = 2-4 pond volumes.hr^{-1})	
Neori *et al.* 1991	Tank	Fish (seabream)	Macroalgae (*Ulva*)		39 - 96
Shpigel *et al.* 1993; Neori *et al.* 2004	Land-based pilot-farm	Fish (*Sparus aurata*)	Macroalga (*Ulva lactuca*) and bivalves (*Crassostrea gigas* and *Tapes semidecussatus*)	3 kg feed → 1kg fish → 3kg bivalve → 7.8kg seaweed	
Neori *et al.* 1996	Land-based fish farm (Intensive multi-tank recirculated mariculture)	Fish (*Sparus aurata*)	Macroalgae (*Uva lactuca*)		50 – 90 according to season (always less than 1.8 mgN/L) and 39 – 49 % of DIN
Shpigel *et al.* 1996	Land-based facilities with open-water system at the NCM (UK)	Fish	Macroalgae and Abalones		
Shpigel and Neori 1996	Land-based facilities with open-water system at the NCM (UK)	Fish (*Sparus aurata*)	Clam (*Tapes philippinarum*), Macroalgae (*Ulva lactuca* and *Gracilaria spp.*) and Abalones (*Haliotis tuberculata*)	2 kg feed → 1kg fish (growth rate GR = 0.5 %.day^{-1} and stocking density SD = 20 kg.m^{-2}) → 2kg clam (GR=0.5 %.day^{-1} and SD = 10 kg.m^{-2}) 25 kg fresh seaweed (GR = 0.25 kg.m^{-2}.day^{-1}) → 1 kg abalone (GR = 0.3 %.day^{-1} and SD = 35 kg.m^{-3})	~90 (55 % from seaweed)
Neori *et al.* 1998	Land-based facilities with open-water system at the NCM (UK)	Abalone (*Haliotis tuberculata*)	Macroalgae (*Uva lactuca*)	7 – 29 (average of 14 ±8 %) of N removal by abalone growth (~2918 mgN.tank^{-1}); Mean ingestion rate of 60 %	58 - 100 of N inputs / < 25 % P removal
			Macroalgae (*Gracilaria conferta*)		0 – 42 of N inputs
Neori *et al.* 2000, 2003	Land-based facilities with open-water system at the NCM (UK)	Fish (*Sparus aurata*)	Macroalgae (*Uva lactuca* and *Gracilaria conferta*) and Macroalgivores (Abalones: *Haliotis discus hannai*)	46 % of seaweed transferred to the abalone	~80 – 90 [85 - 90% (up to 2.9 gN.m^{-2}.day^{-1}) in a 3-stage Ulva tanks]

c) Filter-feeders and macroalgivores

When a new plant can add an economic value to the culture, one can say the system lies under integrated aquaculture. Then, the new plant biomass can be directly saleable (dulse, spirulina, salicornia, etc.), or may be used to nourish various species groups: shellfish, including bivalves for microalgae, macroalgivores such as abalone and sea urchins for macroalgae, and even detrivorous to close the trophic chain. Indeed, various studies have shown that several species of seaweed and microalgae have high nutrient filtering capacities (see tables 5-8) and thus can be viewed as biological nutrient scrubbers (Chopin *et al.* 1999, 2001) with economic-value (Troell *et al.* 1997) as human food (for example, Nori, Wakame, Kombu), or food sources for other valuable aquaculture organisms (Wikfors and Ohno 2001).

The growth of marine bivalves is mainly affected by food availability (for example, ambient seston concentration) and water temperature, which are largely a function of season (Troell and Norberg, 1998) . In addition to somatic growth, the efficiency of bivalve nutrient uptake and assimilation (and therefore ability of bivalves to serve as good biofilters under fish farms) is greater during the growing season (Stirling and Okumus 1995). It has been suggested that increased availability of soluble nitrogen close to salmon farms might stimulate phytoplankton growth (Jones and Iwama 1991), where benthic light levels are sufficient, thereby enhancing bivalve biofiltration rates. However, even if phytoplankton abundances are seasonally low, there is strong evidence that filter-feeding bivalves may utilize non-phytoplanktonic food sources to meet part of their energy requirements (Langdon and Newell, 1990; Riera and Richard, 1996; Lefebvre *et al.*, 2000; Marín-Leal *et al.*, 2008). Thus filter-feeders may contribute to the direct reduction of suspended organic matter released by the main cultivated species (60 - 100 % of POM removal by oysters; Hussenot *et al.*, 1998; Troell and Norberg, 1998; see table 5). Borges *et al.* (2005) stipulated that a contribution to the fish-farmer revenues could be obtained using microalgae as food source for the clam *Tapes decussatus*, diminishing the estimated 30 to 40 % seed rearing costs attributed to microalgal food production. Utilization of bivalve molluscs as biofilters for microalgae produced on wastewater reuse system appears as a very profitable option (Shpigel *et al.* 1991; Hussenot *et al.* 1998; Borges *et al.* 2005). For example, in an integrated shrimp/oyster production system, the excessive nutrient from shrimp feed is used to produce a crop of marine diatoms. The continuous harvesting by the oysters is a key factor in maintaining the diatom production (Wang, 2003). However, it is difficult to quantify this microscopic biomass and thus, the efficiency of this system is not clear.

Studies on macroalgae have traditionally focused mainly on their phytotreatment potential and have not given sufficient attention to the market potential and the value of the algae as an additional crop. It is noteworthy that even in cases where the biofiltration efficiency of the macroalgae is not very high and their market value low, the seaweeds may be used to feed other valuable species such as abalone or sea urchins (Schuenhoff *et al.* 2003). Kelly *et al.* (1998) have studied the feasibility of rearing sea urchins of high commercial value, adjacent to salmon net pens, as one means to capture fish farm wastes accumulated in macroalgae biomass and to convert it into valuable marketable biomass.

However, all of the modules that comprise integrated systems (Fig. 8) have specific limitations that are related to nutrient uptake dynamics, nutrient preference, unwanted conversion processes and abiotic factors (Schneider *et al.* 2005). Integration of species demands a constant, reliable and efficient production of algal cells of known size and composition, implying a steady state with the fish culture, in order to continuously nourish the filter-feeder secondary crop. The concept of “steady state functioning” is very important in the design and operation of integrated production systems, as it is necessary that the demands for resources by both the main cultivated species (fish or shrimp) and the second marketable organism (filter-feeders or macroalgivores) remain constant and in balance (Wang, 2003; Borges *et al.*, 2005).

III.2.4 Integration of the biological components

a) Introduction

Various degrees of integration of saprophagous and herbivorous are reported in literature under names such as recirculating systems, integrated aquaculture systems, integrated agriculture-aquaculture systems, and contructed wetlands. After a short clarification of these systems, their concepts are further presented and developed.

