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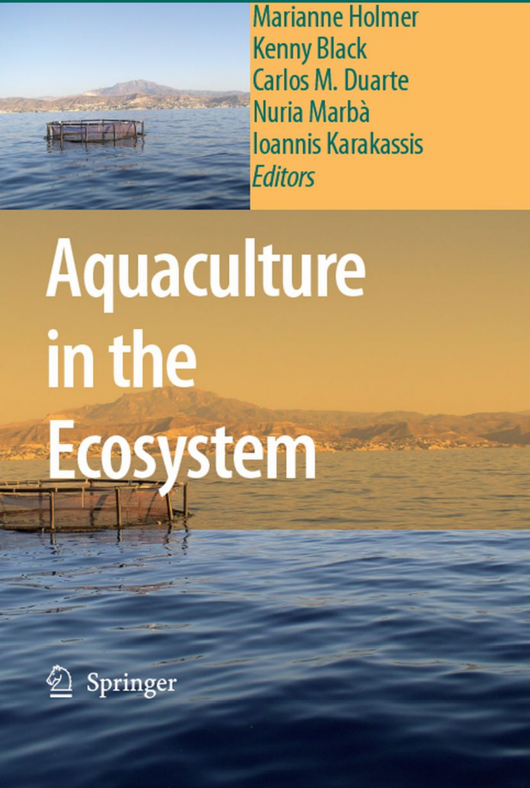

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Foreword

Aquaculture in the Ecosystem – An Introduction

The growth of Aquaculture and its future role as a food supplier to human society has environmental, social and economic limitations, affecting marine ecosystems and socio-economic scales from local to global. These are close links with human health requirements and societal needs for various goods and services provided by marine ecosystems. This book shows this broad spectrum of dependencies of the future growth of aquaculture and highlights both relevant problems and expectations.

Compensating for stagnant wild capture fisheries and the increasing demand for marine products, marine aquaculture is one of the fastest growing industries in the world, comparable to the computer technology industry (Chapters 9 and 10). The demand for marine products is controlled by a complexity of factors in our society, not least the increasing human population and the increasing global affluence that allows the consumer to buy higher priced marine products such as salmon, tuna and shellfish (Chapter 9). The populations of several of these top-carnivore species are seriously compromised and it will be impossible in the future to maintain wild captures at the level of consumer demand. In less affluent areas including SE Asia and Africa, aquaculture for both domestic consumption and export has major nutritional and economic benefits. The production of fish in aquaculture is thus expected to increase under the assumption that the bottlenecks for expansion can be overcome (Chapter 10). This book discusses a range of bottlenecks, not only the environmental, but also technological, social and economic constrains.

Aquaculture is an ancient activity enduring over millennia. Cultivation in historic times was primarily for domestic use but, at the beginning of the 20th century, larger farms started to appear, such as rainbow trout farms in fresh water ponds in Northern Europe (FAO 2006). Since then the number of species domesticated for aquaculture production has increased exponentially now exceeding the number of species domesticated on land (Duarte et al. 2007). There is a large potential for further species in aquaculture as only about 450 species are currently cultured out of about 3,000 aquatic species used for human consumption. Characteristically, the first initiatives in aquaculture were simple, low technology systems with limited demands for maintenance and low operating costs. These aquaculture systems were dependent on high water quality which was often easy to achieve because of their low intensity. It was not until greater intensification of aquaculture in the 1970s, increasing the pressure on the environment significantly, that it became urgent to monitor and regulate aquaculture (Chapter 2). The current expansion rate in world aquaculture production of 3.5–4.6% yr^{-1} can only be sustained if the major pressures exerted on the environment and dependence on natural resources, such as feed and brood stocks (Chapter 10), are reduced.

With regard to regulation and monitoring at present time, the Water Framework Directive (WFD) is being implemented all over Europe and will become important for the regulation of aquaculture and other human activities in the coastal zone (Chapter 1). Chapter 1 clarifies present understanding of eutrophication and provides an insight into water quality models on as they are expected to be used under the WFD, providing examples from Scotland different scenarios for the future regulation of marine aquaculture in the coastal zones. Aquaculture producing countries outside Europe regulate aquaculture activities through a number of different laws and conventions, often with several laws enforced on different aspects of the production cycle (Chapter 2). In Norway, which is one of the top five producers in the world and where the production of salmon in net cages in the coastal zone is an important contributor to the national economy, the monitoring of environmental impacts of the industry has been developed since the beginning of the industry 30 years ago and is now a classified program according to national standards implemented throughout the country (Chapter 2). As an example of a more recent developed program, the monitoring in Malta is presented (Chapter 2). During the 1990s, the Mediterranean experienced an exponential growth in the production of sea bream and sea bass in net cages and, as the environmental conditions in the Mediterranean are unique (e.g. widespread oligotrophy), some of the environmental pressures differ considerably from those in Northern Europe. One example is the prevalence of seagrass meadows of the species *Posidonia oceanica* as a benthic ecosystem along Mediterranean coasts. As this is a sensitive ecosystem, facing general declines in the coastal zone (Marbá et al. 2005), it is important to monitor this ecosystem in fish farm surroundings to avoid accelerating declines (Chapter 2). Tuna farming (or ranching) is a major activity in Malta as well as in several other Mediterranean countries and, although it is debated whether this industry is "real" aquaculture or should be considered as a fattening industry instead, the environmental impacts differ from sea bream and sea bass aquaculture due to the use of wet feed (fresh/frozen fish) instead of dry feed pellets.

A new development in aquaculture monitoring and regulation, which will play an important role for future development, is in considering aquaculture as an integrated part of the marine ecosystem. This means that aquaculture should be managed together with a number of other industries and other users of the marine ecosystem (Chapter 3), but also that the production is a part of ecosystem and has to be managed at different scales, not only the water column and sediment floor in the vicinity of the net cages, but also at larger scales in the coastal zones (Chapter 1). One example of scale can be found in Chapter 5, which addresses the issue of introductions of alien species into coastal zones caused by aquaculture operations. This is particularly important since it is well known that aquaculture is the second most important vector for species introductions after maritime transport. Also the attraction of wild fish to net cages adds constraints to the ecosystem structure and function, in particular in areas such as the Mediterranean, where wild fish are abundant around cages and may be more available to fisheries (Chapter 3). Although the presence of wild fishes at the farms can minimize the environmental impacts, e.g. through reducing inputs of organic matter to the seafloor, there are risks such as transfer of diseases to wild populations (Chapter 3). A related issue is the genetic pollution of wild stocks through either inadvertent (as in farm escapes) or deliberate (as in stocking/ranching) introduction of cultured species into the wild (Chapter 4). Genetic impacts have been extensively studied for salmon in Northern Europe, where there are problems with interbreeding, and are now under consideration for other cultured species such as sea bream and sea bass in the Mediterranean and for other species in the tropics (Chapter 4). Chapter 4 discusses the possible future solutions to the genetic interactions between farmed and wild fish.

One major constrain to aquaculture growth is the availability of fish meal and fish oil for production of carnivore fish (Chapters 6 and 10). There is currently a major research effort in optimizing feed through substituting fish meal and oil with vegetable flour and oil. As there is substantial scientific evidence of human health benefits from consumption of marine products, primarily due to the omega-3 fatty acids, the aims of the current research is to maintain the composition of the cultured fish product while reducing dependence on fishery-derived feedstocks (Chapter 6). There are also other future options for solving the bottle neck of feed availability, which involve not only breakthroughs in feed technology but also changing the way humanity interacts with the oceans (Chapter 10). Such breakthroughs could be through use of marine plants for feed or moving production from carnivore to herbivore species.

Aquaculture is expected to develop along two main lines, either in net cages at sea or on land-based facilities (Chapter 10). To keep up with the production needs the size of the farms will expand and net cage farms will move from coastal sites to open-ocean locations. Land-based farms have the advantage of reuse of the water and treatment facilities, but are at the present constrained by high energy costs.

In addition to technological constrains there are several other bottlenecks, which are less predictable. These are related to attitudinal issues (Chapters 8 and 10) and to the economic development of the industry (Chapter 9). Aquaculture production has for instance become of active interest to a number of non-governmental organizations (NGOs) around the world, which is discussed in Chapter 7. NGO concerns about aquaculture are not solely in its growth or where the product is consumed. Rather, their interest is in the on-the-ground environmental or social impacts that threaten or undermine the NGO's ability to deliver on their overall missions of conservation or social welfare. Public and consumer attitudes and legislation, related to, e.g., ethics, environment and health can play important roles, such as observed with the threatened bird flu pandemic, where suddenly almost every consumer stopped eating chicken. This did affect the sales of salmon from aquaculture positively, whereas the news on high dioxin levels in cultured salmon resulted in a major, if transitory, reduction in the consumption of fish. One possible way to comply with public attitudes and to impose legislation is through resolution of externalities through monetary valuation of the interactions between aquaculture and the environment and vice versa (Chapter 8). Externalities can be used for policy formulation, e.g., through introduction of environmental taxes and make the producer aware of the environmental costs.

Changes in the market may significantly affect the development of the aquaculture industry, as production only takes place if there are economic benefits to the producer. Chapter 9 analyses the past development in the economics of the industry and from this analysis predicts future trends. It is predicted that production will move towards a few high-volume species supplemented with a large number of small-volume species for local markets. High-volume species have the advantage of predictability and can be sold in the large and global supermarket chains, where weekly sales can be promoted founded on the stability of delivery. High-volume productions are characterized by relatively low production costs. On the other hand, the small-volume species can be sold at a higher price at local markets depending on season and demand.

Aquaculture has increased tremendously in the last decades and is predicted to continue this increase. The aim of this book is to provide a scientific forecast of the development with a focus on the environmental, technological, social and economic constraints that need to be resolved to ensure sustainable development of the industry and allow the industry to be able to feed healthy seafood products to the future generations.

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Contents

Chapter 1
Fish Farm Wastes in the Ecosystem

Paul Tett

Abstract Fish farms release dissolved and particulate waste into the ecosystem and the most important impacts on the water column and the sediments are described at different scales (A, B, C zones). An overview of the ethical and legal frameworks for management of aquaculture is given, introducing the ecosystem approach to regulation through the DPSIR (Driver-Pressure-State-Impact-Response) approach and EQSs (Environmental Quality Standards). The Scottish loch Creran is used as a case study due to the existence of long term monitoring and the presence of aquaculture in the loch. Finally the prospects for management of aquaculture within the European Water Framework Directive is discussed, and it is predicted that the implementation may either result in limited changes (e.g., same practice but out-phasing of environmental hazards) or major changes (e.g., ecosystem approach to aquaculture through polycultures) to Scottish regulation.

Keywords Eutrophication, water framework directive

1.1 Introduction

This chapter is about the interactions between fish-farming and its environment, and how these interactions might be managed in the best interests of ecological sustainability. Despite humanity's generally bad record in this respect, there is evidence that we can learn how to live with, as well as in, Nature (Diamond 2005). There is an increasing will to do this, made concrete within the European Union by the Water Framework and other Directives, and an increasing body of scientific knowledge that can be used for management. I aim to give overviews of both the relevant science and an ethical and legal framework for management. This framework grows out of

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the “ecosystem approach”, which is grounded not only in the scientific theory of ecosystems but also in views about how we might or should try sustain our species’ existence on spaceship Earth. Unlike the planetary-scale problem of global warming, the fish farm–environment interaction is more tractable both to management and to discussion within the space of this chapter: it largely takes place on space and time scales that are easy to see. Nevertheless, the general principles are the same, and if we cannot deal with the impacts of fish-farming – and I think we can – we are unlikely to be able to deal with the bigger matters.

Because I am writing for regulators, policy makers, human health and nutrition community, and coastal zone managers, as well as post graduate students in the field of aquaculture, I include in this chapter some accounts of ecological principles and attempt to explain them without assuming any prior ecological knowledge. And so I start by explaining why there are concerns about the environmental impact of marine aquaculture.

1.2 Humans and Pollution

Once upon a time there was (or may have been) an Edenic age in which small bands of Eves and Adams and their children wandered through a unspoilt Mediterranean landscape of small woods and pastures, trapping wild animals and tending wayside gardens where grew the plants that later became fully domesticated (Mithen 2003). These small bands stopped for the night or perhaps for a few weeks before moving on, and, like all humans, they pissed and shat and threw away uneaten bones or fruit. As human population density, and agricultural skills, increased, the settlements grew larger and less temporary: but never long-lasting, because human wastes polluted water supplies, and wood cutting and agriculture damaged local ecosystems. So villages rose and decayed, and populations moved on, or died from disease and malnourishment, until humans began to learn how to regulate their waste.

It became possible to live in cities, giving rise to another period of population increase and environmental pollution. Classical Rome dealt with waste by piping it down a “cloaca maxima” into the Tiber, where it was flushed out to sea; but elsewhere, Roman mining of metals such as copper and silver created toxic zones where the soils were rich in heavy metals and streams ran red with acid water. By the late 19th century most large European cities had recreated Roman sanitation, and by the late 20th century most European countries were trying to decrease pollution by industrial poisons. But at the same time, the growing populations of these cities required, and provided markets for, huge quantities of food, which increasingly tended to be produced by semi-industrial methods.

Some of this food came initially from the exploitation of populations of wild fish: but the supply of this apparently free resource was often unpredictable because the fish had to be caught far from land and in all weathers, and their imperfect management led to overfishing. In consequence, aquaculture has grown to provide a replacement source of marine protein, albeit sometimes by converting small fish into larger ones. And, just as was the case during the early development of human societies, this farming initially generated large amounts of waste, which accumulated in an environment hitherto thought to be pristine.

The metabolism of fin-fish is not dissimilar to that of humans, and, like people, fish produce solid and dissolved wastes. Waste food and faeces voided into the water tend to sink to the seabed. Many farmed fish are carnivores, and so must be fed a protein rich diet, which they use inefficiently compared with the herbivores and omnivores that are farmed on land. Consequently, they excrete dissolved compounds of nitrogen (especially, ammonia) and phosphorus (especially, phosphate) by way, mainly, of their gills. These processes are natural; the problems due to these wastes arise from intensive or semi-intensive farming, which takes in food from an extensive region but concentrates the waste in a much smaller area around a farm.

As an example, a farm stocked with 200,000 young salmon, and harvesting about a thousand tonnes of fish towards the end of a 2-year production cycle, uses about 1,200 t of feed made from 3,600 to 5,900 t of wild fish (according to conversion ratios in (Black 2001)). The food supply represents a share of the primary organic production of hundreds of square kilometres of sea. During the second year of the cycle the farm releases an amount of nitrogen, phosphorus, and faecal matter similar to that in the untreated sewage from several tens of thousands of humans. But whereas these people would inhabit at least a few square kilometres even in the most densely settled European cities, typical netpen farms of this size cover only a fraction of a square kilometre. Furthermore, whereas the most human and industrial wastes are now, in cities in the developed world, collected and treated before discharge, farm waste enters directly into the sea.

Although such wastes are in themselves natural, and so harmful only in excess, some mariculture results in the production of a second category of wastes. These are the man-made chemicals used to treat fish for disease, to make them grow faster, or to prevent seaweeds, seasquirts and barnacles from growing on fish cages. Speed-reducing *fouling* by these organisms has long been a problem for ships, and the success of the British Navy during the Napoleonic wars was partly due to the use of copper plating to prevent fouling of their wooden hulls (Rogers 2004). Copper is expensive, however, and can cause problems due to electrolytic corrosion, and there was a search for other compounds that could be applied to hulls in paint. The invention of the antifouling compound *tributyl tin, or TBT,* seemed to be a break-through. After several decades of use, however, it was found to be harmful to marine invertebrates, causing female dogwhelks to grow penises and farmed oysters to become mis-shapen (Readman 2005). It is now banned from use by fish farms and all small craft that anchor in coastal waters.

Thus, nutrients, organic matter and toxic pollutants have the potential to do harm to marine organisms. Their actual impact depends, however, on the environment into which these wastes are released. The next section looks at the properties of one type of environment much used for aquaculture, and uses this example of a water body to explain the idea of *an ecosystem.*

1.3 The Ecosystem in Loch Creran

The west coast of Scotland is cleft in many places with long arms of the sea. Called *loch* in Scots Gaelic (with the final *ch* a soft sound made in the back of the mouth), most are technically *fjords*: river valleys internally deepened by glaciers during the Ice Age and then flooded with salt water as the level of the ocean rose when the main ice sheets melted. For several millennia, these sheltered sea-lochs have provided highways and food sources to the people who lived in this otherwise unproductive and mountainous region. Now they are both a tourist attraction and a site for fish-farming, especially Atlantic salmon and mussels.

Halfway up this coast, the large fjord of the Firth of Lorne runs north-eastwards, along the line of the Great Glen fault that separates two ancient tectonic plates and continues to shake us locals with mini-earthquakes about once a decade. Big fjords often have little fjords, made by tributary glaciers, and the Firth of Lorne is no exception: loch Spelve, on the island of Mull, and on the mainland side, lochs Eil, Linnhe, Leven, Creran, Etive, Feochan and Craignish. All these have the characteristic feature of a fjord: a narrow and shallow entrance, with at least one deeper and wider basin inside. My friend Anton Edwards once wrote that although there is no such thing as a typical sea-loch, if you make lists of the Scottish saltwater lochs ranked in terms of their physical attributes, such as greatest depth, or freshwater inflow from the rivers discharging to their heads, then Creran comes close to the middle of most lists.

Seem from the top of a nearby hill, Creran looks like a lake: the winding channel that connects it to the Firth of Lorne is hidden behind a wooded hill (Fig. 1.1).

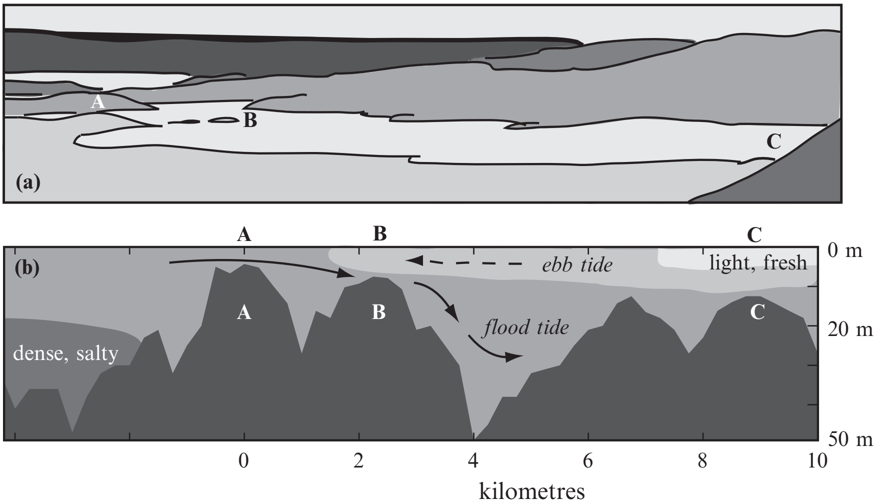

Fig. 1.1 A Scottish site for aquaculture: (a) sketch of loch Creran, looking west towards the larger fjord of the Firth of Lorne; (b) section, showing density and deduced circulation

But through this channel come pouring millions of cubic metres of salt water on each rising tide, and a slightly greater volume leaves on the ebb tide, swirling past small islands where seals lie and black birds perch on the lookout for fish. The outflow volume is greater because it must include the water added by rivers: in normal circumstances only a few percent of the tidal flow, but with a major effect on the circulation within the loch. Fresh water is less dense than salt water, and, where it mixes with seawater forms a lighter superficial layer that floats seawards, while the heavier saltwater, brought in by the tide, penetrates underneath.

This circulation renews water and oxygen within the loch, and creates good conditions for the growth of the fish and seabed animals that feed the seals and birds. On the seabed, there were once-abundant beds of the European oyster, and there still are extensive reefs made from the calcareous tubes of serpulid worms. Both oysters and serpulid worms are members of the **benthos**. Some benthic animals feed on organic matter within seabed mud, but the oysters and serpulids get food by filtering suspended particles. The most nutritious of these are the tiny floating algae of the phytoplankton, too small to be seen, as individuals, by the naked human eye. These micro-algae are well known as the "grass of the sea", the main marine source of organic food made by photosynthesis. When my colleagues and I studied it (Tett et al. 1985; Tett and Wallis 1978), Creran was typically rich in a variety of phytoplankters, especially those belonging to the group known as *diatoms*, which absorb dissolved silica from sea-water and use it to make glassy cases for their cells. The circulation of water through the loch provided a continuing source of compounds of nitrogen, phosphorus and silicon; and the layering created by the freshwater input allows phytoplankters to remain in a superficial layer that is well-lit by sunlight for much of the year.

Phytoplankton is not the only source of organic food in Creran: seaweeds are also important *primary producers*, and there is a further input of dead organic matter from rivers (Cronin and Tyler 1980; Tyler 1984). But I have described enough to make my point: that loch Creran is an *ecosystem*, a term invented by Roy Clapham in 1930, published by Arthur Tansley (1935) and defined by Eugene Odum (1959) as

> *any area of nature that includes living organisms and nonliving substances interacting to produce an exchange of materials between the living and nonliving parts…*

Formally, the *nonliving substances* form the *environment* and the *living organisms* form the (biotic) *community*; but a ecosystem is not simply *environment* plus *community* but also the interactions between and amongst them; it is both *structure* and *function* – the food web and how it works.

Thus, the interactions in loch Creran include the biogeochemical fluxes of organic matter and nutrients amongst the biota and between them and their surroundings; the effects of the serpulid reefs in stabilizing the seabed in Creran; the transport of animal as well as micro-algal plankton by currents; the addition of oxygen by algal photosynthesis and air–sea exchange, and its consumption by the respiration of all the animals and bacteria living in the waters of the loch or on or in its seabed. By analogy with human health, we can say that an ecosystem is healthy when all its parts are in good order and also when the interactions are in balance with the needs of the biota. This is a topic to which I'll return later – but for now, please note a significant difference between the health of a human – for whom the environment is something outside of the body and which is seen as a factor conducive to good or bad health, depending on whether air or water is clean or polluted – and the health of an ecosystem – which includes the state of the non-living part. Suppose we add a fish farm – either fin fish or shellfish – to an ecosystem such as Creran. Should we view the farm as bolted on to the outside of the ecosystem – potentially able to perturb it through waste products and liable to harm if some of this waste, for example, decays and consumes oxygen – or as an addition to the loch's ecosystem, participating in the *exchange of materials*? And what about the humans who operate the farm and truck in fishmeal caught in distant seas?

1.4 Aquacultural Pressures and Potential Impacts on Ecosystems

Any fish farm is a site of concentrated food production. Shellfish such as mussels take their food from the water flowing past them, and so one of their impacts on the ecosystem is the removal of the phytoplankton that forms much of this food. Depending on the extent of water movements, a mussel farm may harvest planktonic primary production from a wide area of sea – an area much greater than the extent of the mussel farm itself.

In contrast, the feed given to farmed salmon is largely made from other fish, caught in a different part of the ocean, but again harvesting the primary production of much wider area of sea than the extent of the fish farm. Think of both types of farm as the drain at the end of a bath, a vortex through which must flow large quantities of material. Both mussels and salmon draw oxygen from the water to support their metabolism of this food, and, because of the vortex effect, can potentially cause oxygen depletion – which would be fatal for the fish and shellfish. The way to avoid this is to site a farm in a region of strong water flow – which will also carry away the potentially toxic ammonia released by the animals' metabolism, and any other harmful dissolved substances such as those involved in ridding salmon of sea-lice or preventing fouling on nets.

