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Implications for the environment of using adaptive feeding systems in the cage culture of Atlantic salmon

Thesis submitted for the degree of Doctor of Philosophy

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Declaration

I hereby declare that to the best of my knowledge and belief, the work presented in this thesis is original and entirely my own work, and that where the work of others has been quoted, it has been given due acknowledgement. The work presented in this thesis was carried out between October 2000 and October 2003. The material presented in this thesis has not been submitted, either in whole or in part, for a degree at this or any other university.

Richard Anthony Corner
December 2004

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Abstract

The use of adaptive feeding systems to deliver feed remotely to Atlantic salmon (*Salmo salar*) cages has the potential to improve the localised environment through a reduction in particulate waste. This can be achieved through improved growth and lower Feed Conversion Ratio (FCR). The aim of this project was to assess whether adaptive feeding systems confer any environmental benefit at salmon farms through by comparing two fish farm sites, one that uses a Computer Aided System (CAS) adaptive feeding system (AKVAsmart UK limited, Inverness, Scotland) (Portavaide fish farm) and one using hand feeding (Rubha Stillaig).

This investigation comprised of 3 elements: 1) a comparative assessment of the quantity and nutrient composition of particulate waste material emanating from the cages; 2) collection of benthic samples plus a video survey along transects at each site including a reference station, with an analysis of differences in benthic fauna, sediment grain size and sediment nutrient composition; and 3) comparison of the distribution of waste under each feeding regime using a GIS-based modelling approach.

Particulate waste was collected via sediment traps. Uneaten feed was caught in only 3 out of 184 separate collections and thus no estimate of feed loss for each feeding system could be made. Samples were analyzed for total solids (TS), faecal solids (FS), faecal carbon (FC), faecal nitrogen (FN) content and faecal sedimentation rate (FSR). The highest deposition occurred under the cages and decreased with increased distance from the cage centre. Maximal deposition of TS at Portavadie was higher than at Rubha Stillaig when feed was included, although average TS, FS, FC and FN per tonne of production did not significantly vary between sites. Carbon sedimentation rate was analyzed using regression analysis and a General Linear Model Factorial ANOVA on faecal waste only and showed no significant differences between sites and, therefore, no difference between feeding methods.

There were no differences observed in the diversity and abundance of benthic species under the two feeding systems. By the end of the production period all stations out to 25m from the cage edge were dominated by *Capitella capitata* at both sites, this species proving a useful indicator of the impact of nutrient deposition. The analysis suggested that *Heteromastus filiformis* and *Corophium* sp. provided useful indicators of the onset of nutrient enrichment. Measurement of carbon and nitrogen levels and particle size in sediment showed no difference between sites. Variations between sites in species abundance and diversity and sediment carbon and nitrogen levels reflected the different sediment conditions prevalent at the start of the sampling period. Univariate and multivariate analysis showed there was no difference in species diversity and abundance between the sites as a result of using adaptive feeding systems.

Horizontal cage movement, measured at up to 10m, reduced the predicted settlement under the cage by 23% and 11% for feed and faecal distribution respectively. There was no significant difference in the predicted settlement of waste particulates under adaptive and hand feeding. The GIS model prediction of carbon flux (g C m^{-2} 15-$days^{-1}$) was validated for faecal settlement using sediment trap data where predictions agreed well with observations from Portavadie fish farm, with an accuracy of ± 53.1% when all stations were included, improving to ± 27.6% when deposition under the cage was excluded.

Overall, the approaches used did not identify specific differences between sites that used adaptive feeding and hand feeding methods. The growth period using the adaptive feeding system was approximately nine weeks shorter than under hand feeding, however, which could be used constructively to increase the fallowing period whilst maintaining current levels of production. This would benefit the localised benthos by increasing the time available for recovery before further production takes place and thus the CAS Adaptive Feeding System could be used as part of a broader sustainable farming strategy for fish culture.

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General introduction

1.1 Introduction

The clearest definition of aquaculture is provided by the Fisheries and Agriculture Organisation (FAO) (1997), as "the farming of aquatic organisms including fish, molluscs, crustaceans and aquatic plants. Farming implies some form of intervention in the rearing process to enhance production, such as regular stocking, feeding and protection from predators. Farming also implies individual or corporate ownership of the stock".

The global production of aquatic species is dominated by finfish and shellfish production with 39.8 million tonnes produced in 2002 (FAO, 2004). Of the finfish 57.8% are cultivated in fresh water (FOA, 2004) using low value species such as carp and tilapias and employing so called "extensive" production methods that require only limited intervention in the rearing process. Other extensively produced species includes shrimp reared in ponds, where the food source comes directly from the pond itself.

Conversely, high value fish species, such as salmonids, sea bream, sea bass and yellowtail, are cultured using intensive methods. A high investment in infrastructure and equipment, the use of high stocking densities and the use of formulated feed to provide all nutritional requirements are key features of intensive culture (Beveridge, 1996). All of these species are either entirely marine or diadromous, spending a high proportion of their lives in seawater. Although aquaculture production of eggs and juveniles occurs in land-based facilities, the majority of the growth period (commonly called on-growing) is conducted in floating cages sited on the near shore in the marine environment.

This review focuses on the production of Atlantic salmon (*Salmo salar* L.) in marine floating cage culture. Salmon aquaculture is highly significant with >99% of the total world market met by the culture industry (FOA, 2004). Cage culture of salmon is concentrated in Norway and Chile; and in Scotland, where it is particularly important for the Scottish economy. Scotland produced 173,373 tonnes in 2003 (Stagg and Allan, 2004) and contributes a significant proportion of the income in rural communities (Taylor *et al*, 1998). However, fish farming also has a real and potential impact on the marine environment.

Fish farming generates significant quantities of dissolved and particulate waste, a function of metabolic processes and the open production system (Bergheim and Asaud, 1996). The effects of dissolved waste from fish farms on the water column and the potential for enhancing algal blooms and eutrophication are not yet satisfactorily identified (Gowen, 1994; Tett and Edwards, 2002). Waste feed and faecal particulates are rich in nutrients and have the potential to cause deleterious impacts on the marine environment (e.g. Findlay and Watling, 1997). However, the action of deposition may or may not be deleterious, with other factors such as bathymetry and hydrography also influencing the degree of environmental degradation. Nutrient deposition can affect the fish farming industry directly, through self-pollution, and can also have a negative impact on water and sediment chemistry and biota. In particular, lowering sediment oxygen levels and production of H_2S and CH_4 in the sediment at impacted sites, alters sediment chemistry and reduces the diversity and abundance of benthic flora and fauna (Black *et al*, 1996[a]).