Recirculation systems (also called recirculating aquaculture systems: RAS) are partially closed systems employed in inland aquaculture production, where the effluent water from the system is partially treated and recirculated to enable its re-use. In general, there is often confusion about what is meant by recycling and what is meant by re-use. Recycling is not a simple re-use of the water but is rather when the water quality is under control of the water treatment system. The control of the water quality in the system could be done by saprophagous and/or herbivorous but practically RAS are mostly based on the use of bacteria biofilters and physical processes (Losordo and Hobbs, 2000).

Integrated aquaculture (or the most recent term integrated multi-trophic aquaculture IMTA) is defined as "when an output from one subsystem, which otherwise may have been wasted, becomes an input to another subsystem resulting in a greater efficiency of output of desired products from the water area under a farmer's control" (Neori, 2007). Thanks to different trophic pathways (*that is,* autotrophy and heterotrophy) present in such systems, mutual benefits are achieved ecologically (Lefebvre *et al.*, 2004). In reality, successful integrated cultures are a managed by imitating natural ecosystems (Neori *et al.*, 2004) or a coastal food production system based on existing ecological principles (Brzeski and Newkrik, 1997). Here, the solution to pollution is the conversion (*that is*, wastes recycling into biomass; Folke and Kautsky, 1992; Neori *et al.*, 2000) rather than dilution of wastes, making eutrophication beneficial by controlling it (Shpigel and Neori, 1996). Integrated production systems generally originate as monoculture systems of high trophic level producers (PIII or PIV, usually high-valued finfish or shrimp culture; see fig. 8 and tables 5-8) that requires feeding. Primary producers (PI: microalgae, macroalgae, plants) are then employed to remove dissolved inorganic nutrients such as N and P (by means of photosynthetic pathways) and filter-feeders (PII: mussels, oysters, clams, etc. for POM and microalgae), or macroalgivores (PII: abalone, sea urchins, etc. for macroalgae) may be used to remove the suspended organic particles (and possibly to regulate primary producer biomass). Finally, detritus feeders (D: seacucumbers, polychaetes, mullets, etc.) can, in few cases, remove organic matter accumulated by biodeposition (faeces and pseudofaeces). The practice of ecological engineering as a tool to reduce effluents have now gained new interest and many suggestions for integrated cultivation system, using different combinations of seaweeds, bivalves, fish and shrimps have been proposed.

There is apparently a significant overlap between the terms "recirculating systems" and "integrated aquaculture". A right definition of integrated aquaculture is when, beyond monoculture, several organisms are produced while production of each of them is optimized. This special kind of polyculture in chain aims at reducing environmental impacts of aquaculture activity (*that is,* effluent problem, at least, among the ecological footprint; McKindsey *et al.* 2006) and increasing the commercial value of the system (by economic diversification with the production of other value-added crops). This adds an additional constraint, economic by nature, since cultivated species are being restricted to species of high commercial value (Buschmann *et al.* 1996). In a review focusing on the biological processes underlying treatments used in recirculating systems, Van Rijn (1996) has proposed the development of a multi-component food chain system in order to mitigate negative effects occurring in RAS, *that is,* the accumulation of inorganic nutrient which are not reduced by bacterial filters alone. Van Rijn (1996) also proposed that RAS achieve only partial water purification and an integration of hydroponics or macroalgae are examined (regarding nitrogen cycle) to keep advantages of RAS (water recycling, high production, etc.) and further ameliorate the system.

Consequently, RAS including biofilters do not fit within integrated aquaculture but integrated aquaculture made in an optimal way could fit in recirculating systems. Such systems called recirculating integrated aquaculture systems (also called partitioned aquaculture (for example, Drapcho *et al.*, 2003) are very few at the moment or theoretical.

Finally, integration could also be made between aquaculture and agriculture leading also to valuable products. However, processes are less under control with many black boxes in the system (see paragraph III.1.7 of the present study).

b) Recirculating aquaculture using biofilters

Treatment based on solids removal and nitrification is probably the most widely used recirculating systems. Whereas many of the principles of this technology are ancient (the Chinese have been practicing various forms of recirculating aquaculture for numerous centuries), RAS have only recently become recognized in “western” aquaculture. Indeed, one of the fastest growing technologies in freshwater aquaculture is the RAS. In 2000, the International Journal of Recirculating Aquaculture was established to address the growing needs of this sector.

In recycling systems, ammonia is transferred into nitrate and normally less than 1% of the water has to be replaced with new water per cycle. Moderate recycling only requires mechanical filtration, CO_2 removal, oxygenation and biofilter for degradation of dissolved organic matter and nitrification, which means that ammonia is transformed into nitrate, and is then diluted out of the system. High levels of recirculation will require a denitrification filter and potentially a filter for P as well. If a denitrification filter is included, a recycling rate with less than 0.1% new water can be obtained (Twarowska *et al.*, 1997).

Biological filter removal efficiency generally depends on the conditions of the system such as inflow TAN concentration, temperature, filter surface area, filter type (such as rotating biological contactors, RBC, fluidized bed, trickling filter), hydraulic loading rate and hydraulic retention time (Losordo and Westers, 1994). RAS are generally highly engineered systems (Lenger *et al.* 2001; Losordo *et al.* 1998) available in various configurations (for example, single or multi-stage biofilter exist among others; Hargrove *et al.*, 1995; see fig. 9), with various modules, filters and reactors (Losordo *et al.*, 2000; Malone and Beecher, 2000; Van Rijn *et al.*, 2006). Filter media with low specific surface area such as trickling filters or RBC have higher nitrification rate leading to higher TAN removal from aquaculture wastewater per surface area per day, than higher specific surface areas filters such as fluidized bed filters or plastic bead filters (Miller and Libey, 1985; Rogers and Klemetson, 1985; Van Rijn and Rivera, 1990; Westerman *et al.*, 1993; Losordo and Hobbs, 2000). The key concept for proper functioning of these quasi-closed systems is based on adequate water flow rate (Losordo and Hobbs, 2000). These systems may be used to rear both freshwater and marine organisms, and they vary widely in the efficiency with which the wastes are treated and the water recirculated (Lewbart *et al.*, 1999). The most common problems of water quality in recirculating systems are the oxygen depletion and the accumulation of organic matter, inorganic N (particularly ammonia), P and CO_2. Reviewing the literature on such systems, an average of 5 to 10% of the system volume (according to the definition of intensive recirculating systems; Losordo and Hobbs, 2000) has to be replaced each day with “fresh” water to ensure good functioning of RAS and avoid fish mortality.