However, although *the answer to pollution is dispersion and dilution*, the dilution of fish farm wastes has to be sufficient for undesirable ecological consequences to be avoided. It is, unfortunately, possible to site a farm in a region of flow sufficiently strong to avoid oxygen depletion or ammonia build-up around the farm, but insufficiently flushed to avoid the accumulation of wastes on a larger scale. Bearing this in mind, let us look at three types of potential ecological disturbance associated with fish-farming. Figure 1.2 exemplifies these in a fjord, but most can occur anywhere in the sea.

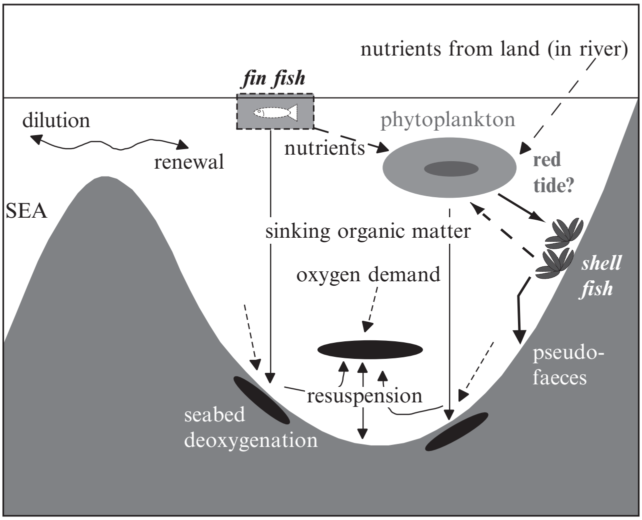

Fig. 1.2 Effects of aquaculture in a fjord

The first type of disturbance is a result of fall of fish faeces, uneaten food, and similar, towards the seabed. Water currents and eddies disperse these particles, and their "footprint" on the seabed depends on water depth and turbulence. In small amounts this organic matter provides food for benthic animals and demersal fish, but when it accumulates on the seabed, it can block the supply of oxygen to burrowing animals and can drive an increase in oxygen consumption by micro-organisms. It may be that all oxygen is removed from the water between sediment particles, leading to the replacement of aerobic bacteria (which release carbon dioxide as a product of metabolism) by anaerobic bacteria, whose by-products are methane, sulphur, and poisonous hydrogen sulphide. The effects of increasing organic input on the benthic fauna in fjords was systematically described by Pearson and Rosenberg (1976, 1978) in relation to the waste from wood pulp processing, and although fish-farm waste is more labile and nutrient-rich, it seems to have much the same effect – shown in simplified form in Fig. 1.3(a).

The second kind of potential disturbance is *eutrophication*, defined by OSPAR (2003) as

> *the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce* ***an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned****…*

These nutrients are the dissolved compounds of nitrogen and phosphorus – especially nitrate, ammonium and phosphate – which are necessary for the growth of photosynthetic organisms. Eutrophication thus defined is different from the effects of the organic matter needed by animals and by non-photosynthetic

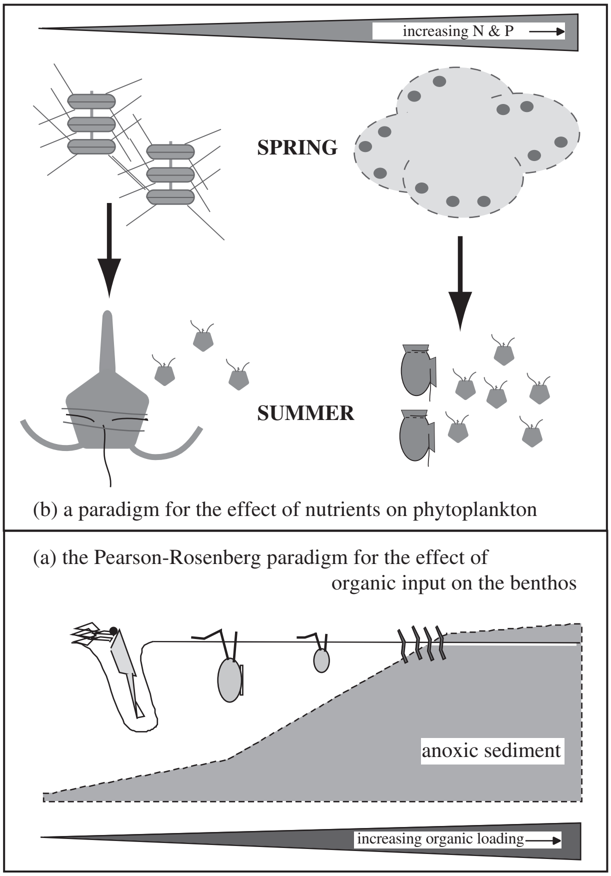

Fig. 1.3 Paradigms for disturbance: (a) Pearson–Rosenberg paradigm Pearson & Rosenberg (1976, 1978), for effects of organic waste, increasing in amount from left to right, leading initially to the loss of water-pumping animals (bio-irrigators) and finally to complete replacement of oxygen-requiring organisms by anaerobes; (b) an attempt, inspired by Margalef (1978) to schematize the phytoplankton response to anthropogenic nutrient enrichment of temperate waters; the diatom-(dino)flagellate seasonal succession is shown giving way to gelatinous colonial algae in the spring and to toxic dinoflagellates and small flagellates during summer

micro-organisms. The key distinction is that the growth stimulated by the mineral nutrients is accompanied by the photosynthetic release of oxygen, whereas growth on preformed organic matter consumes oxygen. Of course, the first may lead to the second, recycling the nutrient elements nitrogen and phosphorus back into their mineral forms, and consuming the oxygen released during photosynthesis. The problems associated with eutrophication typically come about when the coupling between the first and second parts of this natural cycle is weakened because of excess primary production and the formation, in the absence of sufficient grazing by planktonic or benthic consumers, of excess phytoplankton or seaweed biomass.

Thus, the harmful consequences that may result from nutrient enrichment include increasing frequencies and intensities of *Harmful Algal Blooms (HABs)*, including Red Tides, nuisance blooms causing foaming, toxic blooms that can kill farmed fish, and increased occurrences of incidents of shellfish-vectored toxins, such as those causing paralytic shellfish poisoning (Anderson and Garrison 1997). If blooms sink into deeper water, the decay of their biomass can cause oxygen depletion. Increased amounts of phytoplankton attenuate light more strongly, with the consequence that the growth of seaweeds and seagrasses may be retarded. Opportunistic green or brown seaweeds spread over seagrass meadows or over the slower-growing brown fucoid and laminarian seaweeds that are the natural flora of temperate seashores and the shallow sublittoral. Although green seaweed growth can be stimulated close to cages, eutrophication is a phenomenon that is more typical of water bodies, such as lochs or coastal seas, as a whole. It is thus distinct from the local impacts of particulate waste, although the *change in the balance of* pelagic *organisms* associated with eutrophication (Fig. 1.3(b)) can be likened to the changes caused by organic input to the benthos (Fig. 1.3(a)).

The third type of potential disturbance is that from chemicals that are used to prevent or treat fish illnesses or parasitical infections, to improve fish growth, or to prevent fouling of nets or farm structure. Let us look at two groups of such chemicals, starting with the compounds azamethiphos and emamictin benzoate, used to rid farmed salmon of parasitic sea-lice.

These lice are crustaceans that burrow under the scales of the fish, causing sores that irritate the salmon and offer a route for infection by pathogenic micro-organisms. Young lice are planktonic, and so can infect other farmed or wild salmon. For all these reasons, fish-farmers in Scotland are required to treat their fish to keep lice infestation to a minimum. The two chemicals are arthropocides – that is, they are intended to kill lice, which are members of the arthropod phylum, but not salmon, which are vertebrates.

The problem is that many members of the plankton are also arthropods, the group that includes insects, spiders and crustaceans. To be precise, the sea-lice are copepod crustaceans, as their planktonic larvae show, and so chemicals that kill sea-lice are also at risk of killing planktonic copepods and thus of damaging an important link in marine food webs. Azamethiphos, which is applied externally, is a greater hazard than emamectin, which is given to salmon in their food and reaches the lice by way of the fish bloodstream. However, some emamectin reaches the sediment in fish faeces and uneaten food, and here it may harm benthic crustaceans. Both the chemicals are degraded by light and oxygen, and can also be removed by adsorption on particles; and these processes augment dilution and dispersion in bringing concentrations below levels at which harm might result.

Whereas azamethiphos and emamectin are solely of human manufacture, and hence were never present in ecosystems before humans introduced them, the story about antifouling compounds is more complex (Readman 2005). These compounds are used to prevent the growth of bacterial slime and seaweed sporelings on nets and supporting structures. TBT, which did this effectively, was entirely synthetic, but is now banned. Modern paints and steeping liquids use compounds of copper, and sometimes zinc, which dissolve slowly in seawater, releasing ions of copper and zinc. It is these ions that are harmful to micro-organisms that might settle and grow on the netting or cage. Paradoxically, copper and zinc are needed in small amounts by living creatures, being essential for some biochemical reactions, and are toxic only at higher concentrations. So the challenge for the designers of anti-fouling materials is to ensure that they release sufficient copper etc to kill bacteria and algal spores close to the surfaces they are intended to protect, but without dissolving too quickly, which would increase the risk of wider harm and would require more frequent treatments.

Consequently, some manufacturers add "booster biocides" to augment the anti-fouling action. These include the synthetic chemical, copper pyrithione. However, research suggests that when zinc is present, the pyrithione part can swop from copper to zinc, resulting in zinc pyrithione. This compound, used in anti-dandruff shampoos and as a fungicidal additive for plastics, has been found to be highly toxic to copepods as well as planktonic micro-algae (Hjorth et al. 2006; Maraldo and Dahllöf 2004).

The last part of this story is that farmed fish need copper, and so it is added to their food, perhaps in unnecessarily large amounts that the fish excrete into the water or by way of their faeces; because of the latter, the seabed beneath fish cages may contain high levels of copper, which dissolves to increase the concentration of copper ions in the sediment pore waters, and which may diffuse back into the water column.

1.5 DPSIR and EQS

The ***DPSIR*** system breaks the ecosystem effects of pollutants into 5 steps. In this acronym, *D* stands for ***driver***, *P* for ***pressure***, *S* for ***state***, *I* for ***impact***, and *R* for response. The ***state*** is that of the ecosystem under consideration; the ***pressures*** are those generated by human activity whose change provides the ***drivers***. Thus the growth of salmon-farming is the ***driver*** that has led to increasing loading of Scottish fjords with farm waste, with consequential ***pressures*** on the fjordic ecosystems from organic matter, mineral nutrients, and chemicals. A build-up of particulate waste beneath a fish cage, with consequent death of larger sea-bed animals, exemplifies a highly visible ***impact***, and the ***response*** to this impact has been for society to impose more stringent conditions on the location and management of fish farms.

Environmental Quality Standards (EQS) have been used to set limits to pressures. The Water Framework Directive, which we will come to later, defines a standard as:

> *the concentration of a particular pollutant or group of pollutants in water, sediment or biota which should not be exceeded in order to protect human health and the environment.*

As an example, the current Scottish EQS for azamethiphos is 40 ng/L (SEPA 1997, 1998). In laboratory studies, 50% of lobster larvae exposed to an azamethiphos concentration of 500 ng/L died within 4 days. The EQS was set below this value in order to avoid any harm to free-living marine animals, taking into account the natural decay of the chemical when released into the water.

In the case of such toxic pollutants there is an obvious relationship between pressure and impact, and the aim is to avoid any such impact. In the case of pollutants such as nutrients, which cause problems only when in excess, the setting of EQS is more difficult. The aim, of course, is to avoid the undesirable disturbances associated with eutrophication or the smothering of seabed communities by particulate waste from fish farms. The European Urban Waste Water Treatment Directive (UWWTD) of 1991 concerns the prevention of pollution by discharges of sewage, but the causes of such pollution are the same wastes as those from fish farms: organic waste, biological oxygen demand, and compounds of nitrogen and phosphorus; and some aspects of the UK response to the UWWTD can be applied just as well to fish farms as to urban waste water outflows. (There are differences, of course: human waste is treated before discharge; fish waste is not.) The United Kingdom set up a "Comprehensive Studies Task Team" to define standards and evaluative procedures for UK estuaries and coastal waters. The team (CSTT 1997) suggested that:

> *Hypernutrification exists when winter values of nutrient concentrations, outwith any area of local effect, significantly exceed 12 mmol DAIN m^{-3} in the presence of at least 0.2 mmol DAIP m^{-3}… Hypernutrification should not, however, be seen as a problem in itself. It causes harmful effects only if a substantial proportion of these nutrients is converted into planktonic algae or seaweed.*
>
> *A region is potentially eutrophic only if the relative rate of light-controlled phytoplankton growth is greater than the relative water exchange rate plus the relative loss rate of phytoplankton by grazing; and the predicted summer maximum chlorophyll is greater than 10 mg chl m^{-3}… A region is eutrophic is observed chlorophyll concentrations regularly exceed 10 mg m^{-3} during summer.*

The acronym DAIN refers to "dissolved available inorganic nitrogen", a useful and precise way of mentioning those compounds of the element that are useful to phytoplankton and seaweeds – what I have named earlier as nitrate and ammonia. DAIP refers to "dissolved available inorganic phosphorus", for which the shorter abbreviation DIP or "dissolved inorganic phosphate" will do as well.

These CSTT proposals suggest that, in the case of nutrients, it is difficult to set simple EQS, because the impact resulting from a given pressure depends on conditions in the water body receiving the discharge. Sensitivity to pressure is the topic of the next section.

1.6 Ecohydrodynamics and Sensitivity to Pressures

Although laboratory experiments can, for example, measure the concentration of copper or zinc pyrithione that kills 50% of phytoplankton (Maraldo and Dahllöf 2004) or the amount of DAIN that must be added to generate a phytoplankton biomass in excess of the CSTT threshold of 10 mg chlorophyll m^{-3} (Edwards et al. 2003), the uncontrolled variability of conditions in the sea means that it is much harder to predict the impact of waste. For example, the food and faeces sinking from a small salmon farm in sheltered shallow waters might rapidly blanket the seabed beneath the farm, causing conditions to fall below those tolerable, whereas a larger farm moored in more turbulent and deeper waters might have no visible effect on the seabed, because the waste is dispersed by turbulence and spread over a wide area. However, the larger farm's waste has a greater potential to contribute to the widespread build-up of chronically harmful levels. Whereas the smaller farm may suffer from nutrient-stimulated seaweed growth on its cages, the water body containing the larger farm may suffer eutrophication because nutrients remain high for sufficiently long, and over sufficient extent, for phytoplankton to benefit from them.

Such considerations lead to two key ideas: first, that the sensitivity to waste of the waters or sea bed at a particular farm site, depend on ecohydrodynamic conditions at and around that site; second, that the impact of a particular environmental pressure depends on the spatial and temporal scale on which that pressure is applied. *Scales* are considered in the next section. *Sensitivity* can be roughly defined as the ratio of *impact* to *pressure*, and *ecohydrodynamics* refers to the physical conditions at a site and in a water body, and the chemical and biological conditions that would naturally occur under such conditions. An *ecohydrodynamic typology* provides a mean of classifying water bodies on the basis of such conditions. Tett et al. (2007) proposed a typology based on four key factors: lateral exchange; vertical mixing; illumination conditions; and the type and abundance of grazers.

The first distinction in the typology is that between open waters and partly enclosed coastal and transitional waters, called *Regions of Restricted Exchange*, or *RREs*. In RREs, exchange of water with the open sea is an important environmental condition; Tett et al. (2003a) compared a number of European fjords and barrier-protected bays in which the proportion of water exchanged each day varies from 2.5% (in the Swedish Himmer fjord) to more than 200% (in the Portuguese Ria Formosa) of the RRE's volume at mid-tide. The exchange rate for Creran lies between 0.1 and 0.3 d^{-1}. Clearly, well-flushed RREs can accept a greater loading of dissolved waste per unit surface area than can a poorly flushed water body, so long as the outside sea contains a lower concentration of the polluting substance.

The availability of light for photosynthesis is an important factor. Light does not penetrate far into water, because it is scattered by particles and absorbed by water itself, by chlorophyll and accessory photosynthetic pigments in phytoplankton, and by the dissolved substances than can give water a yellow or brown colour. The ***euphotic zone*** includes the part of the water column in which there is sufficient light for the growth of plants, seaweeds, micro-algae and photosynthetic bacteria; its depth reaches up to a hundred metres in clear ocean waters, such as parts of the Mediterranean, but may be only 1 or 2 m in some very turbid coastal waters. The next group of distinction in the typology arises from the relationship between the euphotic zone, the seabed, water column layers, and natural and human supplies of nutrients. A key distinction is that between waters in which the seabed is within the euphotic zone, allowing seaweeds, seagrasses or micro-algae to flourish, and those where it lies deeper, so requiring phytoplankton to provide the primary production. In the first case, nutrient enrichment may lead to replacement of seagrasses or brown seaweeds by green seaweeds or epiphytic micro-algae, and there will be concern if an increase in phytoplankton results in less light reaching the seabed. In the second case, the seasonal pattern of phytoplankton growth, and the ecosystem's sensitivity to nutrient enrichment, depends on seasonal patterns of water layering.

In the second case, we need to distinguish between waters that are well-mixed in the vertical, due to strong stirring by tidal or other currents, or by wind or surface cooling, and waters that are layered in density as a result of surface heating or freshwater input. The term ***pycnocline*** is used by oceanographers to refer to a zone of strong vertical gradient in density (due to temperature or salinity) that separates mixed layers. Phytoplankters growing above such a pycnocline are better illuminated, on average, than those in deep mixed waters. On the other hand, the upper layer tends to become depleted in nutrients during the main season of phytoplankton growth, and this constrains micro-algal growth. Nutrients added to such an impoverished layer can have a striking effect by fertilizing phytoplankton when there are few planktonic animals to eat the micro-algae. Organic matter produced during these blooms can give rise, later to an increased risk of deoxygenation when uneaten material sinks, and decays, below a pycnocline.

At the latitude of Scotland, there is generally too little light for phytoplankton production during the winter, and the typical pattern in coastal seas is that of a spring bloom as the surface of the sea is warmed by the sun and forms a distinct layer. Within this well-illuminated surface layer, algae can rapidly convert winter nutrients into biomass. This is, typically, followed by a summer period of low biomass because of nutrient exhaustion, and sometimes by an autumn bloom as nutrients are remixed into the surface water. In the Mediterranean, in contrast, the main seasons of phytoplankton growth are the autumn and Winter; in summer the surface layer is typically intensely nutrient-depleted, but there may be a subsurface layer of high chlorophyll. As demonstrated by loch Creran (Tett and Wallis 1978), layering (Fig. 1.1) resulting from freshwater input can extend the season of phytoplankton growth, unless the freshwater supply is so great that it brings the salinity down below a level tolerated by marine phytoplankton or flushes the algae from the system.

A final part of ecohydrodynamics takes into account the type of grazers on the primary producers. This is important in relation to eutrophication, for a poor coupling between producers and consumers can allow nutrient enrichment to stimulate a large increase in producer biomass – red tides of dinoflagellates, or blooms of opportunistic green seaweeds, for examples. In shallow waters, removal of pelagic micro-algae by water-filtering benthic animals can be important, but in deeper systems the benthos is passive: its members simply eat what sinks from the euphotic zone. Thus the efficiency of coupling in these waters depends on the numbers of protozoan microplankters and copepod and other mesozooplankters seeking micro-algal food. Algal blooms may be more likely if the growth of these animals is stunted by toxic pollutants. Conversely, adding a shellfish farm to a water body can artificially increase grazing.

1.7 Scales

Now let us consider the scales on which aquaculture can impact on ecosystems. These depend on a combination of the nature of the pressure, the dispersion characteristics of the water at and near the farm site, and the response time for the impact. The CSTT (1994, 1997) proposed that 3 scales be considered, applying to what the team called *zones A, B, and C* (Fig. 1.4). The key defining feature is the residence time of neutrally buoyant particles within the zone: citrus fruits can serve as suitable, and easily seen, particles, and so I like to imagine a modern Nell Gwyn tipping her basket of oranges into the sea from a farm, so that we can ask where are most of the oranges after a few hours (zone A scale), a few days (zone B) or a few weeks (zone C).

The ***zone A*** scale is that the water volume and sediment area immediately influenced by a fish farm, and corresponds to the *mixing zone* at the end of a pipe

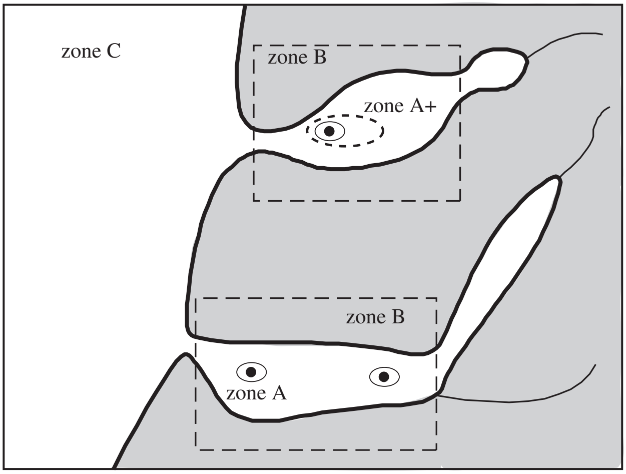

Fig. 1.4 Illustrated the 3 scales proposed by the UK Comprehensive Studies Task Team (CSTT). Zone A is the farm scale; it includes the part of the seabed that receives organic waste sinking from a farm and the part of the water column in which wastes and pollutants remain for a few hours. In tidally active waters, this water column zone is shown as A+. Zone B is the water body scale, and is exemplified by the main basin of loch Creran. Zone C is the regional scale

discharging waste into the sea, within which concentrations are allowed to exceed those specified by a *far-field* EQS. In general, it is easy to see benthic impact (Nickell et al. 2003) but difficult to detect pelagic impact on this scale, although it is sometimes possible to find a local increase in ammonia and a decrease in dissolved oxygen (Gowen and Bradbury 1987), and, in the case of shellfish farms, a local decrease in chlorophyll.

In the simple case of a fish farm in waters without tides or residual currents, the zone A scale is shown by the footprint of the cage on the sea, i.e., the area impacted by sinking waste, and a relatively small volume of water around the farm, the dimensions of which are set by the intensity of eddy diffusion. Under these unfavourable conditions the scale's dimensions are unlikely to exceed twice those of the farm. Now let us add a persistent current, which will transport the imaginary oranges in a downstream plume, broadening as it moves away from the farm. If the main flows are tidal, the oranges will move in an ellipse, returning after one complete tide to somewhere near their starting point, so that in this case, zone A for dissolved waste may be several kilometres long. We may take the (slightly over) 12 hours of a tidal cycle in NW European waters as the upper limit to the zone A timescale, and on this timescale it is impossible for added nutrients to impact on the plankton, although fast-acting chemical toxins may harm plankton before they are diluted by dispersion outside the zone. In order to apply this idea to non-tidal waters, such as those in the Mediterranean, we keep the half-day timescale and consider the limits of the zone in the water column as that reached by the oranges during this time. Unless the farm is sited in very energetic waters, the benthic footprint will likely be obvious, and smaller than the pelagic zone A.