A reduction in waste per unit volume of production has been achieved in recent years by developments in feed formulation and an improved understanding of the feeding behaviour of Atlantic salmon (Kadri *et al*, 1996; Cho and Bureau, 1997). More recently new technological developments have been introduced, notably "Adaptive Feeding Systems" (Blyth *et al*, 1993; Ang and Petrell, 1998), which have the potential to use feed rations more effectively and to reduce the amount of particulate waste entering the marine environment. Such systems are adaptive because they respond to feeding via a computerized feedback loop that adjusts the feeding strategy to the feeding response of the fish. However, there is no real clear understanding of the environmental implications of this technology. Hence a programme of work was planned that would investigate and quantify the amount of waste entering the sediment under cages and to determine if the environment is less impacted when an adaptive feeding system was being used.

1.2 Aquaculture production

1.2.1 World fish and shellfish production

Over 20,000+ species of fish have been described world wide (Bone *et al*, 1995) and of these FAO (2004) list 200+ species that are cultured on a commercial basis. As technology develops and as commercial catches peak, the number of new species introduced to cultivation is increasing and will continue to do so. Aquaculture continues to be one of the fastest growing food production sectors (New, 1999) and in 2002 fish and shellfish production was 39.8 million tonnes (FAO, 2004) compared to a capture fisheries production of approximately 93.2 million tonnes (FAO, 2004). The rate of increase is particularly fast for relatively high value species such as sea bream, halibut, cod and sturgeons. Mariculture, the farming of species in the marine environment, accounted for 50.8% of all aquaculture production in 2002, up from 36.9% in 1996 (FAO, 2004).

China dominates the culture of fish species, in general, with 2 out of every 3 fish produced coming from this country (FOA, 2004). India is the other large producer with 6.7% of the global share (FOA, 2004). Typically, production in these countries is maintained in relatively small-scale fresh water ponds to satisfy localised socio-economic requirements (FOA, 2000). Species include carp and tilapia that require the addition of little or no feed, have low capital investment costs and employ extensive farming practices, often in a polyculture combined with livestock and/or rice production, for example (FOA, 2000).

This is in contrast to the production of salmonids, seabass, seabream, yellowtail and other high value products that are mainly produced in the West and rely almost exclusively on intensive monoculture. Atlantic salmon is an example of an intensively cultured fish species in temperate waters and is the focus of the remainder of this review.

1.2.2 Atlantic salmon (*Salmo salar*) production

Atlantic salmon is a diadromous fish species that, in the wild, spends the initial stage of its development in freshwater lakes, rivers and streams. The fish grow from a yolk-filled egg through juvenile and parr stages before travelling downstream towards the sea as smolts. Atlantic salmon remain in seawater for 3 to 8 years during which time they grow and mature. The final stage in their natural lifecycle is to return to fresh water where energy is diverted to gonad development and for reproduction.

The culture of Atlantic salmon largely follows this process, with tight controls applied to the growth stages, except fish are harvested before gonad development and reproduction takes place. The initial production of eggs and juveniles is conducted in land-based production facilities where day length, temperature and feeding are controlled to maximise growth and development. When the fish are sufficiently large they are either retained in land-based systems or transferred as parr to open cages in lakes where they remain until smoltification. In Scotland approximately 50% of the fish under-going smoltification remains in land-based systems, with the remainder produced in open cages on lochs (Stagg and Allan, 2004).

Smoltification is the natural physiological process during which juvenile salmon adapt from a purely fresh water existence to cope with the marine environment. Under culture conditions this typically occurs at age 1^+ or 2^+ depending upon the growth of the fish in response to the feeding strategy used in the first summer of life and subsequent preferential feeding (becomes an S1 fish) or non-feeding (S2) (Metcalfe *et al*, 1992). Increasingly, developments in hatching and initial growth through the manipulation of light and temperature regimes is allowing smoltification to occur earlier and for fish to be delivered to on-growing facilities during the first year, called $S^{½}$'s or $S^{¾}$'s. Following smoltification fish are transferred to open cages in the marine environment for an on-growing period of up to 24 months.

In 2001 total Atlantic salmon production was 1.033 million tonnes (Fishbase, 2004)) and has continued to grow year on year since 1980 when only 4,778 tonnes were produced (Folsom *et al*, 1992). Farm production on any scale began in and continues to be dominated by Norway, accounting for 436,000 tonnes (Fishbase, 2004), or 42.2% of current global production in 2001. Chile and Scotland are the next largest producers and along with Canada, USA, Ireland and the Faeroe Islands account for a further 56.1% of production, with a total of 19 countries now producing Atlantic salmon on a commercial basis (Fishbase, 2004).

1.2.3 Atlantic salmon production in Scotland.

In Scotland production of Atlantic salmon has increased from just under 600 tonnes in 1980 (Folsom *et al* 1992) to 173,373 tonnes in 2003 (Stagg and Allan, 2004) (Figure 1.1) and currently accounts for 50% (by value) of all Scottish food exports (Scottish Executive, 2003). However, on a world-wide scale Scotland's contribution to overall production has remained level at between 13 and 18% (Fishbase, 2004).

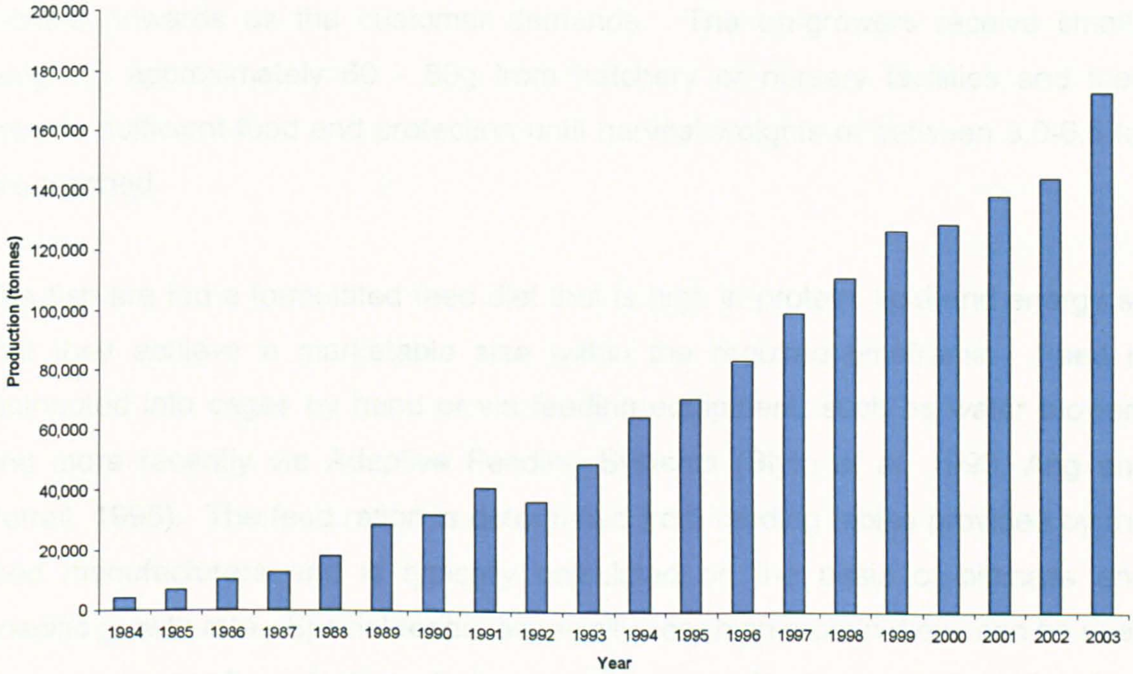


Figure 1.1: Production of Atlantic salmon in Scotland 1984 to 2003. (Fishbase, 2004; Stagg and Allan, 2004).