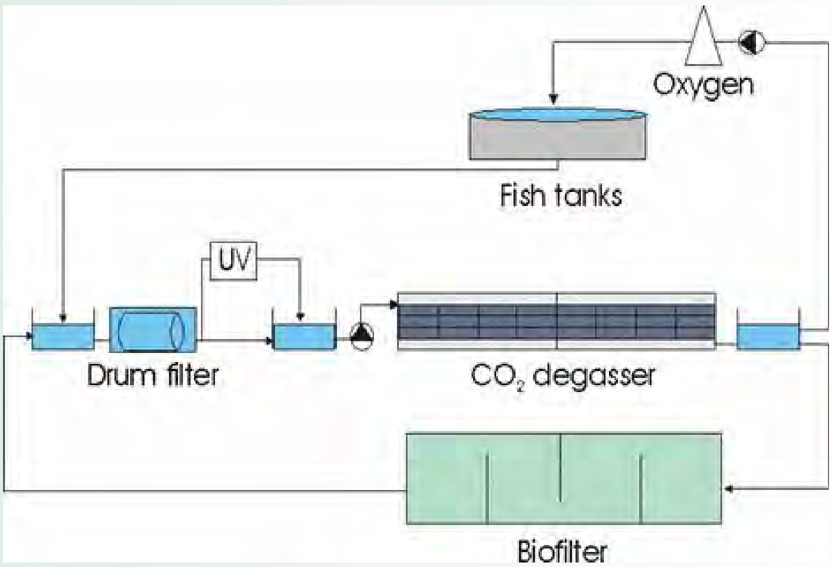


Figure 9: Illustration of a multi-loop recirculating system (after Hargrove et al., 1995)

Many biogeochemical functions of semi-closed or even zero-discharge systems have been investigated (Table 9). Elements recirculation such as P (Van Rijn and Barak, 2000;

Barak *et al*., 2003) or diverse N forms (Van Rijn, 1996; Van Rijn *et al.*, 2006; Neori *et al.*, 2007) have largely been measured in experimental recirculating systems with different design and removal methods. From a review of different studies, a general note remained that parameters need to be maintained under control and managed within an optimal range to ensure the good functioning of such systems (Van Rijn, 1996).

Table 9: Efficiency of different recirculating systems reported in literature.

Author	percentage of waste removed	Techniques used	Species reared
Libey, 1993	40 % of suspended solids	drum filter with a 40 µm screen mesh size	
Scott and Allard, 1984	87 % of suspended matter (> 77 µm in diameter)	Hydrocyclones	
Miller and Libey, 1985	74 – 82 % of TAN removal	RBC	
	23 -52 % of TAN removal	trickling filter	
	8 – 32 % of TAN removal	fluidized sand bed filter	
Rogers and Klemetson, 1985	90 % of TAN removal	RBC	
	80 % of TAN removal	biodrum filter	
	50 % of TAN removal	trickling filter	
Westerman *et al.* 1993	0.25 $gTAN.day^{-1}.m^{-2}$	RBC	
	0.1-0.15 $gTAN.day^{-1}.m^{-2}$	upflow sand and bead filters	
Van Rijn and Rivera, 1990	0.43 $gTAN.day^{-1}.m^{-2}$	trickling filter	
Twarowska *et al.* 1997	Lower flow rate (246 $L.min^{-1}$) = 80.5 ±8.0% / Higher flow rate (469 $L.min^{-1}$) = 64.8 ±19.7 % (= 0.62 mg $TAN.L^{-1}$=0.33±0.11$gTAN.m^{-2}.day^{-1}$) of TAN removal	Biological filter	Tilapia
	Mean suspended solids (SS) removal of 80.0 ±15,8 %	Eco-trap and sludge collection	
	36.5 – 41 % of SS removed	Hydrotech (drum) screen filter	
	35.3 % of volatile solids (VS) inputs removed (17.6 % removed by each component)	two solids-removal components	
	7.5 % of total system volume added per day		
Malone 1992; Malone *et al.* 1993	0.25 -0.50 $gNH_4^+.m^{-2}.day^{-1}$ of ammonia removal (TAN < 1 $mg.L^{-1}$ in the pond)	Expandable granular biofilter	
Shnel *et al.* 2002	85.37 – 96.4 % of TAN removal	fish basins, liquid oxygen contacter, trickling filter, sump, screen filter, sedimentation basin and fluidized bed reactor	Tilapia (in a greenhouse at an experimental station for intensive fish culture - Genosar, Israel)
	Most of the Phosphate recovered (90-100%)	Sedimentation/Digestion basin	
Neori *et al.* 2007	10.75% of C, 14.3% of N and 92.4% of P	fish basins, foam fractionator, trickling filter, sedimentation basin and fluidized bed reactor	Gilthead seabream (*Sparus aurata*)
	Missing nutrient fraction lost in badly-known processes: 70.95% of C and 70.3% of N	Other unexplained aerobic and/or anaerobic processes	

Barak *et al.* 2003	76% particulate-associated phosphorus remain in	Sedimentation basin	Red tilapia hybrids (*Oreochromis nilocticus* and *O. aureus*), then gilthead seabream (*Sparus aurata*)
	13% of the total phosphorus in the system remain in	Fluidized bed reactor	
Losordo and Hobbs, 2000	nitrification rate = 0.5-1.0 g TAN.m^{-2}.day^{-1}	Trickling filter or RBC	Modelling, general estimation from their studies
	nitrification rate = 0.10-0.5 gTAN.m^{-2}.day^{-1}	Fluidized bed reactor or plastic bead filter	

The pattern of filter removal of chemical elements shows generally large spatial variation, since organisms aggregate differently on different substrates (Barak *et al.* 2003).

Efficiency of treatment can vary according to the biofilter media used in a system, for example, organic media such as wood chips and wheat straws, being the most adapted into transition effluent loads, maybe enhancing bacterial growth on their surface (Saliling *et al.*, 2007). The percentage of nitrate-nitrite reduction for different media varies according to the concentration of N among influent. For example, Saliling *et al.* (2007) showed that with an initial concentration of 120.4 ± 9.6 mgN.L^{-1}, the reduction was 90.6 ± 7.5% for plastic media and 98,6 ± 1,0%, for wheat straw media.

Combining physico-chemical and biological processes has been successfully applied in a pilot recirculating carp culture system (Van Rijn and Rivera, 1990; Arbiv and Van Rijn, 1995), where it was found that concentrations of both inorganic nitrogen and organic matter could be kept sufficiently low to ensure adequate water quality for fish growth with minimal water exchange (3 m.day^{-1} = evaporation). Taking into account the flow rate through the reactor and the volume, NO_3 removal (597mgN0$_3^-$N.mm^{-3}.min^{-1}) is almost twice as high as the maximum removal rate (300 mgN0$_3^-$-N.mm^{-3}.min^{-1}) measured in a similar reactor operated without pre-treatment of the organic matter (van Rijn and Rivera, 1990). This treatment system has been applied on a semi-commercial scale but carps have been independently infected by ectoparasites due to non-utilization of prophylactic treatment in order to avoid a possible inhibition or interference with nitrification/denitrification processes.

Twarowska *et al.* (1997) have with RAS with tilapias. They observed that physical filtration removed approximately 35.3% of the feed volatile solids and 80 % of the SS (for sludge collector) with an additional 41% (for the second screen filter). Then, biofilters removed approximately 65% of the average TAN concentration in the culture tank. However, 7.5% of the system volume had to be flushed with 'fresh' water daily to maintain a good water quality.