The main basin of loch Creran provides an example of a stratified ***zone B*** scale water body and a region of restricted exchange. The residence time of water within this basin has been estimated as about a week (Tett 1986), although the contents of the surface layer leave the loch more quickly, within about 3 days, because of the freshwater driven, tidally enhanced, circulation described earlier. Such residence times are sufficient for nutrients to turn into planktonic algae before the latter are flushed out of the loch, and it is this, and the existence of stratification, that makes the loch potentially sensitive to the effects of nutrient enrichment. Extra growth of phytoplankton might be controlled by the grazing of the abundant seashore and seabed animals in Creran, and by the pelagic protozoans found in the water column. Except during times when benthic animals release their larvae into the water, the effect of crustacean zooplankton is small, because these animals tend to get flushed from Creran before they can complete their life cycles within the loch.

The Firth of Lorne, with which loch Creran exchanges, is a much larger body of water. The residence time of this water is not well known, but it is probably in the order of weeks or longer – sufficiently long for nutrients to become phytoplankton and then be grazed and recycled. Thus it is an example of a ***zone C*** scale water body, and provides the *boundary conditions* for loch Creran – that is to say, the water that enters Creran from the Firth already contains a certain amount of nutrients and phytoplankton, depending on the season, and enrichment or grazing within the loch will add to, or subtract from, these incoming concentrations. Thus it may be as important to control nutrient levels of the Firth of Lorne as it is to restrict enrichment within loch Creran. Indeed, we know from the results of a mathematical model that only during the summer, when nutrients are scarce in the Firth of Lorne, does farm input make an important contribution to Creran DAIN and phosphate (Laurent et al. 2006).

Fortunately, the waters of the Firth of Lorne are in a largely pristine condition, their moderate nutrient concentrations being set mainly by natural processes in the sea to the west of Scotland. Fish farms may, of course, become sufficiently to increase nutrients even on this larger scale. The region called the Minch, between the Scottish mainland and the island chain of the Outer Hebrides, has a sea area of about 10,000 km^2. The production of 64,000 t of salmon may have increased the concentration of DAIN and DIP in summer 1999 by a few percent (Tett and Edwards 2002), a scarcely measureable amount. Nevertheless, concerns about the effect of a greater enrichment may set an upper limit to the size of the industry here.

The Mediterranean Sea, being oligotrophic, might be considered at greater risk from enrichment, in that it takes only a little anthropogenic nutrient to double the naturally lowconcentration in each cubic metre of seawater. However, the Mediterranean is large; recent calculations suggest that input from fish farms will increase the total nutrient stock of the sea by at most 1%, whereas total human-driven inputs might double it (Karakassis et al. 2005).

1.8 Regulation of Pollution and Conservation of Species

At the core of the DPSIR scheme are the links between ***pressures***, ***states*** and ***impacts***. As humans became aware that the sea was neither an infinite garbage can for wastes nor an inexhaustible source of fish (McIntyre 1995), our societies began to legislate either to *prevent pollution* of the environment – corresponding to the regulation of ***pressure*** – or to *protect* certain animals or plants – corresponding to the prevention of ***impacts*** on these organisms. This was initially a piecemeal approach, which I will illustrate for the case of Scotland with two United Kingdom Laws – the *Control of Pollution Act (COPA)* of 1974, and the *Wildlife and Conservation Act* of 1981 – as these have used by the Scottish Environment Protection Agency (SEPA) to minimize the environmental impact of salmon-farming and to maintain water quality for shellfish.

My account greatly simplifies the complexities of a legal framework used to apply these UK laws in the separate, and different, jurisdictions of each part of the Kingdom. In most cases the generalities of the Acts of the UK Parliament (and, since 1999, also of the Scottish Parliament) are interpreted by detailed "Regulations" which are also commonly used to implement European Directives. Since the UK's accession to the European Community (as it was then called) on 1 January 1973, it has acquired (Graham 2002), *legal commitments to meet individual directive requirements that, in general, are transposed into UK law by means of regulations or other forms of secondary legislation issued as statutory instruments. A regulation identifies the competent regulatory authority and the actions required of it in order to achieve the directive's requirements. … It is primarily regulations and directions passed by the UK or Scottish Parliament, which impose obligations on SEPA, as the competent authority, to deliver the objectives and standards so transposed from an EC Directive*.

The "Control of Pollution" Act (COPA) of 1974 marked the beginning of marine pollution control in the UK, although it took a decade to implement fully. The main regulatory tool is the "consent to discharge" from a "point source" such as a waste pipe or a fish farm. According to its web site (SEPA 2005a),

SEPA has a duty to control discharges to surface waters and groundwaters *[in Scotland]*, including tidal waters out to the three-mile limit. SEPA does this by issuing a legally-binding consent to discharge under the Control of Pollution Act 1974….. Where consent is granted this will include specific conditions to limit the effects that the discharge may have upon the receiving environment. Monitoring will be carried out by the discharger and SEPA to ensure that the impacts of the discharge remain within acceptable levels.

Thus, anyone wishing to establish or extend a salmon farm in these waters must, amongst other legal requirements, make an application for a consent to discharge the waste from the farm. Then (SEPA 2005b),

SEPA will impose consent limitations on the maximum permitted fish biomass which may be held at any time. This is designed to minimise accumulation of organic wastes on the sea bed to prevent anoxic and polluted sediments and associated deleterious effects on the normal benthic fauna outwith the allowable zone of effects. In certain instances to protect important wild salmonid stocks, SEPA will limit the biomass to that which can be treated at the site using an authorised sea lice medicine [without exceeding environmental quality standards for these medicines]. … *SEPA will* [also] *limit consented biomass to ensure that the receiving water will not be* [at risk of eutrophication].

An *allowable zone of effects*, or *AZE* is a small region beneath fish cages where some impact is allowed. SEPA accepts

that a certain area immediately below and around the cages may experience carbon accretion to a level which may change the community structure of sediment fauna. Within this AZE quality standards ensure a minimum number of sediment re-workers will be available to breakdown wastes and prevent total anoxia developing.

Two salmon farming sites have been consented in loch Creran, each of 1,500 t maximum biomass; however, only one site is available at a time, because each site is required to lie fallow for two years between use, in order to allow recovery of the benthos in the AZE.

The "Wildlife & Conservation" Act of 1981 has been used to implement the European "Habitats" Directives of 1992/1997 and the "Birds" Directive of 1979. It protects wild birds, and certain other animals, and plants that have been officially listed, together with designated sites. UK regional conservation agencies, exemplified by Scottish National Heritage (SNH), work under this law. The agency's web site (SNH 2006b) explains that

> *Special Areas of Conservation (SACs) are areas designated under the European Directive commonly known as the 'Habitats' Directive. Together with Special Protection Areas, which are designated under the Wild Birds Directive for wild birds and their habitats, SACs form the Natura 2000 network of sites. SNH acts as the advisor to Government in proposing selected sites for Ministerial approval as possible SACs. SNH then consults with… owners and occupiers of land, local authorities and other interested parties …* [and] *negotiates the longer term management of these sites. Following consultation, SNH forwards all responses to Scottish Ministers who then make a decision about whether to submit the site to the European Commission as a candidate SAC. … sites which are adopted by the Commission become Sites of Community Importance (SCIs), after which they can be finally designated as Special Areas of Conservation by national governments. All candidate SACs in Scotland were approved by the European Commission as SCIs on 7 December 2004. Scottish Ministers then formally designated all these sites as Special Areas of Conservation on 17 March 2005.*
>
> *Under Regulation 33(2) of the* ***Habitats Regulations*** *once a marine area becomes a designated SAC (European marine site), SNH is obliged to advise other relevant authorities as to a) the conservation objectives for that site, and b) any operations which may cause deterioration of natural habitats or the habitats of species, or disturbance of species, for which the site has been designated.*

Loch Creran have been designated as a SAC because of the

> *biogenic reefs of the calcareous tubeworm* Serpula vermicularis, *which occur in shallow water around the periphery of the loch. The species has a world-wide distribution but the development of reefs is extremely rare: Loch Creran is the only known site in the UK to contain living S.* vermicularis *reefs and there are no known occurrences of similarly abundant reefs in Europe. Biogenic reefs of the horse mussel* Modiolus modiolus *occur in the upper basin of the loch. M. modiolus reefs are an important element of Scotland's marine biodiversity, and are considered to be habitats of high conservation value.*

SNH's advice about Creran (SNH 2006a) includes the following comments:

> *Finfish farming has the potential to cause deterioration of reef habitats and communities through changes in water quality, smothering from waste material, physical disturbance (in the case of rocky reefs), and physical damage (in the case of more fragile biogenic reefs) from mooring systems. There is also potential for accidental introduction of new non-native species and increasing the spread of existing non-native plants and animals…*
>
> [Shellfish farming] *has the potential to cause deterioration of the reef habitats and communities through physical damage (e.g. installation of mooring blocks and continued scouring by riser chains) and changes in community structure caused by smothering from pseudo-faeces (undigested waste products) and debris (including dead shells) falling from the farm. There is also potential for accidental introduction of new non-native species and increasing the spread within the UK of existing non-native plants and animals… through importation and translocation of shellfish stocks.*
>
> [In both cases,] *invasive species have the potential to cause deterioration of the qualifying interest by altering community structure and quality. The … environmental effects* [associated with aquaculture] *are usually localised but the reduced water exchange within Loch Creran may exacerbate these effects and cumulative impacts should be considered.*

It was also noted that domestic and commercial effluents (whether treated or untreated) have

> *the potential to cause deterioration of reef habitats and communities. This would be through the effects of pollution and/or nutrient enrichment, which may cause subsequent changes in community structure* [of the reef].

Some of this advice has to be taken into account when permission for fish farms or other new developments is given by the planning departments of local government: it would certainly prevent farms being sited over reefs, or where their particulate wastes might accumulate on the reefs. An Environmental Statement, submitted as required by the Environmental Impact Assessment (Fish Farming in Marine Waters) Regulations 1999, should bring to light potential impacts of this sort. SEPA has a role to play, both at this stage and during the operation of the farm, as it is (Graham 2002):

> *a 'relevant authority' for European marine sites in Scotland, which are any SACs or SPAs that extend below the mean low water mark of spring tides. SEPA must, as a relevant authority, participate with other relevant authorities in drawing up a single management scheme for any European marine site where any relevant authority considers that one is necessary.*

Shellfish farming is much less strictly regulated, because it is not seen as producing a point source discharge. Instead, the industry is protected by the Shellfish Waters Directive of 1979, and much of loch Creran has been designated as a *Shellfish Growing Water* (SEPA 2004) under this Directive and the *Surface Waters (Shellfish) (Classification) (Scotland) Regulations 1997*. It is thus subject to monitoring by SEPA to ensure compliance with the standards set for metals and organohalogens in the water column and shellfish, faecal coliform bacteria in the shellfish, and dissolved oxygen. The aim is to protect the shellfish from environmental pressures and not to protect the rest of the ecosystem from the shellfish.

In summary, although some of the legislation discussed in this section takes account of links between pressures and impacts, the legal emphasis has been on polluting substances and their effects on particular commercial organisms or rare habitats; there is little of the general concern with the state of aquatic ecosystems that lies at the heart of the ***ecosystem approach***, the topic of the next section.

1.9 The Ecosystem Approach

The ***ecosystem approach*** can be seen, empirically, as a strategy for joined up management of the natural world, and scientifically, as arising from a modern understanding of community ecology and the interconnected processes within ecosystems. A web page of the UK Joint Nature Conservancy Council (JNCC 2004) provides a summary of the empirical view.

> *The phrase 'ecosystem approach' was first coined in the early 80s, but found formal acceptance at the Earth Summit in Rio in 1992 where it became an underpinning concept of the Convention on Biological Diversity, and was later described as: 'a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way.'*

> *Ecosystem-based management is currently a highly topical issue and is being widely discussed in the context of fisheries management. Introduction of the new Common Fisheries Policy (CFP) in January 2003 focused on this approach as the way forward to a sustainable fishing industry. Marine fisheries are one of the remaining examples of human endeavour involving the direct exploitation of wild animal populations. Fisheries are dependent on the productivity of the ecosystem, and fisheries have an effect on, and are affected by, the supporting ecosystem of the target species. It, therefore, follows that prudent and responsible fisheries management should take account of the profound interactions between fisheries and their supporting ecosystem.*
>
> [However,] *ecosystem-based management is not about managing or manipulating ecosystem processes, something that is clearly beyond our abilities. Rather, ecosystem-based management is concerned with ensuring that fishery management decisions do not adversely affect the ecosystem function and productivity, so that harvesting of target stocks (and resultant economic benefits) is sustainable in the long-term. Traditional systems of management, which have tended to focus on individual stocks or species, have not achieved this objective and consequently the economic activity that the ecosystem supports has become compromised.*

To my mind, this account falls short in several ways. First, it tends to suggest that the purpose of ecosystems is merely to produce food, or other services, for humans: it may be prudent to take account of the dynamic interconnections, but they are not valuable in themselves. Second, it is quite evidently not beyond human abilities to manipulate ecosystem processes. The matter at issue is, of course, to manage ecosystems wisely – at least in our own interests, but also, I believe, in the interests of all the creatures within them, and perhaps also in the interests of ecosystems as “emergent systems” whose properties are greater than the sums of their parts. My view is that an “environmental ethic” is also practical: we can only ensure sustainability if we treat all organisms and natural systems as having “interests worthy of consideration” (Johnson 1991).

My standpoint is close to that summarized by Miller’s (2006) account of one millennia-old strand of Chinese thought, that of Daoism.

> *Daoists view morality in medical terms: goodness consists of the optimal health of a system comprised of various interdependent subsystems. This medical concept of virtue can… be useful in constructing an ecological ethics, one that recognizes that humans cannot act for their own good without considering the overall health of the ecosystems in which they are embedded.… the ideal state is achieved through embodying the complex transformative power of nature rather than denying it.*

Such emphasis on “connectedness” does have its own intellectual pitfalls, exemplified by the false “science” of astrology, based on the notion of connection between the human microcosm and the astronomical macrocosm. Nevertheless, I think that most of our present-day ecological science is well grounded in Enlightenment rationality and scientific methods, and the idea of ecosystem health is, at the very least, useful for devising monitoring programmes. I will return to this idea later.

It may be that the western, utilitarian approach, grows from our biblical heritage. In Genesis 1:26 it is written:

> *And God said, Let us make man in our image, after our likeness: and let them have dominion over the fish of the sea, and over the fowl of the air, and over the cattle, and over all the earth, and over every creeping thing that creepeth upon the earth.*

Hence, humans have souls as well as sentience and so are qualitatively different from other living things, and indeed may be seen as inhabiting Earth only briefly whilst on their way to heaven or hell; they are distinct from the rest of Nature as well as entitled, perhaps even required, to look after it as well as use it. As I wrote above, managing ecosystem processes is clearly within our ability: humans have been doing it for millennia. The problem is that we have often done it badly and unintentionally. Thus, although we might look at present-day environmental problems in China and wonder what has happened to the Daoist ideal, I prefer the idea that we are embedded in the ecosystem, and will sink or swim with the rest of Nature, rather than the idea that a better world awaits us somewhere else.

I should not claim that there are clear-cut distinctions between the religious traditions. The relationship between Daoism and science is complex (Ronan and Needham 1978). The Christian tradition has included St Francis of Assisi and the romantic poet, Coleridge, who write the *Rime of the Ancient Mariner* in 1798. These lines, taken from near the end, sum up his philosophy, which seems to place humans on the same plane as the rest of creation:

He prayeth well, who loveth well
Both man and bird and beast.
He prayeth best, who loveth best
All things both great and small;
For the dear God who loveth us
He made and loveth all.

I like to think that if St Francis had had a microscope, he would have loved nematode worms as much as birds, and, indeed, the whole of the magnificent "tree of life" that is being revealed by nucleic acid sequencing studies. Whether or not one accepts the theologies of Coleridge or the saint, the idea that we humans are made of the same stuff as the rest of creation is one to cherish, I believe, both for its own sake and because it may help prevent *Homo sapiens* from going extinct.

And that is as much of a sermon as I wish to offer in this chapter. Now to return to more mundane considerations of how such an ethic can be turned into regulatory and management practices.

1.10 The Water Framework Directive

As already mentioned, there are hints of an integrated approach to ecosystems in earlier laws, but it is in the "***Water Framework Directive***", or ***WFD*** that the approach begins to be clearly visible. The WFD is formally entitled *DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL, of 23 October 2000, establishing a framework for Community action in the field of water policy*, and Article 1 states that:

The purpose of this Directive is to establish a framework for the protection of inland surface waters, <u>transitional waters, coastal waters</u> and groundwater which:

(a) prevents further deterioration and <u>protects and enhances the status of aquatic ecosystems</u> …

(c) aims at enhanced protection and improvement of the aquatic environment, inter alia, through specific measures for the progressive reduction of discharges, emissions and losses of priority substances and the cessation or phasing-out of discharges, emissions and losses of the priority hazardous substances;…. and thereby contributes to:… the protection of territorial and marine waters, and… achieving the objectives of relevant international agreements, including those which aim to prevent and eliminate pollution of the marine environment,… with the ultimate aim of achieving concentrations in the marine environment near background values for naturally occurring substances and close to zero for man-made synthetic substances.

The first part that I have underlined refers to ***transitional waters*** (those *substantially influenced by river flow*, hence, typically, estuaries) and ***coastal waters*** (extending at least to 1 nautical mile from a coastal baseline, to 3 nautical miles in Scotland). These are the waters relevant to marine aquaculture as considered in this chapter. In addition, however, the Directive's protection of rivers, lakes, and their catchments, should improve the quality of discharges to estuaries and coastal waters, and so improve the background conditions here, to the advantage of aquaculture.

The third group of underlined words concerns the reduction of environmental pollution. In this respect, the WFD may be seen simply as intensifying earlier legislation, such as that of the UK's COPA or the Dangerous Substances Directive; but it goes beyond the use of experimental toxicology to set values for EQS. Notice the distinction between the "man-made synthetic" substances, and "naturally occurring" substances that are enhanced in wastes. The former are to be, ultimately, excluded from seawater; the latter should not be allowed to exceed "background" values by very much. The distinction can be made from the *Indicative list of the main pollutants* provided in Annex VIII of the WFD:

Man-made synthetics: *1. Organohalogen compounds and substances which may form such compounds in the aquatic environment. 2. Organophosphorous compounds. 3. Organotin compounds. 4. Substances and preparations, or the breakdown products of such, which have been proved to possess carcinogenic or mutagenic properties or properties which may affect steroidogenic, thyroid, reproduction or other endocrine-related functions in or via the aquatic environment. 5. Persistent hydrocarbons and persistent and bioaccumulable organic toxic substances. 6. Cyanides. 7. Metals and their compounds. 8. Arsenic and its compounds. 9. Biocides and plant protection products.*

Naturally-occurring substances: *10. Materials in suspension. 11. Substances which contribute to eutrophication (in particular, nitrates and phosphates). 12. Substances which have an unfavourable influence on the oxygen balance (and can be measured using parameters such as BOD, COD, etc.).*

Of course, it may be necessary to be a little more subtle than I have been. For example, some copper compounds occur naturally in seawater, whereas others, such as copper pyrithione, are synthetic.

The second underlining, referring to *the status of aquatic ecosystems*, highlights the ecosystem approach. In fact, the WFD implements the approach in two main ways: through the management of river basins (and their coastal waters) as a whole, including the joint consideration of point and diffuse sources of nutrients; and through the ecological component of quality status. ***Quality status*** is defined in article 2 in the following terms:

17. "Surface water status" is the general expression of the status of a body of surface water, determined by the poorer of its ecological status and its chemical status.

18. "Good surface water status" means the status achieved by a surface water body when both its ecological status and its chemical status are at least "good"

21. <u>"Ecological status" is an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters, classified in accordance with Annex V.</u>

24. "Good surface water chemical status" means the chemical status required to meet the environmental objectives for surface waters established in Article 4(1)(a)....

As before I have underlined the key point and novelty: the focus on ecosystem structure and function. Details are given in Annex V, which is the longest single part of the Directive and (in my view) provides its beating heart. The Annex defines *ecological status* as consisting of *biological elements, physico-chemical elements supporting the biological elements* and *hydromorphological elements supporting the biological elements*. The *biological quality elements* for transitional and coastal waters are: *phytoplankton; macroalgae and angiosperms; benthic invertebrate fauna;* and *fish fauna* (in transitional waters only). Table 1.1 presents some general

Table 1.1 Some definitions of quality, from the Water Framework Directive: (a) Annex V section 1.2. Normative definitions of ecological status classifications: Table 1.2. General definition for rivers, lakes, transitional waters and coastal waters

Status	General definition
High	There are no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements for the surface water body type from those normally associated with that type under undisturbed conditions. The values of the biological quality elements for the surface water body reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion. These are the type-specific conditions and communities.
Good	The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions.
Moderate	The values of the biological quality elements for the surface water body type deviate moderately from those normally associated with the surface water body type under undisturbed conditions. The values show moderate signs of distortion resulting from human activity and are significantly more disturbed than under conditions of good status.
Poor	Waters showing evidence of major alterations to the values of the biological quality elements for the surface water body type and in which the relevant biological communities deviate substantially from those normally associated with the surface water body type under undisturbed conditions, shall be classified as poor.
Bad	Waters showing evidence of severe alterations to the values of the biological quality elements for the surface water body type and in which large portions of the relevant biological communities normally associated with the surface water body type under undisturbed conditions are absent, shall be classified as bad.

The *biological quality elements* for transitional and coastal waters are: *phytoplankton*; *macroalgae and angiosperms*; *benthic invertebrate fauna*; and *fish fauna* (in transitional waters only).

Table 1.1 Some definitions of quality, from the Water Framework Directive, continued: (b) Annex V, section 1.2.4. Example of standards for *physico-chemical quality elements*, in *coastal waters*

Status	General conditions	Specific synthetic pollutants	Specific non-synthetic pollutants
High	The physico-chemical elements correspond totally or nearly totally to undisturbed conditions. Nutrient concentrations remain within the range normally associated with undisturbed conditions. Temperature, oxygen balance and transparency do not show signs of anthropogenic disturbance and remain within the ranges normally associated with undisturbed conditions.	Concentrations close to zero and at least below the limits of detection of the most advanced analytical techniques in general use.	Concentrations remain within the range normally associated with undisturbed conditions (background levels…).
Good	Temperature, oxygenation conditions and transparency do not reach levels outside the ranges established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6[but not required to be below background levels] without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	
Moderate	Conditions consistent with the achievement of the values specified… for the biological quality elements [at moderate status].		

definitions of ecological and physico-chemical status. In essence, the ecological status of a water body is high when its phytoplankton, seaweeds or seagrasses, and benthos, all appear to be in a natural condition.