1.3 Marine cage culture of Atlantic salmon

1.3.1 The process of on-growing

The culture of salmon uses sea-based floating cages for the final stage of production (termed "on-growing"), a process that normally takes 18 – 24 months, although grading and harvesting can and does occur from approximately 12 months onwards as the customer demands. The on-growers receive smolts weighing approximately 60 - 80g from hatchery or nursery facilities and then provide sufficient food and protection until harvest weights of between 3.0-6.5 kg are reached.

The fish are fed a formulated feed diet that is high in protein, lipid and energy so that they achieve a marketable size within the required timeframe. Feed is distributed into cages by hand or via feeding equipment, such as water blowers and more recently via Adaptive Feeding Systems (Blyth *et al*, 1993; Ang and Petrell, 1998). The feed ration is determined from feeding tables provided by the feed manufacturers and is typically calculated on the basis of biomass and specific growth rate. Special feeds, especially very high protein diets, can be used to aid recovery after infection. Salmon are known to feed on specific pellet sizes and sizes are increased as the fish grow (Bailey *et al*, 2003).

Harvesting is carried out using techniques that minimise damage to the fish so they are acceptable to the market, which typically involves a blow to the head and an incision to the aorta. The early harvest of a proportion of the fish i.e. at a lower weight, is normal practice and not only provides some return on investment, but also allows stocks to be thinned out as the fish increase in size. Such management of stock and stocking density is important on fish welfare grounds (Farm Animal Welfare Council, 1996; Turnbull *et al*, 2005).

During this on-growing phase in production the health of the stock is paramount and chemotherapeutants are used to control bacterial and viral infections and sea lice infestations (Elema *et al*, 1996; Roy *et al*, 2000; Stone *et al*, 2002).

1.3.2 Site Selection and Environmental Criteria

Fish farming requires considerable investment in infrastructure, equipment and stock and the decision to invest in a particular site requires careful consideration. In Scotland the ownership of the seabed is vested in the Crown Estates which leases space to the fish farming industry, provided certain criteria are met. Under current legislation (See Henderson and Ross, 2000) all farms over 100 tonnes of production require an environmental impact assessment of the proposed site, which assesses suitability. This will typically involve a consultation with the local population, studies of water quality, hydrography, benthic sampling and assessment of the likely effects of siting the cages and buildings. Waste outputs from farms are considered an industrial waste that requires Consent to Discharge from the Scottish Environment Protection Agency (SEPA), who also issue licences to use chemotherapeutants and oversees the statutory monitoring required at farms (SEPA, 2001). A recent consultation paper issued by the Scottish Executive (2004) means site selection will also require planning approval for all future farms, although existing farms are likely to gain automatic approval,

Important site selection criteria include climate, hydrography and hydrodynamics. Wave exposure, wind and storms have been known to cause cage and equipment failure through the shear stresses they apply. The feeding of fish can also be disrupted if the cages are not accessible throughout the year, such as during periods of high wave activity and storms. Atlantic salmon production requires clean water with high visibility and high oxygen saturation, with currents and tidal changes being instrumental factors. Fast currents and tidal flushing act to dilute and remove dissolved wastes from cages (Doglioli *et al*, 2004) and to disperse particulate waste (Perez *et al*, 2002), allowing good water and sediment quality to be maintained at the cage site. However, too strong a current will increase the energy requirements of the fish to maintain position so that feeding and growth may be affected. The movement of water through cages maintains high oxygen saturation, which is particularly important during feeding when the oxygen demand increases. Rosenthal *et al* (1996) recommend a minimum average current speed of 5 cm s^{-1} although in Scotland failure to achieve this value does not necessarily result in a potential site being rejected (IOA, unpublished data).

Fish growth is temperature dependent, with higher growth occurring at higher sea temperatures. Salmon grow optimally between 5°C and 14°C and experience stress outside of this range with temperatures of –0.7°C and 20°C being lethal (Rosenthal *et al*, 1996). Temperate waters therefore provide an ideal environment for salmon farming. In environments where water freezes, such as in some areas of Canada, cages are often designed so they can be moved to better locations during winter (Rosenthal *et al*, 1996). In Scotland the farming of Atlantic salmon is carried out in rural sea lochs and fjords, which provide relatively sheltered sites, with large volumes of clean, aerated seawater, low to moderate currents and sea temperatures that vary between 5°C and 12°C over the course of a year.

1.4 Cage culture equipment

1.4.1 Cages

The basic design of near-shore floating cage systems (floating collar, net, anchorage) have changed relatively little in the last 20 years although the materials used have improved durability. Cages consist of a floating collar, a framework with walkways and a flexible nylon mesh net suspended underneath (Beveridge, 2004). The entire system is linked together and anchored to the seabed in order to minimize cage movement in the prevailing currents. Early designs were made of wood but plastic, aluminium and steel are more robust and are now generally used. Cages are generally round, square or octagonal in shape and size are generally limited to approximately 22m diameter x 15m deep (circles) or 15m x 15m x 15m deep (square). Larger cages with a larger biomass may cause difficulties in treating fish with chemotherapeutants, for example, and nets have to be cleaned on a regular basis so ease of handling is important. Also an increased area makes it difficult to feed all fish evenly.

1.4.2 Feeding equipment

The single largest cost of production is formulated feed (Intrafish, 2005) that makes optimising its use critical to the success or financial failure of a business. Fish are efficient at converting feed to biomass and the distribution of a ration size over an even area is a vital component of achieving uniformly sized fish.

Many farms continue to feed by hand and rely on feed manufacturers recommendations of ration size based on biomass. A variation on this is to employ hand held blowers, which move feed from a hopper through a nozzle that is manually operated by the fish farmer. Simple mechanical demand feeders rely on the fish learning to activate a lever in order to receive a reward of food. They are successfully deployed but are only able to spread feed over a limited area, which can result in territorial behaviour and wide variations in fish size. The understanding of feeding behaviour has improved in recent years (Kadri *et al*, 1996) and technological improvements have allowed a more automated feeding regime.