In term of P removal in a recirculating system composed of a culture fish tank (Tilapia or seabream) and a trickling filter, a foam fractionator, a sedimentation basin, and a fluidized bed reactor (FBR), 76% of the total P of the system removed in the sedimentation basin (associated with particles), and an additional 13% was removed in the FBR (Barak *et al*., 2003), and 10 to 20% uptake by the tilapia culture (Shnel *et al.*, 2002). This efficiency of removal allowed a good recycling of water and a completely close system (few additional tap water compensated for water losses resulting from evaporation and leakage). Neori *et al.* (2007) reported, for the same system with gilthead seabream, a comparable P removal, accounting for 44% in sludge and 92.6% in total waste. On the other hand, nitrification and denitrification processes which control the dissolved N chemistry, removal only 14.3%. However, 70.3% of the N inputs have already been assimilated by fish and biochemical natural reactions (little-known) occurring in the pond.

Limitations of the recirculating systems results in the scarcity of information on microbial populations and metabolic processes involved, since some processes can be passive and/or inhibit according to the conditions in the system (Neori *et al.*, 2007). Losordo and Hobbs (2000) have shown that in a recirculating system, passive nitrification accomplished by bacteria growing on the surface, other than the biological filters, may account for as much as 30 % of the total nitrification. Additionally, there is usually some amount of passive denitrification occurring within the system. In such cases, models are useful to estimate the results of system modifications before time and money is spent (Losordo and Hobbs, 2000).

c) Integrated aquaculture systems with no water recirculation

From the reviewed integrated intensive systems (Tables 5-8), a fish-microalgae-bivalves-macroalgae system showed the highest overall N uptake, 63%, nearly three times more than in modern fish net pen farms (Shpigel *et al*., 1993; Neori *et al*., 2000-2004). Research at the National Institute for Mariculture in Eilat (Israel), has examined the feasibility of a treatment facility comprising oyster culture for organic matter removal and macroalgae (*U. lactuca*) for inorganic nutrient removal (Shpigel *et al*., 1993; table 8). This system was recently developed in the SeaOr Marine Enterprises – on the Israeli Mediterranean coast, but unfortunately without success. Although the increased complexity of the system has been shown to remove nutrients more efficiently and to improve the economics, the system required more highly skilled operators and sophisticated management (Shpigel and Neori, 1996).

The EU project GENESIS (2001 - 2004) studied several types of integrated systems in warm water (Israel), temperate water (Southern France) and cold water (Scotland), with a variety of valuable marine products including fish, crustaceans, molluscs and aquatic plants. The different systems were evaluated based on their performances in respect to water quality, nutrients removal, and waste management. Intensification of aquaculture production systems and the use of nutrients, water and energy were optimized. For example, silica enrichment of the effluent from a fish production system was used to produce diatoms (microalgae) for oyster fattening. The addition of microalgae production resulted in an improved nutrients uptake, and their utilization by oysters allowed the retention of 70% of N introduced with the feed, compared to the 20 to 30% for seabass or seabream rearing in monoculture. This program had also developed suitable products and services for the commercialization of the technology and established the financial viability and consumer acceptance of its products.

The selection of the best suitable seaweed species is essential for establishing good bioremediation systems (Kang *et al.*, 2007). The SEAPURA project (EU project, 2001 - 2004; with fish farms in Spain and Portugal; table 7) selected, developed, and tested different cultivation of high-value seaweed species (red algae: Asparagopsis, Dilmontia, Gracilaria, Gracilariopsis, Laurencia, Palmaria and Porphyra, and green algae: Codium and Ulva) which had not been used before in polyculture. The cultivated seaweed biomass could be used for the human food market, mainly in France, and for fish feed additives with possible antibiotic effects of the cultivated seaweed, or for extraction of pharmaceutical substances. The results from the project showed that the reduction of ammonia concentration in the fish farm effluents was higher (100%) using a cascade phytotreatment tanks system than with only a single passage through one algal tank (50% of N uptake efficiency).

In spite of recent development of research on integrated aquaculture, only the main cultivated organisms are (to a certain extent) well known, and biochemical processes linking each “box” clearly needs further development. In European countries, experiments on the subject have yet to be conducted and many effects to be determined as experiments on integrated aquaculture are just recently developing (in Israel, France, Spain, Italy, Germany, Norway, Scotland, Ireland, Portugal, etc.). For example, little attention is given to bacteria, phytoplankton, periphyton epiphytes, water turbidity, competition between species, effect of medication in the system, etc., which affects directly or indirectly the “true” effect of the targeted species. Moreover, such systems are time-dynamic systems (Lefebvre *et al.*, 2004) since the waste of one resource user become a resource (fertilizer or food) for the others. Experimental studies should be conducted on a year-round basis in order to account for seasonal variation (in physiological nutrient uptake, photosynthesis and growth) of biofilters and thus long-term/commercial scale performance of such cultures. Similarly, the maximum stocking density of species in a certain environment is an essential parameter of a well performing system (Evans and Langdon, 2000).

To conclude, polyculture, which is still not very well developped in Western aquaculture, is theoretically more sustainable than monoculture, due to the reutilization of waste products of one species by another (Nunes *et al.*, 2003). The choice of the suitable organisms depends on many parameters and on the aim of the polyculture (Qian *et al.*, 2001; Schneider *et al.*, 2005). Such assemblage of species must be chosen carefully to complement one another and foster optimal productivity, and multiple species interactions must be investigated to achieve a sustainable ecological aquaculture system. The success of integrated aquaculture may lead to bio-diversity. For the system to be successful, the same natural conditions should be applied to recreate a kind of natural environment and thus improve the productivity of the targeted species.

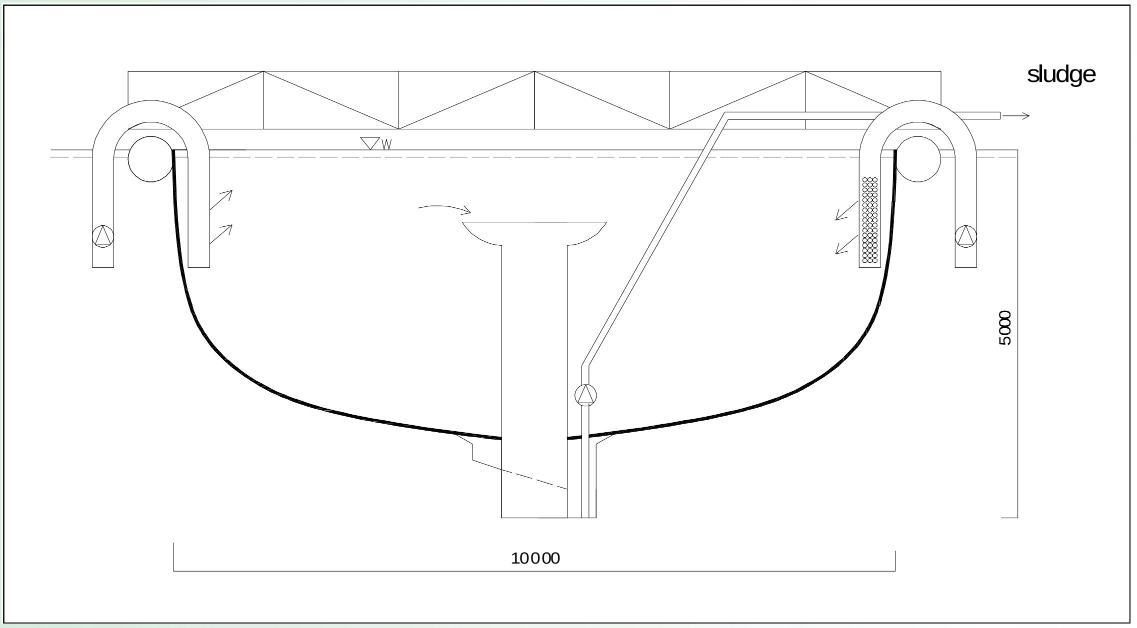
d) Net cage aquaculture in lake or costal sea