Figure 1.5 is a flow diagram to show how a regulator might apply the Directive. The starting point is the definition of water bodies and the identification of the type to which each belongs; then the present quality status of each water body identified in relation to a “type-specific reference condition” as *high* (the same as a water body in the reference condition), *good* (acceptable), *moderate*, *poor*, or *bad*. If the status is worse than *good*, ***Programmes of Measures*** must be implemented in order to improve the quality status, and monitoring programmes put in place to check on this. The WFD is also an integrating framework, bringing together provisions from earlier directives, including the UWWTD and the Habitats Directive. Any special requirements of these directives are dealt with by the concept of *Protected Areas* within which additional management might be

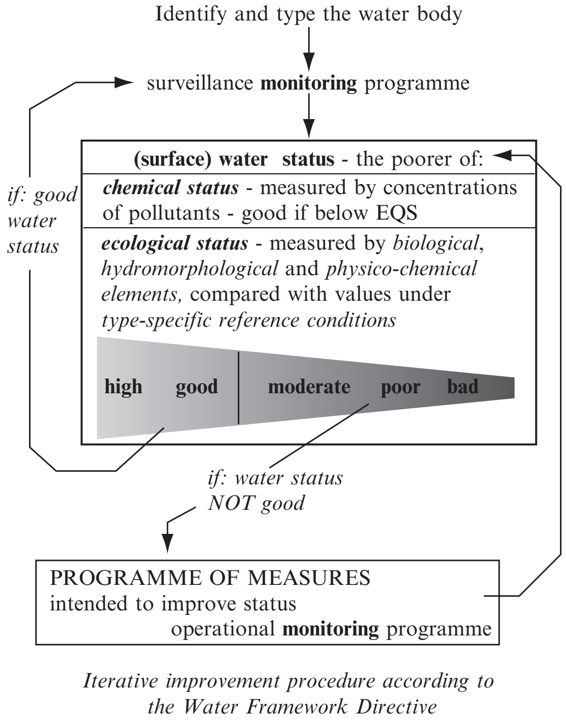

Fig. 1.5 Flow diagram for the operation of the Water Framework Directive – showing the relationship between the objective of maintaining good status, programmes of measures, and monitoring

needed. The timetable incorporated into the WFD requires one complete cycle of evaluation and management to be completed by the end of 2015, and some of the steps have already been carried out.

In Scotland, our (regional) parliament, which met in 1999 for the first time since 1707, used newly devolved powers to pass the *Water Environment and Water Services (Scotland) Act (2003)*, summarized as a law that, amongst other objectives, *make provision for protection of the water environment, including provision for implementing European Parliament and Council Directive 2000/60/EC*. This law gives the Scottish Environment Protection Agency (SEPA) the responsibility of drawing up the *River Basin Management Plans* required by the Directive to report *pressures* on water bodies, the existing quality status, protected areas, monitoring plans, and programmes of measures. Local authorities, analogous to municipalities or counties in other parts of Europe, must liaise with SEPA and take account of the WFD, and of programmes of measures, when giving permission for new building works. It is expected that some of the management measures will result from consent, because the WFD explicitly requires "stakeholder involvement", and that some will be enforced by SEPA using its “consent to discharge” powers under COPA, strengthened and modified in the *Water Environment (Controlled Activities) (Scotland) Regulations 2005* which provide for registration and licensing of discharges.

1.11 The WFD and Aquaculture in Loch Creran

In this section I am going to use Loch Creran as an example of how the WFD might come to bear on aquaculture. Some of my account is factual, and draws on material published by SEPA concerning its implementation of the WFD in Scotland. Some, however, must be conjectural, both because River Basin Management Plans are not due for publication until 2009, and because the Creran river basin, including the loch and adjacent coastal water, is only one of many such basins on the west coast of Scotland, and detailed plans by water body are initially only available for the “protected areas” within each. Nevertheless, I will roughly stick to the format set out for plans in WFD Annex VII, and will include: (i) a description and typing, including identification of reference conditions; (ii) a summary of significant pressures and impacts; (iii) a list of protected areas; (iv) a description of monitoring networks; (v) a list of specific environmental objectives; and, (vi) a summary of the “programmes of measures” required to achieve these objectives. All, of course, with the focus on aquaculture.

The first step in the application of the WFD in Scotland was the identification in 2003 of River Basin Districts, defined in the Directive as: *the area of land and sea, made up of one or more neighbouring river basins together with their associated groundwaters and coastal waters, which is… the main unit for management of river basins*. Most waters in Scotland, including loch Creran, fall into a single “Scotland” RBD. In contrast with many parts of continental Europe, where RBDs correspond to the catchments and coastal waters of single large rivers, the Scotland RBD includes many rivers, especially on the west coast, where rainfall is heavy and short steep rivers discharge into sea-lochs. Hence the Creran river basin and associated coastal water is but a small part of the Scotland RBD, and receives no specific description in the account so far published of the environmental features of the Scotland district. So the reader can turn to the description of loch Creran earlier in this chapter.

Completion of part (i) requires identification of the type of water body exemplified by loch Creran, so that reference conditions can be specified. Annex II of the Directive sets out the principles for two (alternative) typologies, and the UK, in collaboration with the ROI, has implemented these principles as a set of types for the coastal and transitional waters around our islands (UKTAG 2003). Creran can thus be identified as a coastal water of type 12, a “deep sea-loch” in the “Atlantic Ocean” ecoregion of Annex XI of the WFD. It is a coastal water because its depth- and time- averaged salinity is above 30 and hence close to that of seawater (in contrast to transitional waters, in which the mean salinity is less than 30), and a “deep fjord" because its greatest depth exceeds 30m. The adjacent loch Etive, receiving much more freshwater, has been identified as a 'transitional sea-loch' (transitional water type 5). Although there are some sea trout farms in Etive, the low mean and strongly fluctuating salinities make it less good for farming the salmon than the higher mean salinity of the coastal water fjords. In contrast, Etive has a good reputation for mussels, because the intermittently low salinities reduce fouling.

Table 1.2 illustrates the UK/ROI typology for the three fjordic types found in Scotland, and gives details both of the physical conditions that define the type, (UKTAG 2003) and of the proposed reference conditions (UKTAG 2004).

Protected areas in Creran include the serpulid reefs which are a SAC, and the *Shellfish Growing Waters* that occupy the main basin of the loch. SEPA's published description of the shellfish waters (SEPA 2004) gives the following for "land use and potential diffuse pollution sources":

> *The predominant land use is coniferous forestry but there is some extensive livestock agriculture on the north and far western shores. The main freshwater inflow is the River Creran, draining both forest and moorland. Loch Creran is remote from centres of population and is popular with visitors, particularly in the summer months.*

Point-source discharges include those from about 50 private houses, and the consented, major discharges from a fish farm and a fish processing factory. Laurent et al. (2006) used a mathematical model to show that the nutrients from the fish farm could make a significant contribution during summer, when the concentrations in the inflow from the Firth of Lorne are low. Nickell et al. (2003) found high organic loading and oxygen demand immediately beneath the farm, falling off rapidly at 60 m distance and returning to normal background levels for shallow coastal waters at 2 km from the farm. Only in the sediment immediately beneath the farm was the benthic community composition grossly perturbed.

Creran's waters are monitored for shellfish purposes from two sites, one near the mouth and one near the head of the main basin. In addition the river Creran is sometimes sampled for nutrients above its discharge into the upper basin: concentrations are typically low, as might be expected in runoff from granitic rocks and unimproved acidic grassland.

SEPA (2004) reports that:

> *In 2002, all samples from both monitoring sites met all shellfish waters imperative and guideline environmental quality standards. Biannual sampling is continuing for metals and organochlorines in waters along with monthly sampling for T, Sal, DO and pH at South Creagan and North Shian. Mussels will be sampled annually for organohalogens and metals at North Shian. This site will also be monitored quarterly for faecal coliforms in mussels and in addition, collection of mussels for TBT and PAH analysis will begin in 2004 as part of a SEPA Environmental Improvement Plan.... SEPA will continue to pursue a policy of no new discharges of sewage effluent to designated waters, to avoid incremental increase in microbiological loading. In the event that discharges to the designated waters cannot be avoided, they will be subject to appropriate treatment to ensure compliance with the [Shellfish Waters] Directive's standards.... All farms in catchment area will be inspected according to the Scottish Executive's... Plan to reduce point source farm discharges into inland and coastal waters. SEPA intend to initiate an Environmental Improvement Plan of agricultural inspections and improvement requirements, designed to reduce diffuse pollution.*

Table 1.2 The three types of fjord in the UK and ROI set of types for WFD coastal and transitional waters (UKTAG 2003). With descriptions of the reference conditions for some physico-chemical and biological elements (UKTAG 2004)

UK & ROI type	TW5	CW11	CW12
Name	Transitional Sea Lochs	Sea Lochs (Shallow)	Sea Lochs (Deep)
physical conditions	Polyhaline, mesotidal, sheltered	Euhaline, mesotidal, sheltered	Euhaline, mesotidal, sheltered
European water type	TW (not distinguished)	CW-NEA6 shallow fjordic type	CW-NEA7 deep fjordic type
Additional physical conditions		Low current; residence time – days	low current; residence time – weeks to months
Type-specific reference conditions			
Physico-chemical conditions	**Nutrient** concentrations will be elevated above Atlantic Shelf Concentrations by a factor dependent on local geological, hydrological and natural input regimes and characterised by a conservative dilution regime.	**Nutrient** concentrations in undisturbed conditions will be a function (± *x* %) of Atlantic Shelf nutrient concentrations. The winter concentration of nitrates and phosphates correspond totally or nearly totally to regional undisturbed conditions.	
	Dissolved oxygen annual range is 80–100%.	Maximum **temperature** range is 0–24.5°C, but typically between 5 and 15°C (mean 10°C), depending on latitude.	
	Maximum **temperature** range 0.8–21°C, typically between 5 and 15°C (mean 10°C) depending on latitude and type-specifics.	**Dissolved oxygen** annual range is 80–120%, with a mean of 100%. Range 2–9 mg l^{-1} (temperature & salinity dependent).	
	Transparency: Light availability is likely to be naturally lower for TWs than for CWs. It is a natural function of physical processes, estuary size, phytoplankton blooms and other organic and inorganic components.	**Transparency:** Light availability will range from clear to highly turbid, depending on type-specific conditions. This may be a natural function of phytoplankton blooms and other organic and inorganic components.	

Phytoplankton	TWs prone to higher levels of production compared with CWs, though light availability, salinity and hydrological effects may naturally temper this. Patterns of seasonal growth and succession are similar to coastal dynamics but demonstrate greater variability, in peak duration and composition. Nuisance/toxic species are at persistently low levels compared with local background levels. Peaks in chlorophyll a, used as a proxy for phytoplankton bloom biomass, are infrequent and inter-bloom periods are low compared with background levels.	Species richness high. Normal patterns of seasonal growth, biomass & succession, i.e., diatom dominated spring bloom and low summer biomass. Diatoms persist throughout growth-period. Increasing numbers of dinoflagellates from late spring. Transition from heterotrophic to autotrophic dinoflagellates from summer to autumn. Autumnal bloom dominated by diatoms or autotrophic dinoflagellates. Nuisance/toxic species at persistently low levels compared with local background levels. Peaks in chlorophyll infrequent & inter-bloom periods low compared with local background.	
Benthic macro-invertebrates	The littoral habitats of the upper reaches of sea lochs are typically comprised of coarse sediment (shingle, gravels and coarse sand). The habitats are subject to variable and reduced salinity conditions, and are typically species-poor and characterised by oligochaete worms.	Shallow sea lochs typically support species characteristic of mixed sediments. Due to the quite variable nature of the sediment type, a widely variable array of communities may be found, including those characterised by bivalves, anemones and file shells. Where physically very stable muds extend from the extreme lower shore to about 15 m depth, the biotopes can be more specific; anemones, brittlestars (e.g., *Amphiura*, *Ophiura*), the opisthobranch gastropod *Philine* and synaptid holothurians being characteristic of shallow muds. Where small stones and shells are abundant on the sediment surface, these can provide a substratum for hydroids, ascidians and other epifauna to attach	Typically, undisturbed circalittoral fine mud is found in deep sea lochs. These habitats are heavily bioturbated by megafaunal burrowers, such as *Nephrops norvegicus*. The infauna may contain populations of polychaetes such as, *Pholoe, Glycera, Nephtys, Pectinaria* and *Terebellides*, bivalves such as, *Nucula, Corbula* and *Thyasira,* and echinoderms such as, *Amphiura* and *Brissopsis*. The gastropod *Turritella* may also be present in large numbers. Epibenthos such as *Asterias rubens*, *Pagurus bernhardus and Liocarcinus depurator* may also be present. Habitats may occur characterised by conspicuous populations of sea pens such as *Funiculina quadrangularis*, *Virgularia mirabilis* and *Pennatula phosphorea.*

(continued)

Table 1.2 (continued)

UK & ROI type	TW5	CW11	CW12
	Lower shore of tide-swept areas, a mixed substratum (mainly cobbles and pebbles on muddy sediments) with dense aggregations of the mussel, *Mytilus edulis* may be found. In high densities the mussels bind the substratum and provide a		
	habitat for many species more commonly found on rocky shores; the mussels are usually encrusted with barnacles and whelks and small crabs are common amongst the mussels. Areas of sediment may contain polychaetes such as the genus *Arenicola* and *Lanice*, the bivalve *Cerastoderma* and other infaunal species.		
	Subtidally, variable salinity cohesive muddy sediment can be dominated by the polychaete *Aphelochaeta marioni* and the oligochaetes *Tubificoides*. The polychaetes *Polydora, Cossura longocirrata* and *Melinna palmata* may also occur in high numbers. The cirratulid polychaete *Caulleriella zetlandica* may be present.		
Examples: hydrography, phytoplankton	Loch Etive (Wood et al., 1973)	Loch Ardbhar (Gowen et al., 1983)	Loch Creran (Tett et al., 1981; Tett & Wallis, 1978)

Definitions: tidal range: micro-tidal < 1 m, meso-tidal 1–5 m, and macro-tidal > 5 m; euhaline: salinity greater than 30.0; polyhaline = 18 or 20 to 30; deep = greater than 30 m. *This except last from WFD CIS Guidance Document No. 5 - which gives shallow <30, intermediate 30–50, deep >50 (COAST, 2003)*

Sources: text copied, with a few abridgements, from UKTAG (2003) for typology and UKTAG (2004) for reference conditions.

Examples are of reference conditions, in the author's opinion and at the time the reported studies were done.

Much of the substance of these plans will no doubt become part of the "programme of measures" and the sampling networks required to monitor their effect. However, there is not much here relevant to the impact of the salmon farm on Creran, and so I must make an informed guess as to what the "environmental objectives" set for the loch will be. Although there are likely to be some changes in SEPA's regulation of benthic AZEs, I am assuming that changes brought about by the implementation of the WFD will increase emphasis on the phytoplankton biological quality element. The "reference condition" for this element in the CW12 type was given in Table 1.1, and Table 1.3 shows how Annex V of the WFD distinguishes the top three quality states of this element for coastal waters.

The main concern here appears to be that nutrient enrichment will lead to signs of eutrophication such as disturbance to the balance of organisms, increased phytoplankton biomass and bloom frequency, and decreased water transparency. Although the definition of "moderate status" does not mention the "undesirable disturbance" that is diagnostic of eutrophication, even moderate disturbances will require remediation, and the resulting "measures" may include more severe constraints on fish-farming if it can be shown to be contributing substantially to nutrient loads.

In the case of loch Creran we have an unexpected finding. When my research student, Céline Laurent, began to sample Creran in 2003, we had expected, on the basis of a simple mathematical model, that there would be a small increase in

Table 1.3 Definitions of *high, good* and *moderate* phytoplankton biological quality in coastal waters, from the WFD Annex V

High status	Good status	Moderate status
The composition and abundance of phytoplanktonic taxa are consistent with undisturbed conditions.	The composition and abundance of phytoplanktonic taxa show slight signs of disturbance.	The composition and abundance of planktonic taxa show signs of moderate disturbance.
The average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions.	There are slight changes in biomass compared to type-specific conditions. <u>Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the quality of the water</u>.	Algal biomass is substantially outside the range associated with type-specific conditions, and is such as to impact upon other biological quality elements.
Planktonic blooms occur at a frequency and intensity which is consistent with the type specific physicochemical conditions.	A slight increase in the frequency and intensity of the type-specific planktonic blooms may occur.	A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.

Quoted from the Water Framework Directive, Annex V, Table. I have underlined the sentence that refers to the definition of eutrophication.

amount of phytoplankton compared with the period 1970–1976, because of the additional nutrient input by the salmon farm now present in the loch. Instead, we found a decrease in average concentrations of chlorophyll (Laurent et al. 2006). The cause of this has yet to be explained. Are farmed mussels eating more phytoplankton? Is the loch chemically polluted by antifouling compounds? Are new chemicals in use on land surrounding the loch? Have its waters become more turbid? The WFD calls for *investigative monitoring* where:

> *surveillance monitoring indicates that the objectives set out in Article 4 for a body of water are not likely to be achieved and operational monitoring has not already been established, in order to ascertain the causes of a water body or water bodies failing to achieve the environmental objectives.*

However, since the emphasis by the WFD, as shown in Table 1.3, is on accelerated growth of algae, it is not clear that there is a failure to achieve the Directive's objectives. The question may turn on whether there has been *an undesirable disturbance to the balance of organisms*, which takes us to the topic of ecosystem health.

1.12 Ecosystem Health

The WFD's *type specific reference conditions* can be interpreted to imply that there is, or was, an ideal, "natural", or "pristine" state, and that any change from this state is a deterioration. But there is a practical problem in identifying reference conditions, given that *high* status corresponds to a state in which

> *no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements… from those normally associated with that type under undisturbed conditions. The values of the biological quality elements… reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion.*

The practical problem is that there are few sites in Europe that are completely free of such disturbance and there is, indeed, some uncertainty about separating developing human influence from natural changes since the ending of the last glaciation. Given that it is unfeasible to seek completely pristine conditions, a realistic aim might be to describe the way things might have been before the industrial revolution in the 19th century. This is exemplified by a modeling study of nutrient discharge from the river Seine, and its effects on the trophic status of the Seine estuary (Cugier et al. 2005).

I think of this interpretation of WFD as seeking a return to a past Eden or golden age, or at least a tolerable approximation thereto. But we live in the here and now, and it may be better to seek a definition of reference conditions that takes account of this. So let us consider the alternative idea that the ideal state for an ecosystem is that of good health, irrespective of whether this state is natural or the result of human management. According to Costanza (1992), a healthy ecosystem, like a healthy human body, is a system that functions well and is able to resist or recover from disturbance. Ecosystems, which have the emergent property of homeostasis, are most healthy when their self-regulatory ability is fully functioning, and ecologists argue that this requires an appropriate balance of organisms performing different functions within the ecosystem. When the balance is disturbed to the extent that the ecosystem is no longer able to self-regulate properly, and is in danger of collapse or becoming something else, then it is unhealthy. This view envisages an internal rather than an external reference for good status. An unhealthy ecosystem is quite obviously not good for the organisms that form part of it, nor for sustainable human use, and it is clearly undesirable for humans to bring about such disturbances to the balance of organisms.

Mageau et al. (1995) propose that ecosystem health has quantifiable components of ***vigour***, ***organization***, ***resistance*** to disturbance, and ***resilience***. Tett et al. (2007) explore ways in which these components might be monitored in marine ecosystems, focusing on the relationship between ***organization*** and ***vigour*** that is shown diagrammatically in Fig. 1.6. The terms can be illustrated by considering the impact of fish farm organic waste on the benthic community underneath a salmon farm at the start of a 2-year cycle.

Initially the benthic community contains a mixture of species and the full range of "guilds" of functional types, such as burrowers and filter feeders. The first result of extra organic input is that existing animals are better fed, and so grow and multiply better. Initially, then, the *vigour* of the community, as measured by the flow of energy through it, increases. As extra organic matter continues to arrive, however, the burrows of animals that pump aerating water through the sediment become blocked, and these animals either die or move away. Oxygen levels within sediment pore water begin to decrease, creating conditions in which fewer species of animals can survive: those which do survive, typically small, specialized worms, have plenty of food and grow numerous. Under very high levels of organic input, all animal life is impossible, and bacteria capable of surviving in oxygen free conditions multiply, consuming all available oxygen and then turning to other compounds that they can use to oxidize organic matter. They may, for example, use the sulphate ions in seawater for this purpose, excreting either sulphur (which makes a white layer on the seabed) or the gas, hydrogen sulphide, which is poisonous to most multicellular animals including fish and humans. There may be a high flow of energy through the seabed, but little of it is put to good purpose within the ecosystem – at least, judged from the standpoint of multicellular animals, so that *vigour* is much decreased. Certainly *organization*, measured by the taxonomic and functional variety of the benthos, has much decreased.

The *resistance* of the benthic community to the pressure of increased organic input is shown by the community's initial increase in *vigour* with load; it is when the burrowers are overwhelmed that this resistance begins to be exceeded and *organization* begins to decline markedly – a state of affairs captured by the cartoon of the Pearson–Rosenberg paradigm in Fig. 1.3(a).

Now, let us assume that, as required by regulation in Scotland, the impacted benthic zone is confined to a small *Allowable Zone of Effect*, and that after 2 years the farm is moved to a new site. Experience has shown that the benthic community

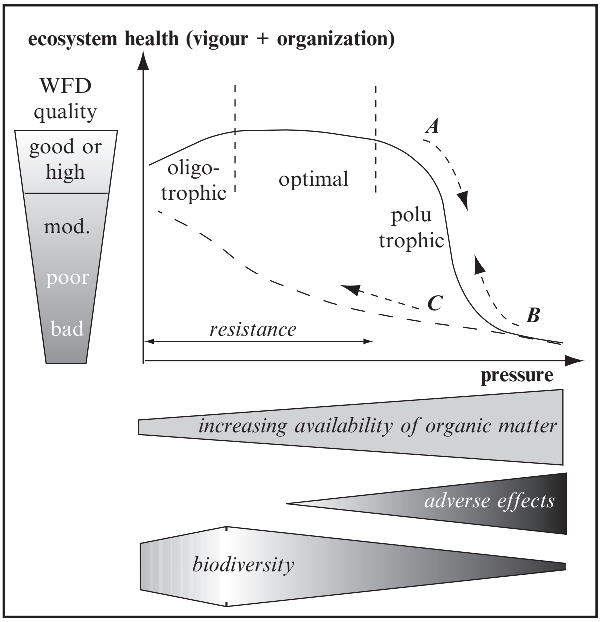

Fig. 1.6 Ecosystem health: changes with pressure. This complex diagram shows one variant of the current ecological paradigm for the behaviour of ecosystems under pressure. It also attempts to relate health to WFD quality. It is based, with modifications, on Tett et al. (2007). Read in the direction shown by arrow A, the main curve shows the response of an oligotrophic (low-production) ecosystem to increasing supply of organic matter, due either to additional inputs from outside, or to nutrient-stimulated primary production. Small increases can add to the vigour and structure of the ecosystem, but larger amounts tend to overwhelm assimilative capacity, so that harmful effects become dominant and the ecosystem state collapses. This is, of course, bad, but a crucial question is whether reducing the pressure leads to ecosystem recovery along curve B, or the persistent change in ecosystem state shown by curve C. Ecosystem resistance denotes the system's self-regulatory property (a function of health) that maintains structure. The diagram uses the term polutrophic (from the classical Greek for "excess nourishment") for the state which is often called eutrophic in contradiction of that word's etymology (from classical Greek for "good feeding"). The WFD would identify the zero-pressure (reference) state as high, and what is here called optimal (because it contains maximum biomass, structure and biodiversity) as, at best, good. If the ecological theory shown here is correct, the line separating good from moderate should be drawn at the point where the ecosystem approaches the edge of the "cliff", after which (from A onwards) its state decays rapidly as pressure increases

recovers rapidly – that is, it shows a high level of *resilience* – because the AZE is surrounded by plenty of healthy benthos to reseed the impacted area with larvae and migration within the sediment. So, on a zone A scale, disturbance to benthic health is of little serious concern so long as confined to one or a few AZEs which comprise only a small fraction of the seabed of a water body such as a sea-loch. But what could happen on the zone B scale?