Intelligent feeders are the latest technological advance, examples of which include catch-eye (Bjordell *et al*, 1993), the Aquasmart AQ1 (Blyth *et al*, 1993) and the AQ1 variant, CAS, for a centralized hopper (Akvasmart UK Limited, Inverness). These systems assess variation in appetite and feeding behaviour, by analyzing pellet wastage and altering the quantity of food added in real time, to meet the needs of the fish on any particular day (feeding to satiation). There are, however, few accounts in the literature of the actual levels of waste generated using this technology within salmon culture, although Huntingford (2001) found that feed waste and feed conversion ratio (FCR = ratio of weight of food added to biomass increase) was reduced at Sea Bass and Sea Bream farms.

1.5 Environmental impacts on the marine environment

1.5.1 General impacts on the marine environment

The marine environment is increasingly being affected by human development with consequential increases in pollution and exploitation. Population growth, coastal urbanisation and industrialisation have all polluted the sea to some degree. Oil spills (Baker, 1990; Jackson, 1996), domestic and industrial waste dumping (Pearson and Rosenberg, 1978; Strain *et al*, 1995) and agriculture run-off (refer to Enell and Lof, 1983) are particular cause for concern in the marine environment. Industrial fishing, whilst not polluting in itself, has removed many fish and some commercial stocks have or are nearly collapsed. Also, the extent to which fishing gear damages the seabed is only now becoming clear (de Groot and Kaiser, 2000). Expansion of the aquaculture industry has alleviated industrial fishing pressure to some extent by supplying an ever increasing market for fish products. It is also argued, however, that the pressure has increased due to increased requirements for fish meal and oil (Naylor *et al*, 2000). Expansion of the industry has also lead to an increased awareness and concern over the detrimental effects of intensive fish production on the environment.

1.5.2 Concern for the environment

Aquaculture has developed almost in parallel with an increase in public awareness of the impacts humans have on the natural environment. This is particularly true for industrialised nations but environmental improvement and sustainable development are also increasing societal objectives in developing countries (Boyd, 2003).

In 1992, 178 nations signed a treaty in Rio containing principles that aim to "eliminate unsustainable patterns of production and consumption" and to reduce pollution (Folke *et al*, 1994). Whilst the treaty does not explicitly refer to aquaculture, the principles of sustainability and the precautionary principle (Francis, 1996) are now embedded in the management practise for aquaculture and have lead to moratoria on farm development in, for example, Canada and Scotland in areas of conservation interest and environmental sensitivity.

Fish farming in Europe typically operates within specific national guidelines (Fernandes *et al*, 2000), without any Europe wide integration. In Scotland, environmental regulation is carried out by the SEPA, who set environmental quality objectives (EQO) and standards (EQS), under the auspices of the Environment Act 1995 as detailed in Henderson and Davies (2000). Thompson *et al* (1995) argued that there was no strategic assessment plan in Scotland and that development had been allowed to continue unchecked. However, new legislation introduced in 1999 requires an Environmental Impact Assessment to be carried out on proposed or modified farms with a maximum biomass level in excess of 100 tonnes (Henderson and Davies, 2000). As part of this process biomass limits, effluent discharge consents and limits for the use of chemicals and chemotherapeutants are agreed and subsequently reviewed under a regulated monitoring programme. More recently the Scottish Executive (2003) published its "Strategic Framework for Scottish Aquaculture" that outlines longer term plans to help coordinate the fast developing industry.

That aside, the extent to which aquaculture impinges upon the marine environment continues to be the subject of much debate. According to Pillay (1992), referring to the nutrient impact of fish farming, "In the global context of the environmental impact of human activities.........the contribution of aquaculture is undoubtedly small". However, Folke *et al* (1994) argue that any level of nutrient impact is unacceptable and that the cost of treating such waste should be added to the farmer's costs. Discussing Atlantic salmon farming in particular, Folke *et al* (1994) argue that because of this input, farming is unsustainable in its present form. They suggest that the costs associated with pollution should be borne by the fish culture industry under the "polluter-pays" principle. Folke *et al* (1994) equate the nutrient effluent produced from farms to person-equivalents, adding this cost to production and thus making the industry unprofitable. Soley *et al* (1994) provide a theoretical framework of how this polluter-pays principle might be incorporated.

The problems with this are 4 fold:

1) the composition of effluent nutrients from fish farms and from human habitation are quite different, will act differently in the marine environment and the thus the effects cannot be compared,
2) present technological development and prohibitive costs make the elimination of waste *per se* impractical (Perez *et al*, 2001),
3) a high proportion of waste is in dissolved form and not collectable except in enclosed cages but these are not currently available commercially,
4) a tax levy in itself will not eliminate waste output and will not improve the environment.

More generally, Asche *et al* (1998) suggest that to some degree such costs have been internalised with improvements in feed formulation, better husbandry and technological advances that have reduced environmental effects over the last 10 years. Also, regular monitoring and remediation in terms of fallowing or biomass consent reduction have acted to inform the fish farmer and have added to the better management of the industry.

1.6 Pollution and aquaculture

There is difficulty in defining what is meant by pollution because words such as "harm" are vague and the use of "substances" ignores energy and other inputs. Acceptable levels of pollution are often defined by what is measurable and take no account of cultural differences (Farmer, 1997). The Department of Environment (DoE) Sustainable Development Strategy defines pollution as "a substance which is present in concentrations which cause harm or exceed an environmental standard" (Farmer, 1997). The fact those environmental standards may be set because of detection limits, however, means that the setting of environmental standards often take no account of potential sub-lethal effects at lower concentrations (e.g. Medina *et al*, 2002).

Perhaps the most comprehensive definition of pollution comes from the EU directive on Integrated Pollution Prevention and Control that states, "Pollution shall mean the direct or indirect introduction as a result of human activity, of substances, vibration, heat or noise into the air, water or land which may be harmful to human health or the quality of the environment, result in damage to material property, or impair or interfere with amenities and legitimate uses of the environment" (Farmer, 1997). But even this comprehensive definition fails to include social impacts such as visual disturbance or the general dislike of fish farming in some quarters.

In this wider sense of pollution, the aquaculture industry has come under intense criticism (for example see Miller and Aiken, 1996; Payne, 1999). In its most obvious form, pollution from the fish farming industry results from nutrient releases of waste feed, faeces and metabolic wastes (section 1.6.2), changes to physio-chemical processes in sediments (1.6.3) and changes to benthic populations (1.6.4). However, many conflicts have arisen not from this aspect but from what can generally be called non-nutrient pollution, discussed briefly below.

1.6.1 Non-nutrient pollution from the culture of Atlantic salmon

The main non-nutrient "pollutants" from fish farming and the causes of the majority of the conflicts surrounding fish farming (Payne, 1999), stem from sea lice infestations and the use of chemotherapeutants used to combat them, and fish losses (escapees), although visual effects and the use of transgenic fish may play an increasing role in years to come.