One of the clear differences between land-based (ponds) and open-sea (cages) aquaculture systems is the degree of control that the operator has over key processes such as maintenance of water quality and nutrient/effluent release (Troell *et al.*, 2003). Troell *et al.* (2003) reviewed 28 studies of seaweeds in integrated systems, and reported values of DIN uptake. For the 23 land-based studies, the range of DIN uptake were reported to extent from 2 to 100%. On the other hand, none of the 5 reviewed open-water studies reported any nutrient recovery values because the dispersion of wastes in such system is more difficult to control. In the literature, only an hypothetical calculation based on modelling (Troell and Norberg, 1998) gave seston removal values by mussels under fish cages in an open seawater environment (table 5). The biofiltration efficiency of different seaweed species has mainly been tested in the laboratory (Chopin *et al.*, 1996, 1999; Gallant, 1993) and many processes that may affect the nutrient uptake process, for example, microbial nitrification (Krom *et al.*, 1995), competition among different macroalgae, effect of epiphytes, etc., have not been taken into account. In addition, the dilution rate of nutrients (limiting factor) has to be controlled in order to achieve the optimal compromise between algal growth rate, nutrient assimilation, competition, and biomass of cultivated species (Buschmann *et al.*, 2001).

Net-pen or net cage fish farms are generally situated in protected coastal waters, with fairly short water residence times, so internal pollution is basically averted by “dilution”. Although there is usually a large biological community (both pelagic and benthic) associated with cage farms (Angel *et al.* 2002, Dempster *et al.* 2002), it is not obvious that the farm effluents are directly utilized by natural communities of primary producers (PI such as microalgae, macroalgae) or secondary producers (PII such as bivalves). However, the work by Chopin *et al.* (2004) and others in large-scale open water systems have demonstrated that multi-trophic integrated aquaculture (IMTA) can both be successful and economically profitable (Ridler *et al.*, 2006; Whitmarsh *et al.*, 2006; Neori, 2007). However, IMTA can be considered as “juvenile” integrated systems since the system would not be optimized in such open systems where the material is lost and/or diluted into the environment, and fluxes are not controllable. Then, ‘extractive organisms’ not only grew on waste derived matter, but also on natural productivity of the ecosystem. Modelling can help to optimise what we can call an integrated coastal zone management as in China (see Grant, Bacher; Troell and Norberg, 1998). Finally, Hernandez *et al.* (2006) have mentioned how uncertain it may be to scale up ecological results obtained under laboratory conditions. This is the reason why particular attention must be paid to open-water systems (Wang, 2003).

One solution could be to “close” the sea cage using bag pens with sludge treatment by dissolved air flotation (DAF). Bag pens can be used to reduce nutrient emissions from farming of rainbow trout. A bag pen is a floating farming basin made of, for example, flexible plastic fabric (Fig. 10). There is no free water movement between the bag pen and the surrounding water body, but the water is pumped into the pen and led out with the help of the hydraulic pressure difference caused by the pumping. Water is pumped tangentially into the pen, which creates a swirl inside the pen. The swirl moves settling solids into a sludge hopper in the centre bottom of the pen. The use of a bag pen enables collection and withdrawal of the solids from the bottom of the pen with a separate pump. Nutrients bound to the solids are thus prevented from getting into the water body resulting in reduced nutrient emission (Jokela, 1999).

Figure 10: A schematic picture of a bag pen (Jokela, 2003).



In full-scale trials, Jokela and Vuori (2004) showed that bag pen farming of rainbow trout reduced P and N emissions by 50 and 10%, respectively compared to net cage farming.

The water-solids mixture pumped from the bottom of the bag pen is very dilute and needs additional concentration, for example, by DAF, to facilitate further solids handling and disposal. The principle of the DAF process is based on very small air bubbles which lift up the solids, forming a thick solids layer on the surface of the DAF tank. The solids separation can be enhanced by adding chemical coagulants. One DAF unit is sufficient for several bag pens. Over 90 % phosphorus reductions have been achieved in DAF pilot trials (Jokela *et al.*, 2001).

Investment and operational costs of bag pen farming can partially be compensated with the increase of the fish stocking density. Stocking densities of over 45 $kg.m^{-3}$ can be used (Jokela 2003). In some cases, pure oxygen addition may be needed. Compared to net cage farming, the additional costs attributed to prototype bag pen farming have been calculated to be €0.5–0.9.kg^{-1} fish weight gain with corresponding production capacities of 180,000–30,000 kg fish.ha^{-1} (Jokela and Vuori, 2004). The increase in production capacity reduces the unit costs. The integrated bag pen technology (bag pen + DAF) is in its emerging phase.

e) Recirculating integrated aquaculture systems

When integration is done properly, water can be re-used and integrated aquaculture could become a recirculating aquaculture (Losordo et al., 1998). Such recirculating integrated aquaculture systems (RIAS) whose partitioned aquaculture systems (PAS; Brune *et al.*, 2003) are at an experimental stage, but many parameters must be controlled in order to allow a good functioning of such systems.

These new theoretical designs enhance the advantages of each process (*that is*, of herbivory and saprophagous pathways) previously seen. Using data from literature and mass balances calculations (where output = input – retention), Schneider *et al.* (2005) have tested a hypothetical application of RIAS. They calculated for each module of the trophic chain of an integrated system (based on results from experimental studies) the amount of nutrient retained into new biomass, with output from one module serving as input for the other subsequent module (Fig. 11).