Suppose that the whole of a sea-loch was given over to mussel farming, so that mussels took by far the largest share of the available primary production and their pseudo-faeces began to buildup a loose layer of mainly inorganic particles on and near the sea bed. Under such conditions the loch would become a "sink" for phytoplankton, consuming more than it produced, and the depletion of food, and benthic changes, might mean that the biomass of plankton and benthos decreased and that some species were eliminated. The increase in turbidity due to the resuspended matter might prevent seaweeds from growing where the sea bed was below the low tide mark. Or, suppose the amount of salmon farmed in Creran was greatly increased, so that nutrient enrichment caused eutrophication, with many algal blooms and increased downwards flow of organic matter, causing many locally impacted areas on the seabed, and depletion of oxygen in the deeper parts of the loch. Again, the effect might be to degrade the *structure* of the benthic community and suppress the natural extent of seaweed primary production. In all these cases the loch's ecosystem would have suffered an undesirable disturbance that can be described in terms of the ecosystem's *resistance* being overwhelmed, so that ecosystem state plunges over the "cliff" in Fig. 1.6, bringing about a state of ill health in which self-regulation is poor.

In such a state there would be little *resilience* within the loch's ecosystem to bring about recovery if mariculture was removed. However, this hypothetically impacted water body is, fortunately, part of a larger world, and it is reasonable to assume that the import of plankton, seaweed spores, and the pelagic larvae of benthic animals, would eventually restore the ecosystem. Nevertheless, we know little about the processes of ecosystem reassembly, and it is possible that the resulting system would be unlike that which was made unwell. Nor do we know how long it would take to restore a zone B ecosystem: very likely, much longer than the 1–2 years required to restore an impacted AZE beneath a fish cage. And finally, as the reader may already have spotted, pressures on zone C scales that lead to a weakening of the health of ecosystems over large parts of coastal seas, will damage the recovery prospects for any zone B scale waterbody for which the zone C scale ought to offer the reseeding potential.

1.13 Phytoplankton Community Index

Ecosystem *vigour* is easy to understand: it refers to the intensity of life, including its production and consumption of organic matter, its turnover of nutrient elements, and its ability to restore a good state after local disturbance. *Organization* is more complicated. If we were dealing with a coral reef, its organization would include the physical structure of the reef, together with the diversity of the organisms living there and there food web interrelationships. A similar account could be written for the benthic community, as shown in Fig. 3(a) where increasing organic loading results in organizational degradation. But what about the plankton? Plankters are passive riders on water motion, and their population abundances can change rapidly.

Can the plankton be said to have *organization?* This section attempts to answer that question for the phytoplankton, by introducing a monitoring tool called the *Phytoplankton Community Index*, or PCI.

An ecosystem is made up of many parts: in the case of loch Creran, of water, mud, dissolved substances and populations of many species of animals, algae and bacteria. These components are continuously changing: water is exchanged with the sea, benthic animals reproduce, seeding the water with planktonic larvae; the balance amongst the populations of species of phytoplankters changes with the season. So, like humans whose every atom is said to be replaced every seven years, ecosystems remain identifiable while subject to flux. There is a way to describe the essence of such changing systems in terms of *state variables*. In the case of Creran, these variables might include the volume and salinity of water in the loch, the concentrations of nutrients and oxygen in the water, and the abundance of species of benthic animals and of phytoplankton. System theory states that a system is in the same state whenever all state variables have the same value. This may seem obvious, or perhaps even tautologous, but it allows us to find ways of describing ecosystems so as to discover whether they are indeed in the same state, which is a precursor to deducing whether the state is "good" (from the perspective of the WFD), or "healthy", or whether it has changed in a way that would be regarded as an "impact" (from the standpoint of the DPSIR terminology). Now let us zoom in to consider phytoplankton alone (but as a component of an ecosystem).

What state variables can we define to capture the essence of phytoplankton in ecosystems? Ecologists have for a long time been interested in species diversity and questions about number of species and the relative abundance of each species (Tett and Barton 1995). However, the list of species of phytoplankters in a typical water sample may be as long as several hundred, and in most cases we know little about what particular species "do" in the pelagic ecosystem. An alternative is to consider that there are a number of functional rôles to be played by pelagic photosynthesizers and that all these rôles must be properly played for proper functioning (and hence health) of the ecosystem. The functions include the cycling of nutrients, and this suggests that there is a distinction between the glassy-walled diatoms, which use and cycle silicon as well as nitrogen and phosphorus, and most other phytoplankters, which do not use silica. Another distinction might be between small phytoplankters, which are suitable food for pelagic protozoa, and larger phytoplankters, which offer a tasty mouthful for copepods and pelagic crustaceans. To cut a potentially long story (Tett et al. 2003b) short, we may view the phytoplankton as being made up of populations of a handful, or double handful, of *life forms*, and a healthy "balance of organisms" being a balance of these life forms able to carry out all the functions that the ecosystem requires of the phytoplankton, and without which it will degrade into an unhealthy state.

This is not the place to list life forms. Indeed, we probably do not know enough about phytoplankton ecology to make a single undisputed list. For the sake of illustration, let us take just two life forms: *pelagic diatoms* (PD, so called to distinguish them from the thick-walled diatoms that normally grow on the seabed but which can be lifted into the phytoplankton by turbulence); and *medium-sized autotrophic* *dinoflagellates* (MAD, a name that emphasizes the need to distinguish photosynthesizing dinoflagellates from their relatives that live by eating other micro-organisms, and which excludes certain large-bodied dinoflagellates characteristic of summer in deep, temperature-layered, waters). Sampling loch Creran at a particular time, counting the phytoplankters in these samples using a microscope, and assigning the relevant counts to these life forms, gives a pair of values: an abundance of the PD and an abundance of the MAD. Next, draw a pair of axes: one for the abundance of the PD and another, at right-angles, for the abundance of the MAD. (For reasons of statistical methods, we actually use the logarithm of abundance.) Onto the resulting Cartesian co-ordinate system, plot the point specified by the abundances of PD and MAD on the date of sampling. It is this point that defines the state of the ecosystem – or at least of its phytoplankton components – on that day.

This, however, is not enough. It is a characteristic of phytoplankton in temperate seas that the absolute and relative abundance of phytoplankter life forms changes with the seasons, and we must take account of this. So we continue to take samples from Creran and to plot additional points until we have several years worth of data displayed on the PD-MAD axes Now we can see that a graph linking the points makes loops on the "PD-MAD" surface, and we can define the "state" of the loch Creran ecosystem as being the area on the PD-MAD diagram that is occupied by all these points. I have made such a diagram in Fig. 1.7, using data obtained from 1979, 1980 and 1981. If we assume that the loch was in a natural and healthy state during these years, and if Creran is typical of its WFD coastal water type, we can argue that this diagram defines a "type-specific reference condition" for the balance of organisms in the phytoplankton. This is the approach taken by the PCI-LF: a "state-space" diagram of this sort is made for a reference condition; an envelope is drawn about this reference condition; and the PCI value is measured by plotting new data onto the same diagram and counting the proportion of new points that fall outside the reference condition envelope (Tett 2006).

We are in course of doing this with more recent data from loch Creran, and some early results are shown in part (d) of the diagram. What is to be expected in a case when a fish farm added sufficient nutrients to disturb the balance of organisms? Because the nutrients would be compounds of nitrogen and phosphorus, but not silicon, they would favor dinoflagellates rather than diatoms; and hence new points should be found more towards the top (the MAD axis) than the right-hand side (the PD axis) of the diagram. This is to some extent what we seem to be finding, although it also seems that diatom abundance has decreased. That decrease might be the proximate cause of the decrease of chlorophyll in loch Creran that was mentioned previously.

I have taken this detour into details of how to assess change in phytoplankton in order to penetrate a little deeper into some of the theory underlying the "ecosystem approach" and to show how such theory may contribute to the practical matter of assessing ecological quality. There are two ways in which a PCI might be used. If it is to be used to quantify the health of the phytoplankton, ecologists need better knowledge of the relationship between organization and vigour in pelagic communities. To be more concrete, which parts of the PD-MAD surface represent a healthy

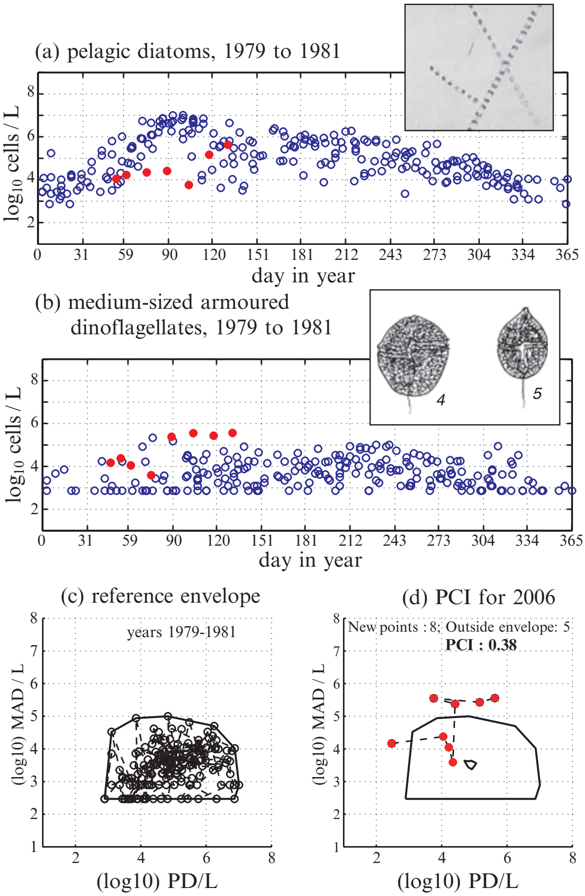

Fig. 1.7 Evaluation of a Phytoplankton Community Index illustrated with data from loch Creran. Part (a) shows the seasonal cycle of pelagic diatoms (PD), illustrated by the common species *Skeletonema costatum*, and was obtained by plotting abundances in all phytoplankton samples taken in the loch during 1979, 1980 and 1981. The vertical scale is logarithmic, in order to show, clearly, the wide range in abundance. The horizontal scale gives days in the year, with day 1 being 1st January. The data from 1979–1981 is plotted as open circles; the small set of filled circles shows observations made during 2006. Part (b) shows a similar graph for the medium-sized autotrophic dinoflagellates (MAD), illustrated by drawings of a typical species of the genera *Gonyaulax* and *Scrippsiella*. In part (c) the 1979–1981 data from (a) and (b) have been plotted onto a surface whose axes are the abundances of the PD and the MAD, and an envelope has been drawn around the points to define reference conditions. In part (d) the envelope has been redrawn, and points from 2006 plotted onto the surface. The PCI is the proportion of new points that remain inside the reference envelope

state and which do not? In the absence of adequate knowledge, the WFD Annex V assessment strategy serves to provide an empirical appraisal of change away from a reference state which is by definition healthy. If a sufficiently large proportion of points fall outside the reference envelope, then the PCI can be used to indicate a change in quality from *high* or *good* to *moderate* or worse. This is the second use, but even it needs agreement about critical values of the PCI – at the *good/moderate* boundary, above all.

1.14 Assimilative Capacity

Given regulation according to the WFD and the need to maintain ecosystem health and sustainable human use, how many finfish or shellfish can be farmed within a water body? The size of a sustainable aquaculture is said to be the ***carrying capacity*** of the water body for the stock concerned; I approach it here from the alternative perspective of the ***assimilative capacity*** of the water body for the wastes of – or, more generally, the pressures generated by – fish-farming and other human activities. What, for example, is a water body's ability to absorb anthropogenic DAIN without significant adverse effect on the health of the ecosystem?

Figure 1.8 shows some of the principles involved in the estimation of assimilative capacity. The horizontal axis represents increasing pressure from anthropogenic activity. This pressure could be quantified as the number of fish farms in a water body or the number of humans who would produce waste equal to the total input to the water body from all sources, but it is better to relate the waste input to the receiving system. Thus, suitable indicators of pressure would be the annual rate of organic matter arriving on each square metre of seabed in the AZE below a farm, or the daily total of nutrients input to a zone B water body, divided by the volume of the water that is replaced each day from the adjacent sea.

The vertical axis is something that measures impact on the ecosystem – that is, the change in state from a reference condition as defined for the WFD or a decrease in the health components *organization* and *vigour*. Examples of such benthic indicators include the AZTI Marine Biotic Index (AMBI) (Borja et al. 2003) and the Infaunal Trophic Index (ITI) (Word 1990). These assess the balance of the several kinds of large benthic animal needed to maintain a healthy ecosystem in the mud. Examples for the water column include the excess of chlorophyll concentration over that in a reference condition, and the PCI described above.

There is a scale issue: the pressure variable on the x-axis and the impact variable on the y-axis must relate to the same scale: A, B or C as defined previously. Given that, the next part of the task is to find a relationship between the two axes, as shown by the diagonal line in the diagram. A simple relationship might be that of linear regression, so that $y = a + b.x$, where a and b are constants. As suggested by the curve in Fig. 1.6, the true relationship in Fig. 1.8 is unlikely to be simple, but this is not a problem so long as it can be expressed by a mathematical equation, or by a table in which values of impact, y, can be looked up for values of pressure, x.

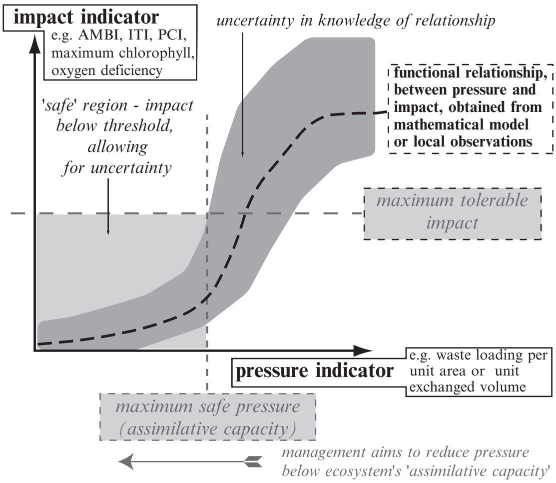

Fig. 1.8 The estimation of assimilative capacity. Note that the impact indicators: "maximum chlorophyll" and "oxygen deficiency" directly indicate impact; the AMBI, ITI and PCI are constructed so that their values are high for high ecological status, and thus, strictly, it is their inverse that is an indicator of impact

The relationship can be gotten in two ways: by observing y for many values of x; or by developing a mathematical model that predicts y from x. In either case, there will be some uncertainty in any prediction of y, and this is shown by the grey area that surrounds the relationship line.

Regulators need to set thresholds for the impact indicators, exemplified for the water column and the zone B scale by the CSTT's threshold of 10 mg chlorophyll m^{-3} in summer. The greatest tolerable pressure is that which takes the y-axis variable up to, but not past, the threshold. This pressure is the *assimilative capacity* of that particular water body for the waste responsible for the pressure, and it is a regulator's or planner's task to consent discharges only up to this capacity, taking account of any natural contributions towards it. It may also be a farm manager's task (as a condition of a consent to discharge) to ensure that the zone A pressures, and the contribution to zone B scale pressures, are controlled so that the impacts do not exceed those allowed.

Note that an impact threshold is not an EQS, which is explicitly defined by the WFD in terms of concentrations of pollutants. An impact threshold can, also, be stated implicitly, by way of an *Ecological Quality Objective or* **EcoQO**, exemplified for the zone A benthos by the Scottish regulator's requirement that there must be at least 2 taxa of Polychaeta worms alive within a fish-farm AZE. Painting et al. (2005) consider the utility of several larger-scale EcoQOs intended to prevent eutrophication.

Account needs to be taken of the uncertainty in the *y–x* relationship. If the greatest allowable pressure were read at the point on the *x*-axis of Fig. 1.8 where the upper limit of the uncertainty crosses the impact threshold on the *y*-axis, the risk of an undesirable impact would be minimized. This is giving the benefit of the doubt to the ecosystem. Alternatively, the benefit could be given to the producers of waste by using the lower limit to the uncertainty, which will maximize the tolerable pressure. Or, the most probable *y–x* relationship (the thick dashed line) could be used, ignoring the risk of some excessive impacts and denying fish-farmers (and other waste producers) the benefit of some possible assimilative capacity. Carstensen (2007) has discussed this topic in the context of identifying to which WFD quality class a water body belongs.

Of course, wastes and pollutants enter ecosystems from many sources. In the case of nutrients, the anthropogenic sources include diffuse agricultural inputs and urban waste water discharges as well as aquaculture. Thus, the ability to estimate the maximum safe loading from all these sources gives regulators the new task of sharing a water body's nutrient assimilative capacity amongst its human users – some of whom have, historically, taken it as an inexhaustible gift of nature rather than something that they might have to share or pay for.

1.15 Sustainability and the Ecosystem Approach to Aquaculture

This chapter has mentioned the use of the terms *Environmental Quality Standard* and *Ecological Quality Objective*. Initially, EQSs were made to prevent pollution by harmful substances, and the term is used in the Water Framework Directive in exactly this way. The concept of specific EcoQOs came later, and the term is not always used explicitly: for example, the Water Framework Directive refers to general "environmental objectives" in its main text and to "quality status" in Annex V. Nevertheless, it is useful to see the statement of precise EcoQOs as a key device for maintaining the status of components of ecosystem quality or health or to prevent undesirable impacts. EQS's and EcoQOs can be enforced only if each is associated with an indicator than can be monitored. In the case of the highly toxic substances which are dangerous at any level, the scheme set out in Fig. 1.8 can be bypassed: the ecosystem has no assimilative capacity for these substances, and the Water Framework Directive aims to stop their release into the aquatic environment. In all other cases Fig. 1.8 summarizes the task facing aquacultural managers, regulators and scientists.

The figure is an outcome from ECASA, a European Commission Framework 6 project, concerned with the *Ecosystem Approach to Sustainable Aquaculture*. ECASA's main product (Box 1.1) is a "virtual toolbox" giving details of the models and indicators that can be used to apply the approach of the previous section. These tools do not in themselves guarantee sustainable aquaculture, because this requires economic efficiency and attention to the needs of local societies in addition to a

Box 1.1 Models for assimilative capacity in the ECASA project

ECASA stands for *Ecosystem Approach for Sustainable Aquaculture*. The project, which ran from December 2004 through November 2007, was part of the European Community's 6th framework program and funded by a contract from the Fisheries Directorate-General of the Commission of the European Communities. Its aims included:

Assessing the applicability (efficiency, cost effectiveness, robustness, practicality, feasibility, accuracy, precision, etc) of selected ***indicators*** *and developing operational* ***tools****, e.g.,* ***models****, establishing the functional relationship between environment and aquaculture activities.*

The models studied during ECASA include the following categories:

1. Models for the biology of a type of farmed organism, including mussels and other shellfish, and salmon and sea bream. Some of these models can be used for best management of the animals.
2. Zone A models for local impact of fish-farms, especially models able to predict the pattern made on the sea-bed by sinking organic waste and its effect on the sediment and benthos.
3. Zone B models for the water-body scale impact of finfish-farming, including effects on chlorophyll, transparency and deep-water oxygen that are associated with eutophication, together with basin-scale models for shellfish production.

Further details may be obtained from the ECASA web site at http://www.ecasa.org.uk, and details about the models can be found in the ECASA "toolbox", at http://www.ecasa.org.uk/toolbox.

concern for ecosystem health, but they can help to manage sites for sustainability by ensuring that conditions remain within the "safe" area in Fig. 1.8.

An indicator of sustainability is categorically different from indicators of pressures and impacts. The latter are like thermometers: their readings help describe the weather at a particular time, or show whether a human is well or sick from fever. A sustainability indicator must take account of time and the overall *state* of the ecosystem. More precisely, if the symbol Y_i refers to values of a particular impact indicator, $\{Y_i\}$ means the set of all relevant impact indicators, and $f(\{Y_i\})$ specifies a function of the values of each member of this set, such as the function that converts monitoring results into a WFD water quality status value. Then a sustainability indicator is $\mathrm{d}f(\{Y_i\})/dt$, and a generalized EcoQO for sustainability requires that: $\mathrm{d}\, f(\{Y_i\})/dt \leq 0$ for a given site, water body or regional sea.

To understand this, imagine a set of diagrams, similar to Fig. 1.8, for each of the CSTT scales. To make things more concrete, let us consider the zone B scale water body that is loch Creran. The known environmental pressures on this scale are from nutrients, oxygen-demanding organic matter, and antifouling and anti-lice chemicals, mostly from several sources. For each, one or more a pressure-impact diagrams can be drawn. Models, such as those examined during ECASA, can be used to guide management of the pressures. Monitoring can be carried out to establish whether impacts within Creran remain within the “safe” region in each pressure-impact diagram. If they do, then human use of loch Creran is sustainable, and we can expect to go on using it in the same way in future as we have in the past.

Of course, this judgement is not eternal. Changes such as those due to global warming, for example, might increase or decrease the loch’s assimilative capacity. In a warmer world, in which water can dissolve less oxygen, the oxygen demand of decaying waste might be more critical than the nutrients released by that decay. So the situation must be kept under review.

It is also possible that the ecosystem approach might be used not only to maintain the health of loch Creran but also to increase the efficiency with which humans can take goods (such as mussels) and services (such as nutrient assimilation) from it. Suppose, for example, that the critical pressure-impact diagram is the one that relates eutrophication impact to nutrient loading. This could constrain finfish-farming by means of setting a limit to the amount of nutrients that the farm could put into the water. In this case, farming shellfish or seaweeds in the loch might be a way of removing some of these nutrients and hence effectively increasing the nutrient assimilative capacity of Creran.

My other theme in this chapter has been the potential impact of the Water Framework Directive on fish farming. The Directive aims to *establish a framework* that *protects and enhances the status of aquatic ecosystems*, and that is – excepting my reservation about the difference between ecosystem health and ecosystem quality status – just what is needed to maintain health and ensure ecological sustainability. How much difference will the WFD make to aquaculture? As in many places in this chapter, I focus on Scotland, the only part of Europe where I am familiar with law and regulatory practice as well as the ecological impact of fish-farming. For Scotland there are two simple and apparently opposed answers: not much; and, a lot.

First, the “not much change” answer. Because the WFD builds on and synthesizes previous directives, and is implemented in Scotland using regulatory methods that are already well developed, the changes in regulation are likely to be gradual and, perhaps, will impact most on the most old-fashioned aspect – that of pollution by synthetic compounds. In my view, fish farmers should expect in the long run to do without these, which has implications for the management of fouling, sea-lice and diseases. It probably means farming fish at lower densities in more highly dispersive environments. However, some fish farmers are already exploring this, and those that do so are able to get a premium on their fish, both out of consumers’ concern for animal welfare and environmental health, and because (in my view) fish thus farmed, taste better.

Second, the “big difference” answer. This is based on the argument that the WFD implements the ecosystem approach. If farmers and regulators become real converts to the ecosystem approach, their world view will change. Farmers will go from reluctantly conforming to AZE regulations to willingly embracing their part in maintaining ecosystem sustainability on the zone B scale, perhaps with collaboration between finfish aquaculture (which adds nutrients) and shellfish aquaculture (which benefits from increased amounts of phytoplankton). Whether such a change can take place in the existing economic environment is a matter for other chapters in this volume.