1.6.1.1 Chemotherapeutant use

Of prime concern to the fish farmer is the health of their stock that requires an ever changing list of chemicals and chemotherapeutants to combat disease and parasitic infections (Weston, 1996). Malachite green, for example, is a bath treatment used for many years to treat parasite and fungal infection but was banned from use in 2002 because it is a known carcinogen. There has been a general trend of reduction in the quantity of chemicals used over the past decade (Taylor *et al*, 1998) as husbandry has improved, as treatments have become more effective and development of in-feed treatments (Rae, 1979) mean chemical products can be used at low concentrations. Vaccination against Furunculosis (Midtlyng *et al*, 1996) and other fish diseases affecting caged salmon, prior to distribution to on-growing facilities, has led a reduction in antibiotic use, for example.

Typical chemotherapeutants used include sulphonomides, tetracyclines, quinolines and pesticides such as dichlorvos and cypermethrin (Beveridge, 2004; Rosenthal *et al*, 1996[a]). Many of these products are known to persist in the environment, however, especially following accumulation in sediment, (Hoy *et al*, 1990). There is also the possibility that use may lead to bacterial resistance and disruption of the sediment breakdown processes by bacteria (Stoffregen *et al*, 1996). New products continue to be developed. One recent pesticide, emamectin benzoate (also called SLICE™), is now licensed and thus far has proved successful by not persisting in the environment and not affecting important polychaete growth under cages (Costelloe *et al*, 1998), these polychaetes being vital for bioturbation of the sediment.

1.6.1.2 Sea lice

Sea lice (*Lepeophtheirus salmonis*) and *Caligus* sp. infestations are thought to be fairly common on fish farms, partly because of the high stocking densities used and the ease with which the sea lice can find further hosts. They not only damage fish but create lesions that are liable to secondary infection. Sea lice from fish farms, specifically, have also been blamed for the collapse of the sea trout (Payne, 1999) and wild salmon fisheries, with calls for the industry to take action to reduce sea lice at farm sites.

Hydrogen peroxide and organic pesticides are commonly used to control sea lice infestations, being applied to the fish externally via bathing in a solution of the chemical. More recently, products which affect the nervous system of lice and arthropod growth inhibitors that inhibit the ability of the sea lice to produce their chitin skeletons, have entered the market (Taylor *et al*, 1998). While the industry and others are calling for the use of these new chemotherapeutants to effectively combat the problem, they all have the ability to affect non-target species of arthropods and crustacea and there are equal calls for the use of such chemicals to be limited (Edwards, 1997). In Scotland chemotherapeutant products and their use is controlled by the Fish Health Inspectorate through Consent to Discharge given by SEPA.

1.6.1.3 Escaped fish

Fish are likely to escape from fish farms through poor husbandry, net damage and occasional catastrophic losses during storms (Beveridge, 1996). There are three concerns related to escapees, 1) escaped salmon may compete for spawning space and displace natural populations (Sægov *et al*, 1998); 2) mating and hybridisation may occur with wild stocks, leading to changes in genetic make-up that could provide inadequate attributes for long term survival (Peterson, 1993, cited in Rosenthal *et al*, 1996[b]; Brodeur and Busby, 1998; Milner and Evans, 2003) and 3) that translocated fish species may out-compete native species for food resources and space (McKinell and Thomson, 1997). Escaped diseased fish may also be vectors for pathogenic viral and bacterial diseases (Windsor and Hutchinson, 1995). Recent news events (Briggs, 2003) suggest that numbers of escapees are on the increase and a recent long term analysis of data has also documented explicit detrimental effects on wild stocks (McGinnity *et al*, 2003).

It is of note that transgenic salmon, that have been genetically modified in some way, particularly by combining farmed fish genes with growth genes from other non-farmed species, are not yet grown on a commercial scale. Research facilities do exist and then only in on-shore sites where accidental escape is virtually impossible.

1.6.2 Nutrient waste from the culture of Atlantic salmon

The intensive cage culture of fish species generates significant amounts of dissolved and particulate waste material, such as waste feed and the outputs of metabolic processes. Cage culture is an open production system so waste enters the sea directly. Such inputs may have either an actual or potential impact on the water column and sediment. Nutrient losses reported in the literature concentrate on carbon, nitrogen and phosphorous (Enell and Lof, 1983; Gowen and Bradbury, 1987; Brattan, 1990; Strain *et al*, 1995; Costa-Pierce, 1996) because of both the importance of these elements in metabolism and the deleterious environmental effects that occur when excess nutrients are experienced.

Until recently, mineral losses (except phosphorous) have almost been ignored, perhaps due to the conservative nature of these elements and the fact that fish gain their requirement mostly through drinking seawater (Lovell, 1998). However, techniques for assessing the lipid composition of sediments have recently been developed in order to evaluate sediment recovery (Henderson *et al*, 1997; McGhie *et al*, 2000).

It is difficult to comprehensively quantify the extent to which fish farm waste components have a deleterious effect upon the marine environment. It often depends on factors that are outside the control of the fish farmer. Water temperature, natural stratification processes, current speed, tidal flushing, sediment type and the nutrient assimilation potential of the water column and sediment will all affect the degree of impact, depending on the level of nutrient input.

1.6.2.1 Dissolved nutrient waste

The primary sources of dissolved nutrients released from cages are nitrogenous compounds in the form of ammonia (NH_3), urea, trimethylamine, creatine and creatinine (Bergheim and Asaud, 1996). In the marine environment nitrogen is considered the limiting factor for growth of microscopic phytoplanktonic organisms at the base of the food-chain. However, high levels of freshwater input into sea lochs may lower the salinity such that phosphorous may become a limiting factor.

It has been estimated that 60-90% of the nitrogenous compounds that originate from fish farms are released as NH_3 via the gills (Costa-Pierce, 1996). NH_3 is produced in the liver following the catabolic metabolism of amino acids (Lovell, 1998), transferred as the ammonium ion (NH_4^+) in the blood and excreted alongside chloride ions during the process of osmoregulation (Bone *et al*, 1996). At the gills NH_4^+ is dissociated to NH_3 + H^+ and the NH_3 is excreted (Bone *et al*, 1996). Upon entering seawater, the ammonia again takes up H^+ ions to produce NH_4^+, although this is a two-way process depending upon temperature and pH (Boyd, 1995).

The major risks associated with dissolved nutrient waste in aquatic environments are those of hypernutrification, phytoplanktonic growth and eventually eutrophication (Aure and Stigbrandt, 1990; Persson, 1991; Silvert, 1992; Talbot and Hole, 1994; Gowen, 1994). Hypernutrification, or the measured increase in nutrients, is not in itself a negative environmental impact. Indeed excess nutrients can stimulate phytoplankton growth that feeds both zooplankters and fish (Sarvala, 1993).