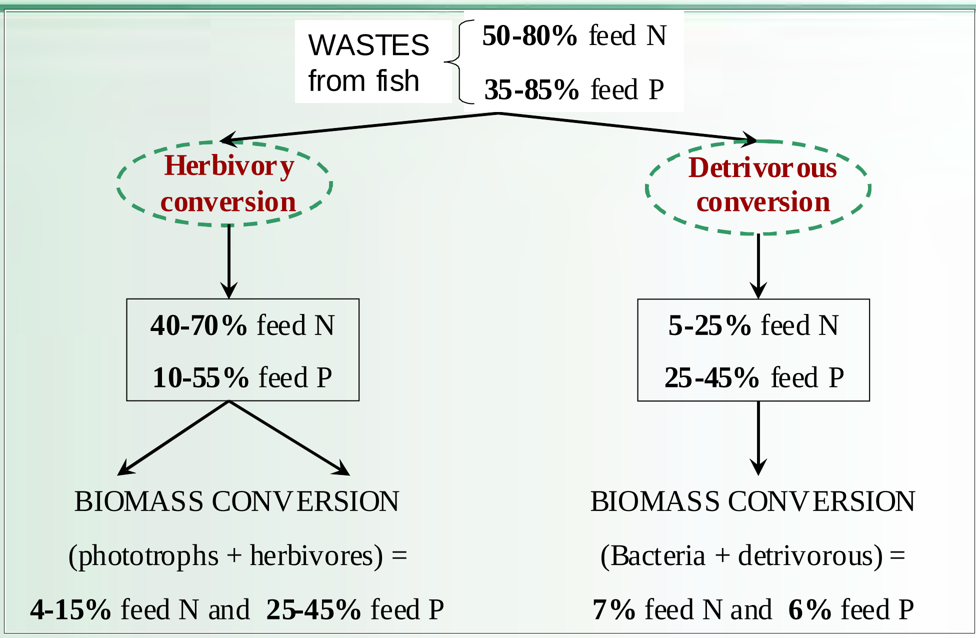


Figure 11: Scheme relating the different N and P nutrient retention in an hypothetical RIAS (after Schneider *et al.*, 2005)

In spite of multiple advantages of this new design, some limitations remain; those of each species used for each module (see relevent chapters above). Although mitigated, one of the problems of aquaculture (except the need for terrestrial area for the farm implantation in land-based cultures) is the need to feed the main cultivated species, which does not solve food and oil fish dependence on fisheries. Actually, these main species with high value are usually carnivorous. For example, the farming of salmons needed to be provided with food being about 6 times larger than the contribution of all industrial activities directly or indirectly involved in the cage-farming (Folke and Kautsky, 1989). Despite some recirculating systems (using the same water) reduce water requirement by recycling (Neori *et al*., 1998), the need for additional food sources remains important for the highest culture level species. Indeed, fish and shrimp (pertaining to the group of Producers III or IV) are the most important seafood crops in Western countries. However, herbivorous and omnivorous fish (mainly Producers II) are ecologically superior to carnivorous fish since they waste less nitrogen than species higher up the food chain, and some of them can recycle their own waste by consuming *in situ* population of algae, bacteria, and other organic particles that develop in the culture water (for example, grey mullets). Therefore, the use of omnivorous or herbivorous as principal cultivated species should be further studied, which could revolutionaries aquaculture view in the future.

f) Agriculture-aquaculture integration including wetlands

At Hatzeva, Israel, water lily ponds act as sediment collectors and biological filters for polluted water discharged from a tropical fish farm, removing BOD and surplus nutrients. The water lilies are harvested and sold to the growing leisure garden pond market. The partly purified aquaculture wastewater is then used for drip irrigation of salt tolerant perennial crops like date palms. The result is multiplied economic return from a given water volume, essential for extending the life span of limited fossil groundwater reserves in desert areas (Boussiba and Leu, personnal communication)

The Ramat Highland district, Israel, collectively uses 4.5 million cubic meters of brackish water per annum. The water is drawn from a depth of between 550 and 1000 m from underlying aquifers at a salinity of 2680-4360 TDS (Total dissolved salts in mg. L^{-1}) and temperature range of 39-41°C. At Kibbutz Mashabe Sade, Israel, Barramundi (Glencross, 2006) is cultivated in the pre-cooled water. The polluted water is collected and partly purified in an oxidation pond and subsequently used for irrigation of olives or jojoba. This system is completely integrated, combining biological with physical processes, aquaculture with agriculture and thus avoiding water waste and negative consequences of effluents on the environment.

SUSTAINAQUA (EU project, 2007-) project aims also to reduce the wastewater discharge to surface waters to nearly 0% discharge, and to increase biodiversity, through 5 particular case studies which combine aquaculture with agriculture systems (with trouts in Denmark; tilapia in Switzerland; carps in Poland, african catfish, pike and pikeperch in Hungary; and eel inthe Netherlands). For example, in Switzerland, a tropical greenhouse has been built (SUSTAINAQUA, 2007) where the production of tropical fish and fruits are linked in terms of water and nutrient flows. In the tropical greenhouse in Ruswil (Sursee, Lucerne, Switzerland), waste heat generated by a natural gas densification plant is being used since 1999 to produce a tropical climate in which integrated aquaculture-agriculture system is maintained (see: www.tropenhaus.ch, cited May 2007). Tilapias are grown in the glass house, and then, the fish wastewater is used to irrigate and fertilize the entire tropical fruit garden (for example, papaya, banana).

Elsewhere, cucumbers, tomatoes and lettuce have also been tested in an integrated fish culture hydroponics vegetable production system, to improve water quality in a fish tank and the economic performance of the system (for example, Zweig, 1989).

At the moment, the Constucted Wetlands are seldom used (mostly in freshwater or brackishwater, since no plants have yet been found able to resist at the salinity of the Mediterranean water) but have demonstrated their efficiency. However, such systems do not fulfil the previous definition of integrated aquculture systems, since many unknown processes remain (for example, passive nitrification/denitrification) and production in these areas can not be valorised. Constucted Wetlands are just treatment services that the nature offers.

III.3 Fisheries

III.3.1 Valorisation of fisheries co-products

Although the trend has been changing over the years and instead of throwing the by-catch into the sea, the practice of bringing them to the shore and selling them to the poorer sections of the consumers is gaining momentum, no further works relating to the transfer technology (by using floating net bags, etc.) is in progress. It is, however, felt that more intensive studies are required on this issue to utilize the by-catch more profitably taking into account the existing market systems, identifying the constraints to distribute larger volumes of fish and to assess the value of the catch to make this activity more attractive. Incentives for landing this by-catch and for the processors to produce a value added product both attractive to the consumers and remunerative to the processors are to be given due consideration. A sampling programme for data collection on population parameters of selected species occurring regularily in the by-catch is necessary, and, in conjunction with the present practices, this would ensure suitable solutions for the use of by-catch in an optimal way.