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Chapter 2
Monitoring of Environmental Impacts of Marine Aquaculture

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Abstract Marine aquaculture is regulated and monitored through international and national legislation that varies significantly between countries and regions around the world. Research is still needed to improve the monitoring programmes, in particular those related to the ecosystem approach at larger scales. Most monitoring programmes include examination of the benthic environment and some also of water quality, although impacts are difficult to detect due to rapid dilution. In the Mediterranean benthic monitoring may include use of the seagrass *Posidonia oceanica*, as this species is widespread and highly sensitive to aquaculture waste products. This chapter provides details of two monitoring programmes: (1) salmon farming in Norway and (2) sea bream/sea bass and tuna farming in Malta.

Keywords Ecosystem approach, Europe, case studies, Norway, Malta

2.1 Regulation and Monitoring of Marine Aquaculture

Marine aquaculture is a diverse production industry involving a variety of different species, production methods and husbandry. In this chapter we will primarily focus on finfish culturing and to some extent shellfish production. Environmental impacts of finfish culturing are widely documented (Hargrave 2005) and include a broad

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range of impacts from aesthetic to direct pollution problems (Fig. 2.1, Pillay 2004). Fish production can generate considerable amounts of dissolved effluents, which potentially affect water quality in the vicinity of the farms, and due to rapid dilution, also at larger scales (km-scale). Due to the rapid dilution it has been difficult to document the effects of dissolved nutrients in farm vicinities, in particular in areas with relatively high nutrient concentrations such as in the Baltic Sea (Christensen et al. 2000). Other studies of fishery landings in the Mediterranean suggest that nutrients are rapidly transferred up the trophic chain enhancing secondary production (Machias et al. 2005), indicating that monitoring of nutrient losses should be done at different scales. Due to rapid settling of feed and faecal pellets in the vicinity of the farms, benthic impacts are much more widely documented (Holmer et al. 2005; Kalantzi and Karakassis, 2006). This input of organic rich material enhances the microbial processes in the sediments, often leading to anoxic conditions (Holmer and Kristensen 1992). This may have major effects on the benthic fauna and flora leading to lower fauna and flora densities under the cages or even defaunated sediments (Delgado et al. 1997; Karakassis et al. 2002). Other environmental impacts include release of chemicals, medicines and pesticides, which are used for treatment of the fish and the farm installations. Interactions with wild populations, spreading of disease and release of parasites from farms are also of environmental concern.

The environmental impacts of marine aquaculture within the European Union are regulated and managed, at a European level, through a variety of European Commission (EC) Directives and International Conventions. There are currently eight EC directives (Table 2.1) directly involved and an additional 50 + Directives, Decisions and Regulations, which have an indirect effect (Read et al. 2001). In addition, three International Conventions on marine pollution cover EU coastal waters (Table 2.2) and there are a further 30 + international agreements that have an indirect effect on the monitoring and regulation of marine aquaculture (Read et al. 2001). Within the European Union, the regulation of the aquaculture sector comes under the remit of the Common Fisheries Policy (CFP). The CFP states that Member States shall adopt provisions to comply with the objectives of regular monitoring of activities and technical controls. At EU level, environmental protection measures have been established at three levels: (1) general policy; (2) specific measures; and (3) regulations that control specific local conditions (Eleftheriou and Eleftheriou 2001).

The regulations controlling aquaculture vary between countries, but most countries use some form of Environmental Quality Objectives (EQOs) and Environmental Quality Standards (EQSs) (Table 2.3). Only a few countries apply a carrying capacity at the moment, but this has been suggested for future regulation within the Integrated Coastal Zone Management (ICZM) approach (see below). Most countries have specific demands for the location of the farms to avoid situating these near habitats of special interest (recreation, wild life, fishing zones) and near industries and sewage outfalls. Requirements on stocking density, feed type and sediment and water quality standards are also included in most regulations. A few countries regulate the production based on discharges, e.g., N and P release per kg

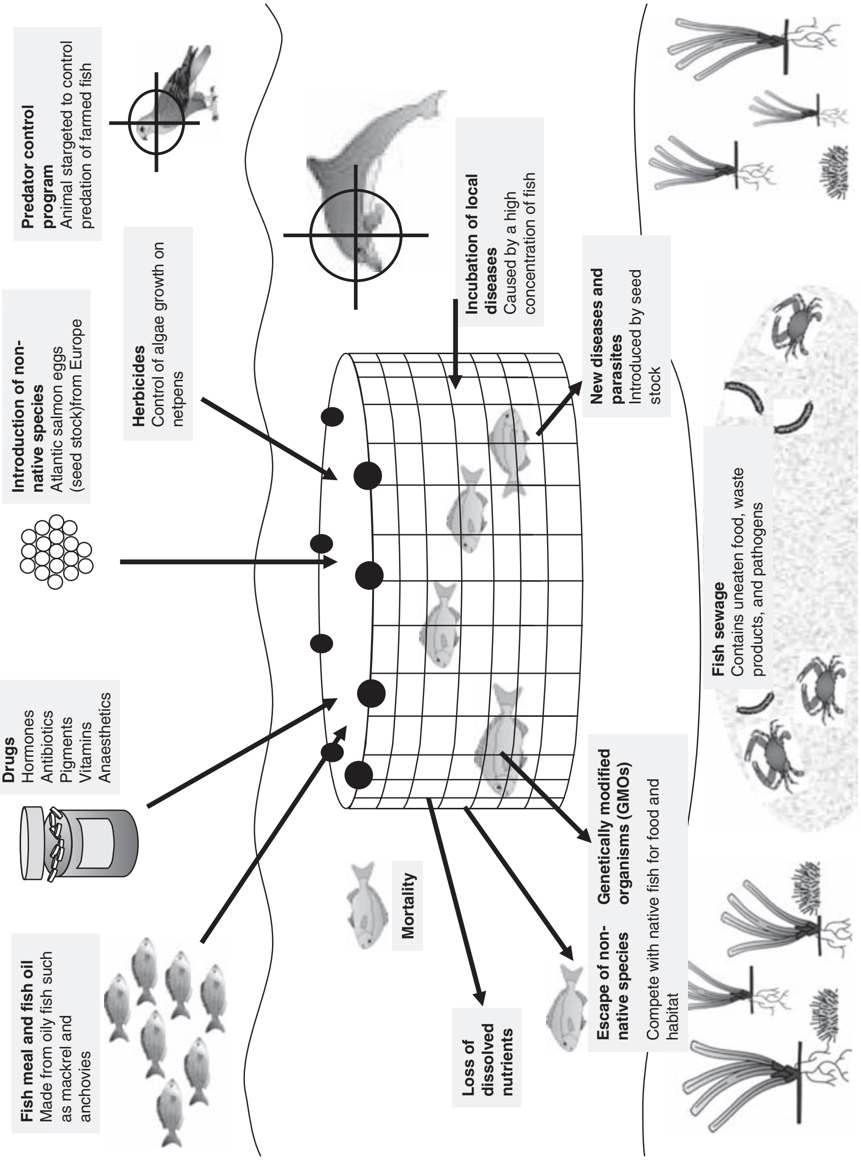

Fig. 2.1 Conceptual figure of aquaculture in the ecosystem (symbols from IAN Symbol Libraries)

Table 2.1 European Commission directives related to management of marine aquaculture

Directive	Target
Dangerous substances directive	Reduce pollution by list II substances
Quality of shellfish growing waters directive	Contribute to high quality of shellfish products through protection or improvement of shellfish flesh and shellfish waters
Shellfish directive	Classification of production areas on the basis of bacteriological criteria
Environmental impact assessment directive	Part of the application and licensing procedures for development
Strategic environmental assessment directive	Identification and assessment of environmental consequences of aquaculture during preparation and before adoption
Species and habitat directive	Protection and conservation of natural habitats.
Wild birds directive	Protection and conservation of natural habitats
Water framework directive	Development of catchment management plans for implementation of integrated management. Operates with assimilation capacity.

Table 2.2 International conventions which apply to marine aquaculture operations

Conventions	Area of cover
OSPAR	Northeast Atlantic
Helsinki	Baltic Sea
Barcelona	Mediterranean Sea

fish produced or total release per farm per year, to encourage the producer to optimize the feed efficiency, as for example is the case in Denmark. Regulations on food standards (fish and shellfish products) may also apply such as maximum residue limits for pesticides and other contaminants in fish or shellfish flesh. Most countries require licenses for medicine and pesticide use, and in some countries, use of pesticides is not allowed at all (e.g., Denmark).

Monitoring strategies of aquaculture vary between countries, dependent on the regulatory control (Table 2.4). Self-monitoring applies in several countries, where the fish farmer collects and submits the results to the authorities or is supported by on-site Authority control at varying frequencies. Other countries have the monitoring done solely by the authorities. In most cases, water quality and benthic conditions are checked at regular intervals (2–12 times per year). Examples of European monitoring programmes are presented in detail for the North-Atlantic (Norway) and for the Mediterranean (Malta) below.

Most countries monitor the food quality of the products, in particular for shell fish, where accumulations of biotoxins and microbes are sampled up to twice a week during intensive production and periods of risks of contamination, e.g., during

Table 2.3 Regulations and legislation controlling marine aquaculture in Europe and Asia (updated from Fernandes et al. 2000)

		Regulatory control				
Country	Type of marine aquaculture	Carrying capacity	Environmental standards	Food standards	Medicines licenses	Pesticides licenses
Asia						
Cambodia	Finfish and shellfish	–	Sub-decree on water pollution control, Fisheries law	Proclamation on fisheries product hygiene	–	–
China	Finfish and shellfish	–	Water quality standards for fisheries	Food hygiene law, Standardization law	Administration of veterinary medicines (1988)	Environmental protection law, Law on the prevention and control of water pollution, Marine environment protection law, Regulations on the environmental management of first import of chemicals, Import and export of toxic chemicals (1994), Control over the safety of dangerous chemicals (2002)
Indonesia	Finfish and shellfish	–	Government regulation	Food Act, Fisheries Law, Decree of the Minister of Marine Affairs and Fisheries	Fisheries Law, Decree of the Minister of Agriculture	–
Korea	Finfish and shellfish	–	Water quality conservation act, Public water management Act, Fisheries Act	Fisheries act, products quality control Act	–	–

(continued)

Table 2.3 (continued)

		Regulatory control				
Country	Type of marine aquaculture	Carrying capacity	Environmental standards	Food standards	Medicines licenses	Pesticides licenses
Malaysia	Finfish and shellfish	–	Waters Act, Environmental Quality Act, Environmental Quality Regulations	Food Act, Food Regulations	Food Regulations establish limits of drug residues in fish products	–
Myanmar	Finfish and shellfish	–	Myanmar Investment Commission	Marine Fisheries Law covering hygiene, food additives, microbiological criteria, packaging, labeling, water standards, application of medical drugs, and HACCP system	Marine Fisheries Law and Drug Law	Pesticides Law
Philippines	Finfish and shellfish	–	Philippine clean water Act, Marine Pollution Decree	Fisheries Code, Consumer Act, FAO	Fish health section	Fertilizer and Pesticide Authority
Thailand	Finfish and shellfish	–	Enhancement and Conservation of National Environmental Quality Act, Fisheries Act	Food Act	Drug Act	Hazardous Substances Act
Australia	Finfish and shellfish	–	Environmental Protection Policy, Aquaculture Regulations	Australia-New Zealand Food Standards Code, Commonwealth Food Standards Australia-New Zealand Act, South Australia Food	Agricultural and Veterinary Chemicals Code Act, South Australian Agricultural	Agricultural and Veterinary Chemicals Act

				Act, South Australia Fisheries Act, Fisheries Regulations, South Australia Primary Produce Act	and Veterinary Products Act, Agricultural and Veterinary Products Regulations, South Australia Veterinary Practices Act, Fisheries (Exotic Fish, Fish Farming and Fish Diseases) Regulations	
Europe						
Denmark	Finfish (land-based and sea cage) (mainly rainbow trout); some shellfish (extensive)	Effluent released to marine waters (maximum of 560 tN and 54 tP)	Nutrient output (560 tN and 54 tP annually); feed type, feed convention ratio	None for finfish at producer stage. Bivalves checked for algal toxins	Prescribed by veterinarian; number permitted is limited	Not permitted
Finland	Finfish (mainly rainbow trout)	No figure	Discharge (8 g P/kg fish produced, 70 g N/kg fish produced)	In accordance with EU Directives	Granted by National Agency for Medicines	Not used due to favourable parasitic situation
France	Shellfish (mainly oyster and mussels); some finfish (mainly sea bream)	20 t $year^{-1}$ for finfish; No figures for shellfish	Shellfish growing water quality, EIS can dictate standards for finfish; threshold levels for chemical residues	Bacteria and algal toxins in shellfish flesh; none for fish	License given by veterinarian	License from Ministry of Health

(continued)

Table 2.3 (continued)

	Regulatory control					
Country	Type of marine aquaculture	Carrying capacity	Environmental standards	Food standards	Medicines licenses	Pesticides licenses
Germany	Shellfish (mainly mussels); some finfish (mainly rainbow trout)	No figure	Effluent water quality; closed season for mussel harvest; shellfish growing water quality	Shellfish flesh quality	Use must be recorded in "operation record book"	Use must be recorded in "operation record book"
Greece	Finfish (mainly sea bass and bream); shellfish (mainly mussels)	No figure	Cage location; stocking density; feed type, water and sediment quality; sanitary measures	Same as legislation for farmed domestic animals	Approved by National Drug Organizations	Approved by National Drug Organizations
Iceland	Finfish (mainly salmon) some char and rainbow trout	No figure	Cage location; water quality; nature conservation	Maximum residue limit for pesticides and other contaminants in flesh; standards for microbes and algal toxins	Permit from Veterinary Officer of fish diseases	Permit from Veterinary Officer of fish diseases
Ireland	Finfish (mainly salmon); shellfish (mainly mussels)	15 kg m^{-3} stocking density for finfish	Cage location, escape prevention, fallowing period; abundance of sea-lice; water and seabed quality	Algal toxins in shellfish flesh; disease and chemical residues in finfish and shellfish flesh	Issued by veterinarian	Issued by veterinarian Organotin antifoulants are not permitted
Italy	Finfish (mainly trout); some shellfish (mussels, clams)	No figure	Water quality	Heavy metals and organic compounds in bivalves	Issued by veterinarian	–

Malta	Finfish (mainly sea bream and blue fin tuna)	15 kg m^{-3} for sea bream and 3–4 kg m^{-3} for blue fin tuna	Water quality; sediment quality; benthic diversity; state of benthic habitats in the vicinity of farm	In accordance with EU Directives	Issued by Veterinarian at the Department of Fisheries and Aquaculture	None issued – pesticides not permitted
The Netherlands	Shellfish (cockles, mussels, oysters)	100,000 metric tons (fresh weight mussels 10,000 metric cockles	Safeguard of food reserves for wild birds; biotoxins and microbes in shellfish growing waters	Biotoxins and microbes in shellfish flesh	Not used in extensive mariculture	Not used in extensive mariculture
Norway	Finfish (mainly salmonids); shellfish (mainly mussels)	Site dependent	Receiving water body: nutrients, oxygen, sediment C and N, benthos, organic pollutants, metals. Site: sediment condition.	EU standards for fish and shellfish	Issued by veterinarian	Issued by veterinarian
Portugal	Shellfish (clams, oysters, cockles); some finfish (sea bass, sea bream, turbot)	No figure		In accordance with EU Regulations	Permission required from Direccao Geral da Veterinaria	Permission required from Direccao Geral da Veterinaria

(continued)

Table 2.3 (continued)

	Regulatory control					
Country	Type of marine aquaculture	Carrying capacity	Environmental standards	Food standards	Medicines licenses	Pesticides licenses
Scotland	Finfish (mainly salmon with some rainbow trout and turbot); shellfish (oysters, scallops, mussels)	Maximum weight of fish to achieve sustainability at a farm	Hydrological character, biological, sediment and water quality	Biotoxins in shellfish flesh, phytoplankton in shellfish growing water; maximum residue limit for chemical in finfish flesh	Permission required from Veterinary Medicines Directorate	Permission required from Veterinary Medicines Directorate
Spain	Shellfish (mainly mussels, cockles, clams); some (salmon, turbot)	No figure	Water quality for shellfish	EC Directives controlling production; maximum residue levels in flesh	Controlled by veterinarian	–
Sweden	Finfish some shellfish (mussels)	No figures for finfish, 10,000 t for shellfish	Water classification according to organic matter and metals in mussel tissue; farm sites, extraction and discharge of water, nutrients	Species (and strains) allowed	Use permitted	Use permitted

Table 2.4 Monitoring programmes used in marine aquaculture (updated from Fernandes et al. 2000)

Country	Type of marine aquaculture	Monitoring: Environmental – Self monitoring	Monitoring: Environmental – Authority monitoring	Monitoring: Food – Self monitoring	Monitoring: Food – Authority monitoring
Europe					
Denmark	Finfish (land-based and sea cages) (mainly rainbow trout); some shell fish (extensive)	Water quality monitored 12 times per year (finfish); sediment monitored twice per year (finfish)	Compliance with finfish standards (periodic)	Bivalves checked for algal toxins post harvest	Compliance with standards (periodic)
Finland	Finfish (mainly rainbow trout)	Water quality; farmers keep daily records of operation for inclusion in report	Water quality (chemistry, plankton and micro-algae); sediment quality (sediment, benthic macroinvertebrates)	–	Compliance with EU Directives
France	Shellfish (mainly oyster and mussels); some finfish (mainly sea bream)	Quality of effluent (finfish)	Benthic survey and nutrient analysis of the water column twice per year; chemical residues and microbiology of water twice per month; shellfish water quality (algal toxins and phytoplankton) monitored twice per month	Bacteria, phytoplankton toxins in finfish flesh (before marketing)	Bacteria, phytoplankton toxins in finfish flesh (without warning); bacteria, phytoplankton toxins in shellfish flesh (twice monthly)

(continued)

Table 2.4 (continued)

Country	Type of marine aquaculture	Monitoring: Environmental – Self monitoring	Monitoring: Environmental – Authority monitoring	Monitoring: Food – Self monitoring	Monitoring: Food – Authority monitoring
Germany	Shellfish (mainly mussels); some finfish (mainly rainbow trout)	–	Control of fishing vessels during mussel harvest and seed fishing. Water quality monitored five times in 3 years	–	–
Greece	Finfish (mainly sea bass and bream); shellfish (mainly mussels)	Regular water quality, sediment and benthic community monitoring	–	–	–
Iceland	Finfish (mainly salmon) some char and rainbow trout	All accessible knowledge collected to control environmental effects	Chemical residues and micro-organisms in fishing grounds	Internal quality control to ensure compliance	Chemical residues and micro-organisms in fish flesh; quality control
Ireland	Finfish (mainly salmon); shellfish (mainly mussels)	–	Sea lice monitored 14 times per year; water at salmon farms monitored monthly; benthic monitoring depends on tonnage (finfish)	–	Shellfish flesh monitored weekly for algal toxins, annual sampling for disease at salmon and oyster sites; analysis of farmed salmon for chemical residues
Italy	Finfish (mainly trout); some shellfish (mussels, clams)	–	macro-descriptor in water monitored seasonally (twice per week from June–September;	–	Bioaccumulation of heavy metals and organic compounds in bivalves twice per year

			ground parameters once per year; maximum trophic index of 5.5		
Malta	Finfish (mainly sea bream and blue fin tuna)	Water quality at some farms	Water quality; sediment quality; benthic diversity; state of benthic habitats in the vicinity of farm	–	Compliance with EU Directives
The Netherlands	Shellfish (cockles, mussels, oysters)	–	Aerial photography and ground counts of inter-tidal mussel stocks. Biotoxins and microbes in shellfish growing waters sampled 2 times per week	–	Biotoxins and microbes in shellfish flesh 2 times per week
Norway	Finfish (mainly salmonids); shellfish (mainly mussels)	–	Shellfish water quality once per month. Finfish farms monitored according to Norwegian Standard NS9410	–	10% of finfish licences sampled every year to check for chemicals in muscle and feed; all medicated fish are checked; shellfish waters monitored throughout the year for biotoxins, microbes and chemicals
Portugal	Shellfish (clams, oysters, cockles); some finfish (sea bass, sea bream, turbot)	–	Mollusc water quality once per month or once every 3 month depending on parameter	–	Compliance with food quality standards

(continued)

Table 2.4 (continued)

		Monitoring			
		Environmental		Food	
Country	Type of marine aquaculture	Self monitoring	Authority monitoring	Self monitoring	Authority monitoring
Scotland	Finfish (mainly salmon with some rainbow trout and turbot); shellfish (oysters, scallops, mussels)	Near-field sampling following an agreed programme (finfish)	Audit self-monitoring of finfish farms; consent compliance of medicines; environmental impact from finfish farms; shellfish growing waters quality (periodic)	–	Assays of biotoxins; microbial quality of water; trace compounds in shellfish flesh and fish tissue
Spain	Shellfish (mainly mussels, cockles, clams); some (salmon, turbot)	–	Periodic water quality, weekly red tide monitoring	–	Compliance with food standards monitored yearly (MRLs)
Sweden	Finfish some shellfish (mussels)	–	Water classification according to organic matter and metals in mussel tissue	Check for algal toxins in shellfish prior to harvest	Biotoxins in mussel meat normally sampled once per week; faecal coliforms, chemical residues in finfish flesh.

North-America					
Canada	Finfish (primarily Atlantic salmon and some Atlantic char, halibut and Atlantic cod). Shellfish (primarily blue mussels with minor production of scallops	Finfish: water temperature and dissolved oxygen measured daily throughout summer/fall periods of maximum growth. Shellfish: befouling (weekly to monthly), algal bloom status through phytoplankton monitoring	Finfish: Federal Government (DFO) annual monitoring of sediment geochemical variables to classify benthic enrichment status, no routine water mass column monitoring for any variable. Shellfish: No annual monitoring. Provincial governments track yield from all leases to record harvested biomass	Finfish and shellfish: Industry quality guidelines for fresh and frozen seafood products are followed	Canadian Depts. Of Food Inspection and Health Canada monitor products (random sampling on irregular basis) for substances such as bacteria and antibiotic residues. Compliance required for meeting exported seafood guidelines. Locally consumed fish (within Canada) is not routinely monitored

phytoplankton blooms. Finfish flesh is monitored for bacteria, chemical residues and phytoplankton toxins once every year, or in some countries, every year before marketing.

2.1.1 Research Support for Monitoring of Environmental Impacts

There have been several papers, reports and other documents dealing with the principles of monitoring and particularly so in the case of the monitoring of fish-farming impacts. The report by GESAMP (1996) addressed this issue by describing possible scenarios of fish farm locations and suitable monitoring programmes. Although the paper included a list of variables used for monitoring the ecological effects of coastal aquaculture, the authors realised that the information provided by some of these variables is of limited use in some situations. A comprehensive series of studies on monitoring and regulation was also undertaken in the framework of the MARAQUA project (The Monitoring and Regulation of Marine Aquaculture in Europe). These studies resulted in a series of papers on the scientific principles underlying the environmental monitoring of aquaculture (Fernandes et al. 2001), on the control of chemicals (Costello et al. 2001), on the genetic interactions between farmed and wild fish species (Youngson et al. 2001) and on the use of hydrodynamic and benthic models for the management of aquaculture impacts (Henderson et al. 2001). However, research on aquaculture-environment interactions has progressed remarkably during the last 5 years, particularly in the framework of EU-funded projects, which have provided useful information for the understanding of various ecosystem processes affected by the presence and operation of fish farms.