On a global scale, changes in climate and hydrography are deemed to be the most important factors affecting phytoplankton growth (Dale and Nordberg, 1993). On a local scale, the level of nutrients available for growth or specific nutrient limitations will have a proportionately greater effect on regional phytoplankton biomass. There seems to be no clear data that show reaching a certain nutrient load will elicit phytoplankton growth or that salmon aquaculture *per se* has caused nutrient loading within a water body to be increased. Reports that directly link aquaculture and eutrophication are based on data from freshwater farms (Persson, 1991) where the geophysical and chemical consequences are different from marine areas. However, the increased use of coastal regions for aquaculture is 1 of 4 reasons given to explain recent increases in harmful algal blooms (Hallegraeff, 1995). According to Hallegraeff (1995) the effects of hypernutrification and eutrophication in the aquatic environment should not be ignored and the aquaculture industry should recognise that "the likely outcome of an increase in nutrient load will be an increase in phytoplankton blooms" and eutrophication effects.

It has been estimated that 1 μmol-N dm^{-3} is an acceptable level of nitrogen loading (Silvert and Sowles, 1996) for a marine water body, although 3 μmol dm^{-3} are also believed not to cause eutrophication (Turrell and Munro, 1989). Tidal flushing, wind flushing and freshwater flow are also important factors influencing concentration (Silvert and Sowles, 1996; Panchang *et al*, 1997) and thus the influence of aquaculture on nitrogen loading will vary from place to place. As a consequence, it is therefore likely that assimilative capacities and levels of acceptable dissolved nutrient input from salmon culture will tend to be specific to a particular location or body of water.

1.6.2.2. Particulate nutrient deposition and dispersal

Particulate deposition under and around marine cage systems occurs for various reasons. Fish produce waste, the open system of production allows feed pellets to escape without being eaten and current technology does not allow the capture and disposal of waste products. It is also known, for example, that water currents are altered by the influence of cages and nets, slowing the speed by up to 60% (Inoue, 1972; Black, unpub. data). This increases the deposition of naturally occurring particulate material, such as phytoplankton, where particles that would otherwise have remained in suspension in the water column are able to settle out onto the seabed.

The amount of waste leaving a fish farm is not insignificant. At a Feed Conversion Ratio (FCR, where FCR = the ratio of fish wet weight gain to amount of dry food fed) of 1.2, one tonne of production requires 1.2 tonnes of feed over the growing cycle leading to 6 kg per tonne of nutrient rich feed reaching the seabed if a 5% feed waste is assumed. With modern day farm production ranging from 100 to

3000 tonnes per farm there is a significant potential to cause an impact in the sediment. Add to this the faecal material produced and various workers have suggested that between 205kg and 2500kg of solid waste is produced per tonne of fish production (Cho, 1991; Enell and Ackerfors, 1994, respectively).

The extent to which the sediment is impacted by particulate waste material from the fish farm stems from a combination of factors. It is not only dependant on fish biomass and the quantity of food and faecal material added, but also on sediment grain size (eg Lumb, 1989), water content, feed formulation, husbandry, assimilative capacity and season. An important factor is the area over which the waste is spread.

Gowen *et al* (1989) modelled the spatial distribution of waste as a function of current speed, water depth and the settling velocity of particles. This initial model has been refined (Silvert 1992, Gowen *et al*, 1994) and altered to take into account variation in settling velocity (Silver and Sowles, 1994; Chen *et al* 1999; Wong and Piedrahita, 2000), bathymetry (Hevia *et al*, 1996) and hydrodynamic features such as turbulence and varying current speed with depth (Silvert and Sowles, 1994). If the depth is sufficiently shallow then re-suspension and re-deposition by storms is likely (Dudley *et al*, 2000). Chen *et al* (1999) have also shown that a current speed greater than 4 cms^{-1}, as may arise during storm driven mixing, results in sedimented particles being moved by saltation.

Verification of such models is an important part of their validation and studies have shown a general agreement between modelled and measured inputs (Gowen *et al*, 1994). However, few models (Cromey *et al*, 2002; Telfer, 1995) include variations over time (season), such as variations in feed intake and fish growth both of which vary with sea temperature.

Findlay and Watling (1997, and references cited therein) have identified 8 effects of waste deposition on the seabed beneath and surrounding cage culture sites and these are discussed below under 2 broad headings 1) chemical flux, sediment chemistry changes and bacterial processes, and 2) changes in fauna.

1.6.3 Chemical flux, sediment chemistry changes and microbial processes

The majority of ocean sediments receive low levels of carbon deposition (2 x 10^{-5} gC m^{-2} d^{-1} ~0.2% organic carbon) but those areas designated as continental shelf receive 3 gC m^{-2} d^{-1} (2%) due to higher phytoplankton growth and shallower depths (Berner, 1982, cited in Cranston, 1994). The deposition of particulate material is therefore a naturally occurring process. The breakdown of this settled material is driven by bacteria and microbes. In these unpolluted sediments bacterial and other metabolic processes utilise oxygen as the terminal electron acceptor in the Krebs cycle during the function of aerobic respiration and the production of energy. In sediment this typically occurs down to a depth of 7mm depending on sediment type (Blackburn, 1978). Oxygen penetration is deeper in coarser sediments containing large interstitial spaces that allow water to flow through and is shallower in finer sediments where grains are more tightly packed and interstitial space is limited.

These aerobic processes are replaced by nitrate reduction and dissimilatory iron or manganese reduction (Davies *et al* 1996) deeper in the sediment and through the redox discontinuity layer (RDL) until anaerobic sediment is reached centimetres to meters below the surface (Blackburn, 1978). In the aerobic layer the bacterial growth that drives these processes is lower when the C:N ratio is high and increases as the C:N ratio is lowered and N is preferentially mineralised (Boyd, 1995). Sloth *et al* (1995) have shown that NO_3^- dominates the nitrogen efflux in sediment with low organic content. Below the RDL, anoxic sediments dominate where sulphate from pore water is reduced and replaced with ammonium, occurring down to a depth where methanogenesis takes over. Both sulphate reduction and methanogenic processes do not require free oxygen, gaining the oxygen components from oxidised compounds such as CO_2 instead (Boyd, 1995).

In sediments under fish cages the concentration of organic matter is increased over normal levels due to the sedimentation of waste feed and faeces. Increased organic content raises the biological demand for oxygen in the sediment, which can quickly become depleted if the flux of oxygen saturated water into sediment is insufficient to replace it (Findlay and Watling, 1997).

Oxygen flux rates into sediment have been measured at rates of 6-8 times higher at fish farms than would normally be experienced at sites without farms (Hargrave *et al*, 1993). This increased oxygen demand and faster rate of use has the effect of compressing the boundaries of the various processes, described above, up towards the sediment surface (Davies *et al*, 1996), often to the extent where H_2S and CH_4 can bubble into the water column (Gowen and Bradbury, 1987; Weston, 1990; Black *et al*, 1996). These sediment processes are fundamental to understanding the environmental impacts of cage farming because they affect the rate of assimilation in the sediment, dictate faunal changes, can result in damage to fish stocks and will dictate the recovery time after fish farming has ceased.