Fish waste management has been one of the problems having the greatest impact on the environment. Marine capture fisheries contribute over 50% of total world production and more than 70% of this production has been utilized for processing (*FAOSTAT, 2001*). As a result, every year a considerable amount of catch is discarded as processing leftovers and that includes trimmings, fins, frames, heads, skin and viscera. In addition to fish processing, a large quantity of processing by-products are accumulated as shells of crustaceans and shellfish from marine capture fisheries (*FAOSTAT, 2001*). Therefore, there is a great potential in marine bioprocess industry to convert and utilize more of these by-products as a valuable products. Treated fish waste has found many applications among which the most important are animal feed, biodiesel/biogas, dietetic products (chitosan), natural pigments (after extraction), food-packaging applications (chitosan), cosmetics (collagen), enzyme isolation, Cr immobilisation, soil fertiliser and moisture maintenance in foods (hydrolysates).

a) Biodiesel/Biogas

Kato et al. (2004) evaluated the use of ozone treated fish waste oil as a transportation diesel fuel. The obtained oil was found to have suitable properties for use in diesel engines, such as higher heating value and density, lower flash and pour points, no production of sulphur oxides, lowered or no soot, polyaromatic hydrocarbons, and carbon dioxide emissions as compared with commercial diesel fuel.

b) Natural pigments

Carotenoids are responsible for the colour of many important fish and shellfish products. Most expensive seafood, such as shrimp, lobster, crab, crayfish, trout, salmon, redfish, red snapper and tuna, have orange-red integument and/or flesh containing carotenoid pigments (Haard, 1992). Shrimp waste is one of the most important natural sources of carotenoids (Shahidi *et al.* 1998) used for carotenoids extraction with various organic solvents and solvent mixtures. The recovered carotenoids can be effectively used instead of synthetic carotenoids in aquaculture feed formulations, and the residue available after extraction may be used for the preparation of chitin/chitosan (Sachindra *et al.*, 2006).

c) Other uses.

A large amount of offal generated from fish processing, would be a potential source to produce good quality fish oil for human consumption, especially from fatty fish processing by-products. Fishmeal and fish oil are also used as supplement in aquaculture feed and pet food.

The recovery of chemical components from seafood waste materials, which can be used in other segments of the food industry, is a promising area of research and development for the utilisation of seafood by-products. Fish skin and bone waste could be used as potential source to isolate collagen and gelatin, and these molecules are used in diverse fields including cosmetic industries.

Fish skin collagen is used in medical and pharmaceutical industries as carrier molecules for drugs, proteins and genes. Fish skin gelatins have shown an antioxidant and antihypertensive activity. Fish muscle derived peptides have shown numerous bioactivities for medical applications, like treatments for hypertensive, anticoagulant and antiplatelet properties, antioxidant activities or acceleration of calcium absorption.

IV. Modelling and decision support system tools

Modelling is defined as a simplification of realities. From all known laws and mechanisms that drive a process, only the relevant ones will be chosen to simulate the behaviour of a particular system (Straskaba and Gnauss, 1985). This simplification and the undergoing choices are made under assumptions related to the objective/question of interest for the scientist and/or the manager (Jorgensen, 1988). This means that a model is usually structured depending on specific aims/questions and a generic purpose of application is often missing (Zonneveld *et al.*, 1998). This statement may rely on the two main approaches available in modelling, that is, empirical or mechanistical ones (Straskaba and Gnauss, 1985). Empirical refers to black box modelling and it is done from the measurements of state variables of a given system. The understanding is limited to the system investigated, the predictions are exact on the short term but the management could be done only for a narrow range of conditions. On the other hand, mechanistical modelling is based on theory and experiments on relevant processes, and the understanding is more general and for the given class of system. While this white box modelling is less exact on the quality of prediction, it can be used for the long term and in a broader range of situations. Actually, experimental studies alone are either not possible at all or do not capture entirely the dynamic properties of food production systems. In addition, there is no possibility to predict the responses of the system in other contexts than the ones experienced. And finally, there is no possibility to optimize the diverse components of such systems as biology, physics, geochemistry and economics. For these reasons, modelling in general and particularly mechanistical modelling is therefore of interest for the complexity of waste management in agriculture as in aquaculture. In this context, fisheries are out of the direct aim of waste management modelling and will not be treated in the following text.

IV.1 Nutrient modelling and farm-management software tools in agriculture

Due to its very nature, diffuse pollution can be more difficult to recognise and, at an individual farm level, may be regarded by the land manager as unimportant, even if the owner recognises that diffuse pollution is taking place. However, when the pollution increased across a catchment or similar area, the environmental impact can be substantial. Helping farmers to recognise and understand the problem is an important starting point for achieving change (www.environmentsensitivefarming.co.uk).

Occurrence of pollutants in streams and lakes adjacent to agricultural fields have prompted much discussion about whether to ban or greatly restrict agricultural activities that would potentially impair water quality. Equally important in this debate has been the recognition by producers that not all fields behave the same, and thus there is a need to develop more site-specific information that can be used for planning and management.

(www.ctic.purdue.edu/KYW/Abstracts/Hatfield.html).

However, site-specific nutrient management strategies require considerable efforts for compiling and handling site-specific data, such as field history and weather data. Therefore, some farmers feel reluctant to calculate site-specific fertiliser rates. Many farmers use empirical values instead, which often lead to over fertilisation and nutrient losses to the environment. Farm management software can help to solve these problems.

The list below gives a few examples of nutrient models and management software that are available on the internet.

Daisy

Daisy is a Soil-Plant-Atmosphere system model designed to simulate water balance, heat balance, solute balance and crop production in agro-ecosystems subjected to various management strategies. The water balance model comprises a surface water balance and a soil water balance. The surface water includes a model for snow accumulation and melting, a model for interception, through-fall, and evaporation of water in the crop canopy, and a model for infiltration and surface run-off. The soil water balance includes water flow in the soil matrix as well as in macropores. Furthermore, it includes water uptake by plants and a model of water drainage to pipe drain. The heat balance model simulates soil temperature and freezing and melting in the soil. The solute balance model simulates transport, sorption, and transformation processes. Special emphasis here is on nitrogen dynamics in agro-ecosystems.

Mineralization, immobilization, nitrification and denitrification, sorption of ammonium, uptake of nitrate and ammonium, and leaching of nitrate and ammonium are simulated. Degradation, sorption, uptake and transport of agro-chemicals like pesticides are also simulated. The crop production model simulates plant growth and development, including the accumulation of dry matter and nitrogen in different plant parts. Furthermore, the development of leaf area index and the distribution of root density are simulated.

Competition for light, water and nitrogen between plant species are also simulated. The agricultural management model allows for building complex management scenarios. For more information including a free model download, see www.dina.kvl.dk/~daisy/index.html.

EU-Rotate_N

This model consists of a number of subroutines to simulate plant growth both below and above ground, N mineralisation from the soil and crop residues, subsequent N uptake and balance between supply and demand to regulate growth. These will all be regulated by weather factors such as rainfall, temperature, and radiation. Results from the model (simulations) are the flow of water and N into the plant, subsequent evapotranspiration or leaching (www2.warwick.ac.uk/fac/sci/whri/research/nitrogenandenvironment/eurotaten/model/model_description_8_august_2007.pdf). This model can be used to evaluate the effects of different N fertilisation and crop rotation strategies, with regard to both crop yield and N losses. For more information including a free model download, see www2.warwick.ac.uk/fac/sci/whri/research/nitrogenandenvironment/eurotaten/.

NDICEA

The program NDICEA is a nitrogen planner that presents an integrated assessment on the question of N availability for crops. This is more than a simple N budgeting for each crop: crop demand on one side, and expected availability out of artificial fertilizers and manures, crop residues, green manures and soil on the other side. The release of N as a result of the mineralization of the different types of organic matter in the soil are calculated, depending on soil type, temperature and rainfall. Losses due to leaching and denitrification are also calculated. During the growing season, the resulting net available N is compared with the crop demand in time steps of one week.