The effects of aquaculture on marine benthos, particularly on macrofauna, have been known for long (Gowen and Bradbury 1987), and in general, they seem to follow the pattern described by Pearson and Rosenberg (1978) regarding the succession of macrofaunal organisms along the benthic enrichment gradient. However, more than 40 articles in the scientific literature (review in Kalantzi and Karakassis 2006) have studied these affects using in total 120 biological and geochemical variables, most of which were highly intercorrelated. A meta-analysis of the most commonly used of those variables by Kalantzi and Karakassis (2006) showed that their values are determined by a combination of distance from the farm with bottom depth and/or latitude. Although the benthic effects are relatively easy to detect, there are some concerns regarding the cost of the associated faunal analysis, which becomes more and more difficult due to the rarity of experts in the taxonomy of benthic organisms (GESAMP 1996). A series of papers have addressed this issue by studying the potential use of surrogates and their effect on data quality. Karakassis et al. (2002) have used sediment profiling imagery (SPI) as a means for monitoring the effects of fish farms on silty bottoms and found that SPI can provide very reliable information on the state of the benthic environment. The use of different levels of taxonomic resolution (Karakassis and Hatziyanni 2000) and the use of various levels of taxonomic resolution, sieve mesh size and sample size (Lampadariou et al. 2005) can also be used as a basis for cost-effective monitoring protocols for the assessment of the state of the benthic environment in the vicinity of fish farms.

An alternative method for monitoring the effects on the benthic environment would be to focus on some geochemical variables that reflect the organic content of the sediment, such as total organic carbon (TOC) or organic matter, usually measured by means of the loss on ignition method (LOI) which provides straightforward results. Hyland et al. (2005) have shown that TOC can be used as an indicator of the quality of the benthic environment since it can predict quite reliably macrofaunal diversity. On the other hand, studies on the recovery process do not show good relationships between TOC and the benthic fauna community, but instead correlate with oxygen demand, indicating that it is the labile pool of TOC controlling faunal distribution (Pereira et al. 2004). It is also worth noting that TOC or LOI values in samples taken beneath fish farms could be misleading since their concentrations at the surface layer could remain fairly constant although the depth of the farm sediment measured through sediment profiles could change remarkably with season (Karakassis et al. 1998). Also, pools in the surface layer may show significant seasonal variation, as has been found in a Danish farm, where the TOC pools in the surface layer correlated with the seasonal changes in fish production (Holmer and Kristensen 1992). At larger spatial scales, the effects of fish farms on macrofauna are rather negligible and particularly so in the case of coarse sediment sites and, therefore, it could be expected that those effects are unlikely to disturb other (remotely located) uses of the coastal zone. In both Norway and Canada monitoring systems have been constructed with more detailed analysis of the sediments, but based on relatively simple measuring techniques, which allow the fish farmer himself to follow the benthic impacts at the farms (Hangen et al. 2001; Brooks and Mahnken 2003), but in Norway the authorities require that the monitoring is performed by an independent firm or institute. In Scotland, the authorities are implementing benthic monitoring along with modelling, which strengthens the field sampling (Cromey and Black 2005). If the sampling and models deviate, field conditions are up for a more detailed examination. Monitoring in both Norway and Scotland operates within different zones around the farms, where sites at increasing distance from the farms are allowed different degrees of benthic impact (Ervik et al. 1997; Cromey and Black 2005).

The effects on water quality are probably those causing more concern regarding the quality of the marine environment. It is well known that fish farms release large quantities of dissolved nutrients in the ambient water, particularly nitrogen and phosphorus (Holby and Hall 1991; Hall et al. 1992). Furthermore, these nutrients are mainly released during the summer period when light availability is high and therefore it could be expected that phytoplankton blooms are likely to occur in the vicinity of fish farms. However, numerous studies (Pitta et al. 1999, 2006; La Rosa et al. 2002; Soto and Norambuena 2004) have failed to detect significant changes in chlorophyll a or particulate organic carbon (POC) in the water column in the vicinity of fish farms. This paradox could be attributed to the dispersive nature of fish-farming sites, i.e., to the fact that phytoplankton cells do not stay long enough to capitalize on nutrients (Gowen et al. 1983), or to experience rapid grazing by zooplankton as suggested by Machias et al. (2005). Several studies have measured significant diel changes of nutrient concentrations in the vicinity of fish farms in oligotrophic waters (Karakassis et al. 2001; Pitta et al. 2006), indicating that dispersion is a very efficient mechanism at those sites. However, it has recently been shown (Dalsgaard and Krause-Jensen 2006) that *in situ* incubation of phytoplankton and of *Ulva* sp. can be used as a relatively low-cost monitoring strategy to document the distance from the farms where pelagic primary production is affected. This method has the advantage that it is not affected by episodic events such as those affecting concentrations of nutrients and particulate material in the water column, whereas the incubation period of the bioassays allows for estimates based on integration of the water quality conditions over several days. It is worth noting that even though these bioassays have been able to detect changes up to a distance of 200–300 m from the fish farms, the intensity of these effects decreases rapidly with distance. However, when several farms are aggregated in a fish-farming zone producing thousands of tonnes, it is reasonable to ask: what are the large scale effects of this aggregation which should be detectable despite the nutrient dispersion? A recent survey in the Mediterranean (Pitta et al. 2005) showed that most of the significant changes in nutrients as well as chlorophyll a or PON were found at the deepest layer of the water column below the thermocline, indicating that they are related to the remineralization of benthic organic material.

Wild fish communities are also affected by aquaculture. Partly, this effect is related to the attraction of some fish species to the floating structures (see Chapter 3 this volume), but fish communities can also be affected at large spatial scales (Machias et al. 2004, 2005, 2006; Giannoulaki et al. 2005) probably because of the changes in primary productivity in the area and the rapid transfer of nutrients up the food web. This effect has been documented in the Mediterranean where oligotrophic conditions and the structure of planktonic communities seem to favour this process. In this context it has been suggested (Machias et al. 2005) that fish communities are probably a good indicator of the increased material flux since they are long-lived organisms integrating processes over longer time periods, and their predators are unlikely to respond promptly to an increase in their biomass.

The effects of fish farms on seagrass meadows have been documented by many recent papers (Delgado et al. 1997; Holmer et al. 2003; Marbà et al. 2006; Diaz-Almela et al. submit). In the recently finished EU-funded project MedVeg (Effects of nutrient release from Mediterranean fish farms on benthic vegetation in coastal ecosystems) four sites were monitored along the Mediterranean for benthic fauna, sediment geochemistry, water quality and seagrass-related variables. The results showed that the distance of detectable effects varied greatly among the variables used. In particular, seagrass mortality seemed to be the indicator detected at greater distance than any of the others determined in this project (Marbà et al. 2006; Frederiksen et al. 2007; Diaz-Almela et al. submitted).

This is not surprising since it is well known that, in particular, *Posidonia oceanica* is a very sensitive endemic species in the Mediterranean that has been shown to suffer population reductions due to anthropogenic stress (Marbà et al. 2005).

All the above indicate that there are many different ecological processes and biotic communities affected by aquaculture. Some of these may be easily detected and monitored, such as the effects on macrofauna, although these are usually confined to a small area beneath and around fish farms. Others, such as water quality and plankton dynamics, need new protocols for assessing the degree of change imposed by aquaculture and further research to increase our understanding of the related processes. Monitoring of fish communities seems to be a promising tool for integrating the effects at larger spatial scales although there is need for defining exact protocols, while taking account of fisheries and habitat heterogeneity. In the Mediterranean and the tropics, the effects on seagrasses are probably the most important since they are related to key ecological species with prime importance for biodiversity. However, there is a need to study further these impacts and to gather long term monitoring data in order to have a conclusive picture of the processes and the risks involved. In any case, it should be emphasized that each one of these groups of variables indicates processes operating on different spatiotemporal scales and therefore monitoring focusing only on one group can hardly be a proxy for the entire health of the ecosystem.

2.2 Monitoring Environmental Impact from Norwegian Aquaculture

2.2.1 Introduction

During the last 30 years, Norway has developed an aquaculture industry based on production of marine fish, mainly Atlantic salmon (S*almo salar* L). In 2005 580,000 metric tonnes of salmon and approximately 60,000 metric tonnes of other fish species and shellfish were produced. Norwegian aquaculture facilities are located along 2,000 km of coastline with numerous fjords and archipelagos and a temperature regime that is favourable for cultivation of cold-water species. More than 1,800 sites are located in the fjords and archipelagos where they are protected from the open sea but where water movement is sufficient to maintain production. Initially, fish farm facilities were placed in shallow areas but today many sites are located at a depth exceeding 100 m. Due to the natural conditions and a well-developed infrastructure, the coast is well suited for aquaculture. During the growth of the aquaculture industry and in concert with the increase in production, a number of environmental effects and problems have been encountered. Some of these have been minimized or resolved whereas others have increased in importance and new ones have emerged.

2.2.2 Environmental Objectives for Norwegian Aquaculture

In 1993 the Norwegian authorities decided on environmental objectives for Norwegian aquaculture, providing a national consensus (Anon. 1993). Defining the objectives was a joint project between the authorities concerned with aquaculture in Norway: the Directorate for Nature Management, the Directorate of Fisheries, The Norwegian Pollution Control Authority, The Norwegian Board of Health, The Norwegian Medicines Control Authority, and the Ministry of Agriculture Department of Veterinary Services. The report outlined the political objectives that the government and parliament had decided upon and which served as overriding objectives. The Stortings propositions no. 32 and no. 36 stated: "The development of the Norwegian aquaculture industry must be sustainable and based on respect for nature's thresholds of toleration." The report also presented the international conventions and treaties that Norway had agreed upon and which must be followed.

The environmental objectives for Norwegian aquaculture were divided into five major areas: escapees, diseases, medicines, chemicals and organic waste and nutrients. A description of each was provided and both short-term result goals and long-term environmental objectives for each type of impact were set. The report was followed by annual reports on the results achieved (e.g., Directorate for Nature Management 2000), and in 1997, the environmental objectives were reviewed (Directorate for Nature Management 1997).

The environmental objectives and the annual reports presented an important overview of the situation with regard to the environmental problem areas and provided a practical tool for following up on goals. These also made it possible to include changes in problem areas as well as to redirect focus to emerging issues, and have been used as guidelines for what should be monitored. However, they did not describe how to monitor the various effects, how often monitoring should take place, and which environmental quality standards (EQS) to use.

2.2.3 Environmental Impacts and Monitoring

A number of regulations, acts and laws administered by various ministries, directorates and other authorities regulate the Norwegian aquaculture industry with regard to licensing, production, food safety, disease control, the use of medicines and chemicals, and environmental impact (Maroni 2000).

The escape of salmon from farms is considered a serious problem since farmed fish may interact with wild salmon. To minimise the escape of fish from farms, a risk assessment must be carried out at each farm and all farms must comply with a standard for technical specifications (Anon. 2003). In the case of an escape event or suspicion of escape, the Directorate of Fisheries must be notified and recapture of escaped fish in a radius of 500 m from the farm initiated as stated in the Aquaculture Operation Regulations (Anon. 2004).

Diseases and ectoparasites have been a problem in the fish-farming industry since the beginning. Many of the major infectious diseases have however been combated by vaccination and improved hygiene, which has dramatically reduced the usage of antibacterial agents. However, sea lice infestations have proved difficult to overcome and the transfer of sea lice is still considered one of the major problems in Norwegian mariculture. At all fish farms, sea lice must be counted at least every second week when the water temperature exceeds 4°C and the results are reported to the Norwegian Food Safety Authority. If the number of lice per fish exceeds the threshold limits, the fish farmer is obliged to delouse at the farm (Anon. 2000).

All use of medicines is prescribed by a veterinarian and is registered by the Norwegian Medicines Control Authority and the Fisheries Directorate. Antibacterial agents, which were widely administered in the late 1980s, are presently only used in low amounts mainly on broodstock and early life stages (1,215 kg used in 2005, The Norwegian Institute of Public Health). Traditionally, sea lice medicines have been administered as bath treatments, first organophosphates and hydrogen peroxide and later pyrethroids, but in-feed medicines are becoming more widely used. Chitin synthesis inhibitors such as teflubenzuron and diflubenzuron were initially employed on a trial basis, but have not been used since 2002 due to their potential impact on non-target organisms. Instead, the use of avermectins has increased and 39 kg were sold in 2005 (The Norwegian Institute of Public Health). At the present there is no mandatory monitoring requirements for medicines and their residues in the marine environment. However, the environmental authorities may require monitoring with reference to The Pollution Control Act (Anon. 1981).

The most frequently used chemicals in Norwegian fish-farming are antifouling compounds for the net pens. The most common is copper, although this compound is meant to be phased out and of application should be significantly reduced before 2010 in accordance with the Declaration of The Hague of March 1990 (Anon. 1990). However, it has proven difficult to find a substitute, and there are still large amounts of copper in use. As is the case for medicines, there is no mandatory monitoring requirement for copper in the sediment but the environmental authorities may require monitoring with reference to The Pollution Control Act (Anon. 1981).

According to the Environmental Objectives of Norwegian aquaculture, organic wastes from fish farms must not result in unacceptable effects on the environment locally or regionally and permitted threshold levels of impact must be determined (Directorate for Nature Management 1997). Due to large variations in hydrographical conditions and depth at fish farm sites, the amount of organic waste that settles on the sediment will vary considerably. Furthermore, the size and the management of the fish farm will also influence the sedimentation. The impact, such as changes in sediment chemistry and in the benthic fauna community, will therefore also have a large variability between sites.

Overloading of sites and accumulation of organic material in the form of waste feed pellets and faeces can, besides the effects on the environment, be a cause of stress, poor growth and disease in the farmed fish, with the associated spread of infectious agents and need for medication. Organic material can therefore be influential for several types of environmental impact, even if the effect is greatest on the sediment under the cages.

2.2.3.1 Monitoring Benthic Impact

Parallel to the work of determining environmental objectives, a management system was developed which mainly focused on monitoring and modelling the impact of organic waste from fish farms. The system (MOM: Modelling – Ongrowing fish farms – Monitoring) combines modelling of potential impact with monitoring benthic impact and provides environmental quality standards (EQS, Ervik et al. 1997). The amount of monitoring carried out depends on the extent of the environmental impact and the EQS sets a limit for maximum allowable impact and makes it possible to distinguish between different impact levels. The monitoring programme of the MOM system (Hansen et al. 2001) has been used to make a Norwegian standard: "Environmental monitoring of marine fish farms NS-9410" (Norwegian Standards Association 2000). Mandatory environmental monitoring is performed according to NS-9410 as established in the Aquaculture Operation Regulations (Anon. 2004) and the responsible authorities are the Fisheries Directorate and the County Governor's Department of Environment. The standard describes methods for measuring bottom impacts from marine fish farms and gives detailed procedures on how environmental impacts from individual fish farm sites shall be monitored and includes EQS. All Norwegian standards are reviewed every 5 years and the Norwegian standard NS-9410 is currently under review with a new version scheduled in 2007.

NS-9410 focuses on methods for determination of sediment conditions at and in the vicinity of fish farms. Traditionally, monitoring of benthic impact at fish farm sites has been faunal community analysis. This type of monitoring is maintained in NS-9410, but mainly in the receiving water body, and at the site less time demanding and expensive surveys are used. The scientific benefit of the more advanced faunal community method was balanced against the advantage of a higher number of samples and more frequent surveys. Furthermore, due to smaller sampling gear, sediment samples can be retrieved from between net cages in compact net cage groups. Threshold values for environmental impact are set such that fish farm sites may be in use over a long period of time and aim to ensure favourable living conditions for the farmed fish as well as to prevent unacceptable impact on the surrounding area.

Presently NS-9410 describes monitoring of organic waste but sampling for medicines and chemicals in the sediment can conveniently be added.

2.2.3.2 NS-9410

The monitoring programme in NS-9410 includes three types of surveys (A, B and C investigation). The A- and B-investigations survey the potential and actual impacts on the sediment under and in the immediate vicinity of the fish farm. The C-investigation aims to obtain a picture of the impact on the receiving water body as a whole.

Table 2.5 The relationship between degree of exploitation and level of monitoring. The more severe the impact at the site, the higher the frequency of performing the A- and the B-investigations. Site condition 4 corresponds to overexploitation

	Level of monitoring (frequency of performing investigations)	
Degree of exploitation/site condition	A-investigation	B-investigation
1	every 3 months	every 2 years
2	every 2 months	Annually
3	monthly	every 6 months
4 (unacceptable)		eventual extended B-investigation

Two terms are employed to adjust monitoring depending on the impact at the site: the degree of exploitation and the level of monitoring. The degree of exploitation is an expression of the amount of impact from the fish farm compared with the holding capacity of the site. The site is overexploited if the holding capacity is exceeded and the division between acceptable and unacceptable sedimentary conditions is set as the highest level of accumulation within which burrowing bottom fauna can survive in the sediment. The higher the degree of exploitation at a site, the higher the level of monitoring that is required (Table 2.5).

The A-investigation consists of a simple measurement of sedimentation rate on the sea floor under a fish farm, and can give information on high point-source loading. The survey is easily done and is carried out by the fish-farmer himself. The survey gives information on potential bottom loading and is particularly useful in combination with the B-investigation. EQS are not used in the A-investigation.

The B-investigation comprises a simple trend monitoring of the bottom conditions under a fish farm. Because the survey is repeated regularly, at intervals determined by the extent of the environmental impact, the development of the environmental impact can be followed closely. At least ten grab samples are collected at the site and both the average condition at the site and the conditions under different parts of the fish farm are revealed. The B-investigation comprises three groups of sediment parameters: (1) presence or absence of animals larger than 1 mm in the sediment, (2) pH and redox potential, and (3) qualitative determination of outgassing, smell, consistency, colour of the sediment, grab volume and thickness of the layer of deposits. All parameters are assigned points according to the extent to which the sediment is affected by organic material. The points are added and the higher the sum the more affected the sediment. Since many parameters are used in concert, the survey is less sensitive to anomalies in individual parameters. EQS have been established which divide the sediment condition into four categories equivalent to the four degrees of exploitation (Table 2.5).

The C-investigation is a survey of the bottom conditions at the fish farm and outwards into the receiving water body. The main element is a survey of the bottom faunal communities, carried out according to another Norwegian Standard: “Water quality – Guidelines for quantitative investigations of sublittoral soft-bottom benthic fauna in the marine environment NS-9423”, which describes guidelines for sampling and sample processing of macrofauna in soft sediments (Norwegian Standards Association 1998). In addition, information is obtained on other parameters that may be used to determine if organic material is of fish farm origin. The pollution control authorities have defined threshold values for environmental quality of fjords and coastal waters (Molvær et al. 1997), and these are applied to the C-investigation in the receiving water body. However, specific threshold values are provided in NS-9410 when the investigation is made close to the farm.

Both the B- and the C-investigations are carried out by private firms and research institutions.

2.2.4 Models and Coastal Zone Planning

The use of models is not compulsory in environmental regulation of Norwegian aquaculture, but models have been developed which may be helpful. In conjunction with the development of the MOM monitoring programme, a model was made to estimate the maximum production of fish that could be allowed at a site without exceeding the holding capacity at the site (Stigebrandt et al. 2004). The model comprises four sub-models (a fish model, a water quality model, a dispersion model and a benthic model) and is linked to a previously developed model on environmental quality in fjords (Aure and Stigebrandt 1990). The sub-models can be altered individually as new knowledge is acquired or as new management procedures or fish species are introduced. The scope of the model system may also be expanded to include other environmental effects of fish farming related to the use of chemicals and medicines. The model was developed so it can be utilised by both environmental administrators and fish farmers.

Additionally, a growth and advection model for pelagic sea lice copepods has been developed (Asplin et al. 2004). The dispersion of sea lice in coastal waters and fjords depends on the production of sea lice larvae, and thus is influenced by farmed fish at various locations, and by the hydrography of the waters and currents, which are in turn greatly influenced by the wind. The model is currently being tested and so far the results of the model have compared well with observations in a major fjord (Sognefjorden).

In the future, environmental impact is expected to gain increasing focus as the competition for space and resources in the coastal zone grows. Sustainability and integration with other coastal activities are therefore fundamental for the viability of the aquaculture industry. In Norway, a system is under development that covers both the planning and the operational phases of aquaculture, and which can ensure an efficient use of areas available for aquaculture and can adjust the environmental impact of the industry to the holding capacity of the area. Information on topography and hydrography, as well as an overview of allocation of different uses and environmental status, will be combined with simulation models to locate aquaculture activities and to adapt the environmental impact to local and regional conditions. Monitoring will be an important element, which will ensure that the holding capacity is not exceeded.

2.3 Monitoring the Environmental Impacts of Aquaculture in Malta

2.3.1 Introduction: Development of Aquaculture Activities in Malta

Aquaculture on an industrial scale started in Malta around 1991, following initial land- and sea-based experimental and pilot projects undertaken in the mid-1970s and early 1980s. During the period 1991 to 2000, the activity mainly involved culture of sea bass (*Dicentrarchus labrax*) and sea bream (*Sparus aurata*) in offshore cages located a few hundred metres or less from the shore. Production of these two species increased steadily from around 100 tons in 1991 to 2000 tons in 1998 (Axiak et al. 1999). The offshore sea bass and sea bream farms, owned by some six different operators (Schembri et al. 2002), were sited in eight localities, all of which were relatively sheltered and supported extensive seagrass (*Posidonia oceanica*) meadows. Water depth at the different fish farm sites ranged between 10 m and 22 m. On the other hand, land-based coastal aquaculture activities contributed only 2% (equivalent to an annual production of 50 tons of sea bream) to the total local aquaculture production. However, the land based operations also included two hatcheries for sea bream, one of which was located within the National Aquaculture Centre and which at peak production was contributing up to 2.5 million sea bream fry per year, most of which were exported to Europe (Axiak et al. 1999). In the late 1990s, strong competition from fish farms based on mainland Europe, and the high operational costs incurred by local farms, particularly freight and levy charges imposed on the exported product, led to a general decline in culture of sea bass and sea bream, and the attention of some local fish farmers turned to the relatively new and lucrative activity of tuna-farming (Schembri et al. 2002).

Tuna farming, also commonly referred to as "tuna penning", is a relatively recent but highly successful enterprise that was introduced to Europe in 1979 and adopted on a large commercial scale in the 1990s. Tuna farming is classified as capture-based aquaculture and differs from traditional aquaculture in that the farmed stock is derived from catches taken from wild populations, while the captive tuna are fed fresh fish (e.g., herring and mackerel) (Ottolenghi et al. 2004). In Europe, tuna penning has been (to date) restricted to the Mediterranean, where the main species farmed is *Thunnus thynnus*, the Atlantic Bluefin Tuna. The intensity of tuna farming has increased steadily in the Mediterranean over the last decade or so, reaching a current total annual production of around 16,000 tons. However, information on the influence of tuna farming on the marine environment, both outside and within European coastal waters, is somewhat lacking and there is a dearth of published data on the environmental impacts of the activity, while data on potential adverse effects resulting from indirect activities, for example, the impacts of the baitfish fishery that supplies the fresh feed for tuna, is unavailable. Moreover, in view of the large and increased catch effort of tuna fishers to meet the farms' demand, information on the potential adverse impact of tuna penning on wild stocks of *Thunnus thynnus* is unavailable and this has placed the activity at the centre of much controversy and debate, and harsh criticism has been levelled at it by national and international environmental NGOs (e.g., the World Wide Fund for Nature; see WWF 2004).