Increases in organic matter can also result in cessation of nitrification/denitrification processes in the sediment (Kaspar, 1988; Sloth *et al*, 1995; McGraig *et al*, 1999). This in turn can increase the amount of ammonium in pore water (Cranston 1994; Sloth *et al*, 1995), which can percolate into the water column and increase the aqueous BOD.

In sediments H_2S formed through sulphate reduction initially combines with ferrous ions to produce characteristic black sediments, but thereafter is liable to escape into the water column. H_2S is toxic to the benthos and fish since it inhibits the action of cytochrome C oxidase (Black *et al*, 1996[a,b]) at concentrations of 500-1000 ppm (Raas and Liltrerd, 1992). Using juvenile Atlantic salmon Black *et al* (1996[a]) showed that low level H_2S will not necessarily damage gill tissue sufficiently to inhibit growth, although 22-29 μmol l^{-1} can cause permanent gill damage. There does not seem to be any evidence of the effects on larger fish that may be better able to cope with this level of H_2S exposure. Holmer and Kristensen (1992) showed that under fish farms 99% of the sulphate reduction occurs in the top 40 mm of sediment during the summer, coincident with increasing sea temperatures, at rates of 5 – 8 μmol cm^{-2} d^{-1}. These figures are in the same order of magnitude identified by Hargrave *et al* (1993). A high concentration of H_2S during the summer is likely to diffuse from the sediment as gas bubbles. However, the extent to which fish are damaged or are caused stress by this is partly dependent on water depth. Gas bubbles that are released are devoid of H_2S within 9 – 12m from the sediment (Black *et al* 1996[a] and references cited therein), so when fish cages are located in depths shallower than this, accumulation in surface waters and cages is possible (Lumb, 1989).

1.6.4 Faunal changes to the seabed

The seabed provides a complex habitat for fauna living in and on the sediment. The inter-relationship between biological factors, such as competition for space and resources; physical factors such as grain size; and chemical factors such as those described above, vary but in general act to provide an equilibrium state. Additional stresses, such as increased nutrient loading, act to alter the natural balance of physical and chemical factors in particular, that have a corresponding effect on the biology.

Typically, the seabed is inhabited by a range of phyla; Annelida, Mollusca and Echinodermata amongst them. Under environmentally stable conditions no one species dominates and many are long-lived. However, when a stress is applied, such as the nutrient deposition from fish farms, short-lived more tolerant species (also called opportunistic species) adapt to the prevailing conditions faster because of their ability to rapidly reproduce and colonize an area and overall species diversity and abundance is reduced. Pearson and Rosenberg (1978) investigated a typical scenario, of nutrient enrichment from the outflow of a sulphite pulp mill. Generally, near-source species numbers were reduced to a few opportunistic species at high abundance, followed by a gradation of increasing diversity and lower dominance as the distance from the disturbance is increased and nutrient loads reduce to background levels.

Faunal changes under and around fish farms, resulting from nutrient enrichment, are reasonably well documented (Gowen and Bradbury, 1987; Brown *et al*, 1987; Weston, 1990; Kraufvelin *et al*, 2001; Kempf *et al*, 2002). In Scotland the volume of data gathered each year far outweighs that published. For example, SEPA have required the collection of benthic samples for many years at each fish farm with companies required to submit an annual report. However, these data remain out of public hands and no detailed analysis of this comprehensive dataset has been published. Monitoring remains fundamental to ensuring the farming industry does not excessively damage the marine environment.

1.7 Feed, feeding behaviour and feeding technology

Sustainable development (including aquaculture) implies the long-term viability of an enterprise that meets the needs of the present without compromising the ability of future generations to meet their needs (Farmer, 1997), whatever they may be. Thus, in an ideal situation the cage culture of fish, such as Atlantic salmon, would have no impact upon the marine environment. However, it has already been shown that as an open production system intensive culture of Atlantic salmon results in large quantities of waste entering the marine environment.

Whilst practitioners continue to assess the fate and impact of wastes from salmon farms, it is equally important to reduce the amount that is produced through better feed formulation, an improved understanding of feeding behaviour and advances in feeding technology. This is being achieved by the various means discussed in this section.

It should be noted that developments in feed formulation, an improved understanding of feeding behaviour and advances in feeding technology stem primarily from the drive to reduce costs and increase efficiencies within the industry. Despite this, a reduction in the amount of waste has resulted from these activities (and better husbandry) and in light of recent initiatives (EU, 2002) all future developments in Europe, such as new feeds, should have to consider environmental impacts as part of their development.

1.7.1 Feed

Atlantic salmon production uses an intensive monoculture system that relies entirely on addition of formulated feed. In the 25 years since salmon production began on any large scale there has been a great improvement in our understanding of the nutritional requirements of these fish (Hardy, 1998). In the early 80's few papers existed on feed formulation for Atlantic salmon (Hellend *et al*, 1991), but now minimum requirements have been established for the majority of the essential amino acids, vitamins, minerals and lipid contents needed for effective growth (see Lovell, 1998).

Salmon diets typically contain 45 – 50% protein and fish are adept at using it as an energy source, with 60-90% of ingested nitrogen re-released as NH_3 following catabolism and osmoregulatory processes (Hall *et al*, 1992). The major protein source is fish meal, although sustainable alternatives to this continue to be investigated in order to reduce the reliance on already over-targeted wild fish stocks (Carter and Hauler, 2000; Opstvedt *et al*, 2003; Mundheim *et al*, 2004). Lipid is the other main energy source in salmonids so that raising the relative proportion of lipid increases available energy for growth (Solberg, 2004). Salmon have an essential requirement for n-3 and n-6 fatty acids (Tocher *et al*, 2001), with 18:3n-3 and 18:2n-6 being important in determining the optimal tissue ratio of longer-chain fatty acids, such as 20:5n-3 and 20:4n-6 (Sargent *et al*, 1999; Ruyter *et al*, 2000; Bransden *et al*, 2003). Fatty acids are thought to be important in biochemical and physiological functions, such as cell membrane permeability and as precursors to other biological components (Sargent *et al*, 1999). This requirement can be met from fish oils, although the composition varies depending on the fish species used in feed manufacture (Johnsen *et al*, 2000) and from other feedstuffs, such as sunflower (Bransden *et al*, 2003), linseed and rapeseed oils (Bell *et al*, 2003). Carbohydrate is typically provided using grains and although pelletisation by extrusion methods makes the starch more digestible, carbohydrate is poorly digested by salmonids (Storbakken *et al*, 1998).