The NDICEA model has been developed, tested and used for more than ten years by the Louis Bolk Instituut, Netherlands. For more information including a free model download, see www.ndicea.nl.

STANK

Provision of an individual advisory service concerning plant nutrition and use of manure is an effective way to minimise N and P leaching. STANK is a planning computer program developed primarily as a means to achieve a good management of manure, but it can also be used on farms which do not apply manure and on farms growing field vegetables. Use of STANK in the advisory service for vegetable growers has shown that the program works well for these crops. (www.actahort.org/books/506/).

N-Able

N-able is a N crop response model, which simulates the growth response of 28 crops to applications of N fertilizer. It includes the effects of organic material, climate and leaching, complete with graphical output. This on-line dynamic model will deliver results in a few seconds (www.qpais.co.uk/nable/nitrogen.htm).

A comprehensive list of agro-ecosystem models can be found at www.wiz.unikassel.de/model_db/models.html. However, it is noted that many models in this data base are scientific tools rather than farm management tools.

IV.2.2 Decision support system tools in aquaculture

Due to their complexity and human driving processes, the use of modelling is of great interest to understand and to manage aquaculture systems in general and to management wastes in particularl. Two main kinds of waste are produced by aquaculture facilities; particulate and dissolved material whatever the environment of production (land-based or open water systems, marine or freshwater systems). The physical or biological processes that can limit and/or reduce the production of these wastes are rather different so as the associated modelling approach.

However, modelling has been historically under used in aquaculture research and management, but this tendency has been reserved over the past 15 years (Lefebvre, 2003). Applicability of modelling in aquaculture waste managements can be drawn from diverse model examples in biology, geochemistry, hydrodynamics, and physics (Piedrahita, 1995).

Special modelling aims for particulate material wastes will be for instance the simulation of the deposition of faeces around sea cages and/or the regeneration rate of organic matter into nutrients by bacteria and their consecutively oxygen consumption and degradation of the benthic environment (Cromey *et al.*, 2002; Doglioli *et al.*, 2004). Hydrodynamics could be also optimised to favour settlement of particulate waste as in Huggins *et al.* (2005) in raceways for instance. In the same way, Islam (2005) proposed a simplified, conceptual, nutrient mass balance model to derive the level of nitrogen and phosphorus discharged from a hypothetical cage culture system. The model was intended to estimate the approximate level of N and P added to the environment for every ton of fish produced, based on various assumptions on feed loss, FCR, N and P content of feeds and fish and various nutrient dynamics within the cage culture system. The author used the final effluent nutrient loading value from the predictive hypothetical model to predict the total global loads of nutrients by simply multiplying the nutrient loading rates with the assumed global cage aquaculture production in the marine environment.

Special modelling aims for dissolved nutrients will be for instance to simulate their use by autotrophs (plant, algae and bacteria) and indirect incorporation into valuable predators (bivalves, abalone, sea urchins…; Lefebvre, 2003). Some relevant models were developed in the context of pond aquaculture (for example, Jimenez-Montealegre *et al.*, 2002) and even in some bays to improve coastal zone management (Duarte *et al.*, 2003). Moreover, some models can be found for special case studies such as recirculating systems (Pagand *et al.*, 2000b) or integrated aquaculture systems (Ellner *et al.*, 1995; Lefebvre and Pomarede, 2005).

In all cases, modelling is interesting to optimize the size and biomass of the different compartment of aquaculture systems and to limit and reduce their wastes. Finally, in order to account for economy, management and biology simultaneously, decision support system tools (DSS) have been recently developed for aquaculture and should be generalised to aquaculture waste managements (Pond by Bolte *et al.*, 2000; Aquafarm by Ernst *et al.*, 2000). However, to be efficient and applicable to a broad range of situations, these tools must be based on mechanistical modelling approaches as much as possible (Lefebvre, 2003).

General Conclusion

As the world population and economy grow, water becomes an increasingly scarce commodity. Fish farming takes place across Europe in a variety of environments and is a big user of either freshwater or seawater. Aquaculture, along with all animal husbandry, produces effluents that contain dissolved and particulate nutrients, which can lead to ecological disturbances in the receiving ecosystem. The European Union is therefore committed to promoting and encouraging the sustainable use and efficient management of water resources across the continent. Innovative projects that help industry to optimise water use and reduce the impact on the environment are a part of that commitment.

To reduce outputs of FFA industries, that is to say the amount of waste, two kinds of solutions may be applied:

1. To reduce and optimise the inputs (that is, fertiliser, feed)
2. To convert and valorise the outputs

The first solution lies in feeding improvement techniques, with optimal rearing conditions for species reared, to reduce stress and to promote optimal growth. In agriculture, manures and fertilisers spread on fields have to be measured adequately and matched with the type of soil and the need of the culture. In aquaculture (and terrestrial animal culture), the quantity as well as the quality of feed (formulation, digestibility, etc.) can be improved with the knowledge of each species physiology. However, there is a need of reducing the pressure on stock caught for fish meal and fish oil (which represent about 50% of the fisheries in the world), because this kind of fisheries is unsustainable and lead to a depletion of natural stocks, in addition to the low rentability (FCR>1; to rear 1kg of fish, 2.5 to 5kg of wild fish are necessary). European aquaculture should move towards cultivation of herbivores or omnivores rather than top predators, feed with sustainably produced plant-based food. Consequently, fisheries could become more independent and the current rising of an ‘Ecosystemic Approach of Fisheries’ could really be applied and protect natural ecosystems. The key word for agriculture and aquaculture is then a common management, with agriculture becoming an important source of food for aquaculture.

The second type of solutions lies in re-use of outputs (uneaten feed, released). Many techniques exist but few are actually in use, commercially exploited because of the low final benefit or low understanding of processes involved. Here again, the more sustainable option seems to integrate aquaculture and agriculture management, re-using wastes from one industry which could become resources for species reared by the other industry and recycling water. Different aquaculturists (mainly in the North of Europe) are already using fish sludge to the neighbouring farmers to fertilise their land as the results were positive (project FAIR CT 98 9110; Aquaetreat, 2007). This should be done in close environment in order to avoid effluents and escapees. Then, natural ecosystems could be preserved from agri-aquacultural threats. There is a growing need of valorising the by-catch among fisheries, avoiding discards; and this can be achieved by many ways, beginning with more selective fishing gear and responsible fisheries. Finally, by-catch can become agro-aquafeed (but not the essential part) and the recovery of wild stocks could begin with suitable management options (MPA, Artificial reefs, etc).

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CSF policy packages: http://www.defra.gov.uk/corporate/consult/waterpollution-diffuse/ria-ds.pdf

Model download:
DAISY : www.dina.kvl.dk/~daisy/index.html
EU-ROTATE-N:
www2.warwick.ac.uk/fac/sci/whri/research/nitrogenandenvironment/eurotaten/
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