The advent of tuna farming in Malta in 2000 raised concerns at the Malta Environment and Planning Authority (MEPA – the local agency concerned with environmental protection in Malta) and the public, mostly because of the potential adverse impacts resulting from the large scale of operations of the projected tuna farms. As a result, MEPA stipulated that tuna cages must be sited at least 1 km offshore, in waters having sufficiently strong water currents, and distant from benthic habitats that have a high ecological value (e.g., seagrass meadows and maerl beds). Furthermore, prior to granting tuna farm operators a development permit, MEPA requested that appropriate surveys be carried out in offshore areas having the required characteristics in order to determine the specific location of the cages. The result was that all four tuna farms that started operations in the early 2000s were sited in waters having a depth of around 50 m, and had their cages located over a "bare sand" habitat. When initiated locally in 2000, tuna farming had an annual production of 300 tons that increased steadily to around 3000 tons in 2005, making the country one of the largest current producers of farmed tuna in Europe. The three farms that are currently operating are located off the northeastern coast of the island of Malta (Fig. 2.2) at a distance of around 1 km from the shore, on a seabed consisting mainly of bare soft sediment (muddy sand), and in waters characterised by strong currents and having a depth of between 46 m and 55 m. Tuna penning activities usually start around July and extend to December/February, after which there is a 4–6 month fallowing period.

Recently, the Fisheries Conservation and Control Division of the Ministry for Rural Affairs and the Environment (responsible also for local aquaculture) lodged an application with MEPA to designate an "Aquaculture Zone" located about 6 km off the eastern coast of the island of Malta (Fig. 2.2). The zone covers an area of some 9 km^2 and is located in waters having depths of between 65 m and 105 m. The aim is to locate future tuna penning installations in one offshore area that is distant from land in order to minimise impacts on the coast and the shallow waters off it. Following approval by MEPA, tuna penning operations within this zone started in July 2006 in a sea area of 3 km × 1.5 km.

2.3.2 Environmental Monitoring of Sea Bass and Sea Bream Farms

At around the same time that local aquaculture activities reached industrial production levels, the central government set up a Planning Authority as the main regulatory body for development, and this institution later merged with the then Environment Protection Department and changed its name to the Malta Environment and Planning Authority (MEPA) to become the local planning, development control and environmental

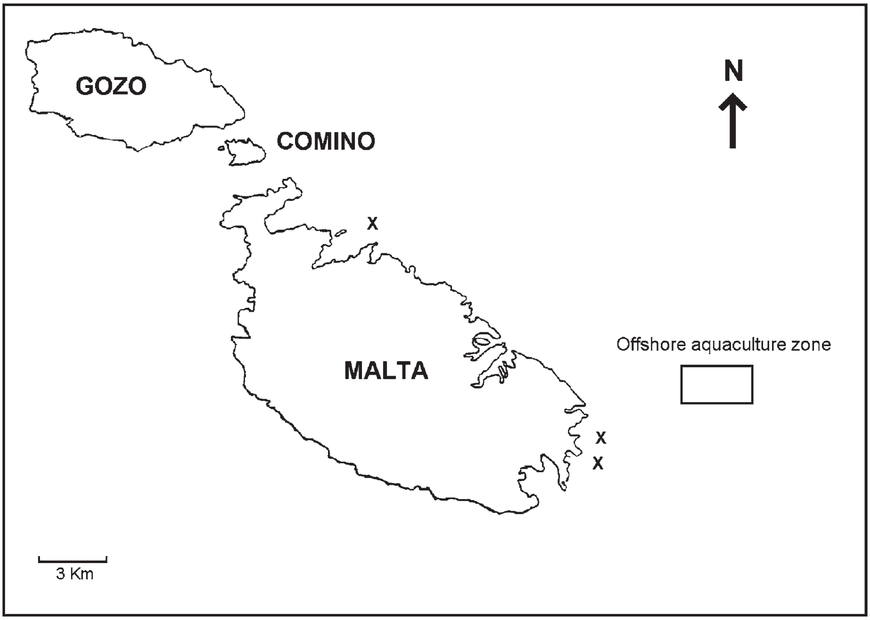

Fig. 2.2 Map of the Maltese islands showing the location of the three currently operating tuna farms (x) and the recently designated offshore aquaculture zone

protection agency. Since its establishment in 1992, MEPA, as its predecessors, was granted the overall responsibility of processing aquaculture development proposals and to oversee any required environmental monitoring of the activity. Consequently, in 1994, the then Planning Authority issued a set of Policy and Design Guidelines for Fish-farming (PDGF; see Planning Authority 1994). The "monitoring" chapter of the guidelines required an environmental monitoring programme for each fish farm to enable assessment of the impact of the activity on the environment. According to the PDGF, the environment monitoring programme should:

- measure changes, if any, in specific environmental attributes such as currents, temperature, dissolved oxygen, and levels of nutrients and bacteria;
- monitor the state of benthic assemblages and habitats, and accumulation of waste products in the vicinity of the farms;
- record material introduced in the environment by the fish farms (such as chemicals and physical items forming part of the cage structures);
- record other inputs and impacts on the environment in the general area of the fish farms, but which are not directly related to the fish-farming activities (e.g., discharges from outfalls and other major sources of pollution, and fishing activities).

The guidelines also stated the specific physico-chemical and biological attributes to be monitored, the frequency of data collection, and the number of sampling points, which were to vary depending on the size and location of the respective fish farm (Table 2.6).

Table 2.6 Details of specific attributes, sampling stations and frequency of the required environmental monitoring programme for aquaculture activities in Malta, as required by the Malta Environment and Planning Authority (source: Planning Authority, 1994)

Environmental attributes to be monitored	No of sampling stations	Monitoring frequency
Currents (speed and direction)	1–2 stations at various depths	Every 2 months
Water column: temperature; salinity; dissolved oxygen; turbidity; chlorophyll a; nitrates; phosphates; ammonia and total bacteria	Several stations at various depths	Every 2 months
Sediments: granulometric properties; and organic carbon and organic nitrogen content	Several stations	Every 6 months
Benthos: benthic habitats and communities	Mapping of all benthic communities within the area occupied by the cages and their moorings, together with collection of samples at stations as necessary to establish the species composition of benthic communities	Every 6 months

To enable assessment of changes in the monitored environmental attributes after initiation of the aquaculture activities, the guidelines required collection of baseline data before the start of the operations, so that these may be used as a reference against which to compare data collected during monitoring. Collection of baseline data would form part of the Environment Impact Assessment (EIA) as a requirement for the granting of a development permit. Furthermore, the guidelines specified that an environmental monitoring report, including all raw data collected, should be presented to MEPA and the Directorate of Veterinary Services. Submission of reports to the latter agency would ensure that appropriate practices in relation to health management of fish stock and product quality are in operation.

Environmental monitoring at sites supporting sea bass and sea bream farms was initiated around 1994, however, this was sporadic and certainly did not satisfy MEPA's guidelines to the full (Schembri et al. 2002). In some cases, the required baseline survey for a specific fish farm was made (as this could not be avoided since it formed part of the development application process), but no monitoring was undertaken following initiation of the fish-farming activities, while other farms claimed to have carried out monitoring of water quality at their own laboratories. Apparently, no data on currents has been collected at any of the fish farm sites. As a result of the irregular and incomplete environmental monitoring for sea bass and sea bream farms, the data available for these operations is scanty.

Since all sea bass and sea bream farms were located in the vicinity of *Posidonia oceanica* meadows, monitoring of benthic habitats and assemblages centred on assessing the spatial distribution, coverage and state of health of the seagrass within the area occupied by the cages and their moorings. This essentially consisted of mapping surveys of the seabed to assess potential changes in the spatial distribution and coverage of seagrass habitat resulting from the fish-farming activities.

In an attempt to fill gaps in knowledge of the environmental impacts of sea bass and sea bream farms, a number of studies were undertaken by the University of Malta, most of which formed part of undergraduate and postgraduate research projects (e.g., Cassar 1994; Dimech et al. 2002). Some of these studies included collecting data on environmental attributes that went beyond the minimum requirements set by MEPA's PDGF, for example, measurement of *P. oceanica* meadow and shoot attributes (shoot density, mean number of leaves and leaf length per shoot, shoot biomass and shoot epiphyte loading; see Cassar 1994; Dimech et al. 2002).

Overall, the results of environmental monitoring at sites used to farm sea bass and sea bream indicated that seagrass meadows located directly below the fish cages underwent severe regression or were completed decimated, and that the effects of the aquaculture activities on the monitored seagrass attributes (shoot density, mean leaf length, number of leaves per shoot, and epiphyte load) extended a considerable distance (in the case of one farm, around 200 m; Dimech et al. 2002) from the farm site. The results of a study aimed at assessing the impact of sea-based fish cages (located in waters having a depth of 10 m) on the decapod, mollusc and echinoderm fauna associated with *P. oceanica* beds indicated the presence of three distinct zones in the vicinity of the farm (Dimech et al. 2002):

(i) Zone 1, comprising the area occupied by the cages and an additional band of 30 m around the farm. The macrofaunal assemblages present within this zone were characterised by a low species richness and the dominant trophic groups were grazers and deposit feeders (decapods, polyplacophorans and gastropods), which exploit the abundant epiphytes and deposited organic matter present close to the cages.

(ii) Zone 2, comprising the area located at a distance of between 30 m and 90 m from the farm. This zone supported macrofaunal assemblages that had the highest species richness and abundance, while the fauna was dominated by the same trophic groups in Zone 1.

(iii) Zone 3, comprising the area located at distances exceeding 90 m from the farm. This zone supported macrofaunal assemblages having species richness and abundance values that were intermediate between those recorded from Zones 1 and 2, and in which the dominant trophic groups comprised grazers, deposit feeders, suspension-feeders (mostly bivalves) and predators.

This "zoning" pattern, consisting of differences in the species composition and structure of the benthic macrofaunal assemblages with increasing distance from the farm site, is very similar to that recorded in the vicinity of offshore salmon farms (e.g., Brown et al. 1987; Ye et al. 1991).

2.3.3 Environmental Monitoring of Tuna Penning Activities

As part of the permit conditions issued by MEPA for tuna penning activities, the farm operators were required to commission a comprehensive environmental monitoring programme to be carried out by independent consultants approved by MEPA. Since the environmental characteristics at the tuna penning sites were very different from those of the near-shore sites where sea bass and sea bream farms were located, while the type and scale of operations were also very different, it was immediately realised that MEPA's environmental monitoring guidelines contained in the 1994 PDGF could only be applied to tuna farms following modification. Given the circumstances, in 2001 MEPA amended the 1994 PDGF such that the revised guidelines stated that no aquaculture development would be considered in areas less than 1 nautical mile from the shore, or in sites having a water depth less than 50 m (give or take 5 m) (Planning Authority 2001).

In granting development permits for tuna-penning activities, MEPA requested that monitoring of tuna penning activities should include monitoring of: (1) sediment attributes; (2) benthic diversity; (3) the gross physical and biological characteristics of the seabed below the tuna cages through underwater videography; (4) the state of seagrass beds and of biological characteristics at important dive sites located in the vicinity of the farms using underwater mapping and videography (in some cases, even if these were present at distances exceeding several hundred metres from the tuna cages); and (5) water quality. The specific requirements for environmental monitoring of aquaculture operations in the PDGF of 2001 are given in Table 2.7. The environmental monitoring programmes for all tuna farms were initiated in 2000 and are still ongoing.

Samples to monitor sediments and benthic diversity have been collected annually (since 2000) at each tuna farm from a number of stations located: (1) adjacent to the tuna-pens, (2) at a distance of some 100 m from the tuna pens, and (3) at a number of reference sites; the sampling programme being mainly based on a Before-After-Control-Impacted (BACI) design (Borg and Schembri 2005). Using this design, an adverse impact is deemed to have occurred if a significant change (at the 0.05 level of significance) for one or more of the monitored attributes is recorded between the baseline condition and that following the tuna penning activities. In the case of benthic diversity, this would be a significant decrease in the total number of species and/or abundance of the selected indicator species.

Monitoring of the gross physical and biological characteristics of the seabed below the tuna cages is being undertaken through surveys carried out by SCUBA divers using direct observation and underwater videography. During initial surveys of the seabed below the tuna cages, it was immediately realised that the main impacts on the seabed resulted from the presence of large amounts of uneaten feed-fish that accumulated on the seabed below the cages. However, the amount of feed-fish below the cages varied greatly, even between cages within the same farm. It was therefore considered appropriate to develop an index to enable an objective semi-quantitative assessment of the amount of uneaten feed-fish present (Table 2.8).

Table 2.7 Details of specific attributes to be monitored, sampling stations, and frequency of the required environmental monitoring programme for aquaculture activities in Malta (source: Malta Environment & Planning Authority 2001)

Environmental attributes to be monitored	Number of sampling stations	Monitoring frequency
Water column: temperature; salinity; dissolved oxygen; turbidity; chlorophyll a; nitrates; phosphates; ammonia; faecal coliforms and total bacteria	A sampling site underneath each cage; Sampling at points along a perimeter around the cage site 25 m away from the cages; At least two sampling points 100 m away from the cage site (according to the direction of the prevailing currents) Sampling sites in areas that are of ecological, commercial, tourism, or recreational interest (this is to be decided on a site-by-site basis)	Monthly, for as long as the fish are kept in the cages
Sediments: granulometric properties; and organic carbon and organic nitrogen content	Several stations within the cage site	Annually, in the same month each year
Benthos: species diversity; photographic/video evidence regarding the state of the seabed; mapping of benthic communities; core samples for faunal, granulometric and sediment analysis as described for sediments above; seagrass morphological parameters (e.g., shoot and leaf density, shoot length, etc.) where applicable	Mapping of all benthic communities within the area occupied by the cages and their moorings, together with collection of samples at stations as necessary to establish the species composition of benthic communities	Annually

Table 2.8 The "uneaten food index" devised by Borg & Schembri (2001) for the purpose of quantifying and comparing the amount of dead uneaten feed-fish under the different tuna-pens

Index value	Description of amount of uneaten feed-fish present on the seabed
0	No uneaten feed-fish present
1	<1 uneaten feed-fish present per m^2 of seabed
2	>1 uneaten feed-fish present per m^2 of seabed, but the fish do not form a continuous layer covering the seabed
3	>1 uneaten feed-fish present per m^2 of seabed. Fish form a single, uninterrupted layer within at least a 1 m^2 area on the seabed.
4	>1 uneaten feed-fish present per m^2 of seabed. Fish form two or more uninterrupted layers on top of each other within at least a 1 m^2 area on the seabed.

The general state of seagrass beds and habitats, including those at popular dive sites, is being assessed through mapping surveys and underwater videography, carried out by SCUBA divers. During the surveys, the divers record the state of health and spatial extent of the main marine benthic habitats in the respective study area. Potential changes in the state of the benthic habitats and their spatial distribution are assessed by comparing maps of the situation recorded before initiation of the tuna-penning activities with that after each monitoring session.

Monitoring of water quality consists of surveys of the same physico-chemical and bacteriological attributes that have been monitored in the vicinity of sea bass and sea bream farms. Samples of water for these surveys are being collected at depths of 1 m and 5 m below the surface at several stations located in the immediate vicinity of the tuna farms and at reference stations located at a distance from the tuna cages.

Monitoring at the new offshore aquaculture zone (Fig. 2.2) commenced in June 2006 prior to the start of tuna penning activities there. The baseline survey for the sediments and benthic diversity monitoring components was based on the same design used at the other three tuna farms located closer to the coast, with samples being collected remotely using a standard 0.1 m^2 Van Venn grab. However, because of the deep waters that characterise the area, monitoring of the seabed using the same underwater videography and SCUBA diving techniques that have been used to date at the other tuna penning sites located in shallower waters, is not possible, and it is planned to use remotely operated video cameras instead.

Overall, the results of the various monitoring components undertaken since 2000 for the three tuna farms located 1 km offshore indicated that, where detected, the main adverse impacts resulted from accumulation of large amounts of feed-fish on the bottom under and in the vicinity of the cages. The results from the video surveys carried out near the tuna pens indicated that, towards the end of each penning season (in autumn), considerable amounts of dead uneaten feed-fish were present on the seabed directly below the tuna pens, and this resulted in alterations in the physical and biological characteristics of the seabed under the cages. The recorded changes in biological characteristics included the disappearance of certain megafaunal species (e.g., the irregular sea urchin *Spatangus purpureus* and the crinoid *Antedon mediterranea*) that prior to the start of the penning operations were characteristic of the soft sediment habitat where the tuna pens are located, and the appearance of high population densities of detritus-feeding and scavenging macroinvertebrates (e.g., the ophiuroid *Ophiura texturata* and the crab *Inachus* sp., and the fish *Gobius* sp.). Gross changes in physical characteristics of the seabed included the presence of large quantities of fish bones and a few anthropogenic items originating from the tuna farms. The video surveys also showed that the amount of feed-fish present varied considerably between different tuna farms and between cages within the same farm, with some cages only having a few fish beneath them and others having multiple layers. Overall, a consistent pattern was evident where a decrease in the amount of uneaten fish occurred only when tuna were no longer present in the pens during the fallowing period. The remaining uneaten fish decompose slowly and, where the uneaten fish are present in large numbers, form a continuous layer of decomposing organic material that continues to decay gradually. Sometimes, following storms and possibly due to strong bottom currents, this layer is admixed with the underlying mobile sediment. In places where the decomposition process is complete, the only remains are fish bones that eventually disperse in the sediment leaving little or no trace of the original uneaten fish on the surface. Once the source of the impact (periodic addition of new uneaten food) is removed, the slow recovery to the original state is signalled by the reappearance of some of the megafaunal species that formed part of the original benthic assemblage characterising the bare muddy sand bottom over which the tuna pens are located (Borg and Schembri, unpublished data).

The results of benthic diversity monitoring indicated that, at times, a significant decrease in species richness, and in the abundance of the indicator macrobenthic species, occurred in the vicinity of particular tuna farms, but this effect was mainly restricted to the area directly below the cages. Similarly, significantly higher levels of organic carbon and/or organic nitrogen and/or significant changes in mean sediment grain size were recorded in some of the monitoring sessions, but the observed changes were again mainly restricted to the seabed area directly below the cages.

The mapping and videographic surveys of important habitats and dive sites located in the vicinity of the tuna farms did not detect any changes in the physical and biological characteristics of the monitored sites. Likewise, the water quality studies did not show any consistent trend in the levels of the monitored variables that could be attributed to the tuna penning activities (Schembri et al. 2002). Lower levels of oxygen, reduced water transparency, and elevated nutrient levels were at times recorded at the tuna penning sites relative to the reference sites during the farming season (July – December), however, the observed changes in the monitored variables were sporadic and not statistically significant. Data collected in June 2006 from the new offshore aquaculture zone are still being analysed and consequently, results from the monitoring programme for tuna farms located within this area are not yet available.

2.3.4 Conclusions and Recommendations in Malta

Guidelines for environmental monitoring of aquaculture activities in Malta were issued by the responsible local agencies relatively early during the period of initiation and expansion of local fish-farming involving culture of sea bream and sea bass. However, most fish farms failed to adhere to the environmental monitoring requirements, at least on a regular basis, while it appears that enforcement was not effective (Schembri et al. 2002). As a result, few monitoring data on the impact of sea bream and sea bass aquaculture activities on the marine environment exist. Where data are available, the results of benthic environmental monitoring indicated an overall adverse impact on seagrass beds in the vicinity of sea bream and sea bass cages. However, site characteristics such as the current regime, water depth and exposure, together with the size of the fish-farming operation and the farm management programme at a specific locality, appear to be crucial in determining the magnitude of the adverse impact. The results of the water quality monitoring programmes for sea bream and sea bass farms did not indicate any large adverse changes in water quality attributes resulting from the fish-farming activities.

There is currently only a low level of production of sea bream and sea bass in Malta, as the attention of aquaculture operators is presently on tuna penning. However, some operators still retain a permit to culture these species in addition to penning tuna (Schembri et al. 2002), while it is likely that production of sea bream and sea bass, as well as of additional species that are being introduced into aquaculture in the Mediterranean, will increase in the future, depending on the vagaries of the market for tuna and other species, and as new operators enter the field and wild tuna stocks dwindle. The environmental impact of any new (non-tuna) farms is not likely to be as severe as that of the early sea bream and sea bass farms since it is unlikely that such farms will be allowed to locate inshore or close to sensitive habitats, particularly since the technology for siting farms in deep water now exists, and because the environmental impact monitoring requirements of aquaculture projects are nowadays much more rigidly enforced.

Overall, the results of environmental monitoring of tuna penning operations during the last 6 years (2000–2006) revealed a consistent pattern of a localised adverse impact that mainly resulted from the uneaten feed-fish which accumulate on the seabed during the tuna farming season (July to December). The amount of feed-fish present decreases only when all the tuna have been harvested, following which, any feed-fish remaining on the seabed continue to decompose slowly. These results are characteristic of a "pulse disturbance" where the physical and biological characteristics of the seabed are temporarily altered during the tuna penning season but return back to more or less the pre-disturbance condition before the start of the next tuna penning season. Nonetheless, repeated accumulation of feed-fish on the seabed in the vicinity of the tuna pens may prevent complete recovery of the benthic assemblages following each tuna penning season, potentially leading to a "press disturbance" where environmental conditions become permanently altered.

The observed differences in the amount of feed-fish present on the seabed below the cages indicate potential differences between different tuna farms and/or cages within the same farm in: (1) feed management, or (2) the rate of food intake by the tuna, or a combination of (1) and (2). It appears that the key to preventing this from happening is to implement a rigorous feed-management strategy that includes:

- careful monitoring of the feeding behaviour of the tuna and stopping the supply of food as soon as the tuna are satiated in order to avoid as much as possible uneaten food ending up on the bottom; and
- removal of dead uneaten feed-fish from the bottom should inordinate amounts accumulate below the cages either due to overfeeding or to accident.

On the other hand, the results of recent (2004–2005) monitoring surveys indicated an overall large improvement in feed-management at local tuna farms. For example, values of the index for uneaten feed-fish (Table 2.8) recorded in 2005 averaged 1, compared to values of between 3 and 4 recorded in 2001–2003. Furthermore, it should be emphasised that all tuna farmers have, in general, adhered to the environmental monitoring requirements, while one particular operator has even taken the initiative of including details of the monitoring and results obtained on their web site (e.g., http://www.ajdtuna.com/). The possibility of developing an alternative feed source for tuna should be explored, as this could potentially reduce adverse impacts on the seabed, while alleviating fishing pressure on wild stocks of feed fish.

While the accumulation of decomposing organic matter on the seabed is the key source of marine benthic impact of the Maltese tuna farming operations, it is not the only potential adverse factor. Mass deaths of tuna have occurred at least on two separate occasions, however, it seems that the farm operators have taken remedial action and recovered the carcasses from the seabed at the earliest opportunity; thus during the 6 years of monitoring, tuna carcasses were only encountered near the cages on two or three occasions, and then as single dead fish. The accidental introduction of anthropogenic items, most of which are related to the tuna-penning activities, is also of concern. This can be mitigated relatively easily by enforcing a strict policy of not throwing anything into the sea and by implementing periodic "clean-ups" of the seabed. Additional impacts result during feeding and harvesting of the tuna, when entrails and oily slicks transported by surface currents have been reported. These observations highlight the importance of guidelines for operational procedures and mitigation measures to reduce adverse environmental effects on the marine environment.

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