The availability of nutrients to metabolism and growth is a function of the amount of the nutrients in the feed, the quantity of feed added per kilogram of production and their digestibility (Talbot and Hole, 1994). The degree to which component feedstuffs are digestible, the Apparent Digestibility Coefficient (ADC) (Cho and Kaushik, 1990; Cho, 1991; Cho and Bureau, 1997), varies depending upon the feedstuff used and the chemical structure of the components. Ingredients such as fat, for example, have an ADC of 85-95% (that is 85-95% of the fats (e.g. fish oil) will be absorbed, Hillestad *et al*, 1999), so waste from this feedstuff will be low. According to Hillestad *et al* (1999) ADC for protein, fat and carbohydrate are generally accepted as 87%, 90% and 65% respectively, thus the amount of the food absorbed, and hence the level of faecal waste will vary depending on the combination of ingredients. Cho and Bureau (1997) suggest the amount of faecal waste equals the feed consumed (dry weight) x {1 – ADC}. Increasing carbohydrate in the diet could be a means of reducing the use of fishmeal, for example, but this also results in increased faecal output because of carbohydrate's lower digestibility. ADC is dependant on both temperature (Azevedo *et al*, 1998) and digestible energy levels (Azevedo *et al*, 2004) and to a large extent depends on optimising the nutrient balance.

Einen *et al* (1995) and Lovell (1998) suggest that there is an interaction between feed allowance and optimum dietary nutrition. Fish not fed to satiation for each nutrient requirement, will eat more to compensate for deficiencies in that diet. For example, some amino acids are unstable during heat treatment in the manufacturing process (Booth *et al*, 2000) but deficiencies can be made up by eating more food. This is unsatisfactory, as eating more will affect feed conversion ratio (FCR), growth (Talbot and Hole, 1994; Einen *et al*, 1995; Morris *et al*, 2003) and faecal output.

Meeting specific nutrient requirements is not the only factor affecting salmon growth. Provision of sufficient energy is also thought to be a major factor (Paspatis and Boujard, 1996). Fish are generally able to compensate for low energy budgets in the same way they do for nutrient deficiencies, by eating more food (Cho and Bureau, 1995) but again this increases waste as already discussed. It has been calculated that the minimum energy requirement is 15 MJ of digestible energy (DE) per Kg of feed with 22-24 g of digestible protein per MJ DE (Cho and Woodward, 1989; Cho, 1992). Importantly, reducing protein and replacing with lipid can increase energy available, whilst having the effect of reducing nitrogenous waste (Talbot and Hole, 1994; Sveier *et al*, 1999).

More generally, in feed development there is a need to:

1) Optimise the nutritional balance of protein, carbohydrate, lipid, minerals and vitamins to provide the minimum required for maximal growth.
2) Increase the energy content of the feeds, especially through the use of lipids, that up to a limit provide a protein sparing effect (Sveier *et al*, 1999)
3) Decrease the concentration of indigestible components in the diet.
4) Increase the digestibility of components through careful selection of ingredients and processing technology (see Booth *et al*, 2000).

(After Talbot and Hole, 1994)

Improved feed formulation has resulted in reduction in waste output but in practical terms the efficiency of the production practice is measured by FCR. FCR is described by the relationship between the specific growth rate and ration size, (Talbot and Hole, 1994; Einen *et al*, 1995). Both growth and FCR are affected by abiotic factors such as temperature, oxygen concentration, body weight and stress. For example, the FCR will be lower in smaller fish (Hemre *et al*, 1995) but increases with increased fish size (Brett, 1979, cited in Nordgarden *et al*, 2003), presumably as the energetic requirements of larger fish change and specific growth rate is reduced (Alsted *et al*, 1995).

The industry average FCR is presently 1.1-1.3, with the minimum achievable calculable through physiological energetics (Thorpe and Cho, 1995). An FCR of 0.8-1.0 is already achieved under hatchery and tank conditions (e.g. Opstvedt *et al*, 2003) and was thought to be an achievable target under normal farming conditions (Austreng, 1994, cited in Einen *et al*, 1995), but has yet to materialize in practice. Implicit in this ratio of the amount of dry feed used to wet weight gain are direct feed losses, i.e. that part of the food added to the cage but never consumed by the fish, although the quantity of food lost is affected by both biotic and abiotic factors rather than ration size and growth *per se* (Talbot and Hole, 1994).

As has already been established, wastes from aquaculture operations contain a dissolved component, a solid (faecal) component and a waste feed component. Nutritional strategies are currently designed to minimise the output from the first two of these. Whilst feeding the correct ration, based on energy and nutrient requirements to maximise growth and FCR, indirectly reduces the amount of feed waste, further improvements can be gained by an understanding of feeding behaviour and feeding strategy and these are discussed in the following section.

1.7.2 Feeding behaviour and feeding strategy

Atlantic salmon is a naturally active predatory animal near or at the top of the food chain. As such they rely on sight as their primary sense in the capture of food (Stradmeyer, 1992; Talbot *et al*, 1995; Ang and Petrell, 1998). It is this feature that makes them prime candidates for culture in cages, having to capture food in the water-column as they would do naturally. Huntingford and Thorpe (1992) note that most cultured species are, in evolutionary terms, only a few generations removed from their wild counterparts. This is significant in terms of fish behaviour and affects subsequent feeding rates, prey preferences and feeding rhythms.

Feeding has two main aims; 1) to encourage the rapid and positive uptake of food and thereby increase ingestion, minimise leaching of essential nutrients and reducing waste and 2) to minimise the metabolic activity of feeding and thus increase energy available for growth (DeSilva and Anderson, 1995). Further, waste is reduced if the correct ration size is fed at times of the day that elicit the most positive response.

Food intake is governed in the first instance by stimulation of appetite from metabolic and neurological feedback and hormonal control (DeSilva and Anderson, 1995). It is recognised that at any stage the food pellet might be rejected and if this occurs after food has been grasped then the rejection can damage the pellet and it may end up as waste. Studies have shown that 1-40% of feed ends up as waste, although 5 - 15% are more typical values, with these lower values used in feeding studies and waste assessments (Blyth *et al*, 1993; Findlay and Watling, 1994; Beveridge *et al*, 1997; Cho and Bureau, 1997). Considering that one million tonnes of feed was produced in Europe in 2000 (Intrafish, 2000) even these lower figures represent considerable potential losses. Understanding feeding behaviour and having a feeding strategy that maximises growth and minimises waste are key to maintaining profitability and environmental sustainability.

An important part of a feeding strategy is to produce fish at harvest that are of a uniform size. Studies have shown that feeding at restricted spatial and temporal patterns increases aggressive and territorial behaviour (Olla *et al*, 1992; Noakes and Grant, 1992; Kadri *et al*, 1996) which results in a skewed growth distribution and may also result in the smaller fish not feeding even when sufficient food is available. This is one of the problems associated with feeding a restricted ration in a large-scale production facility as proposed by Cho and Bureau (1998). Small scale studies in tanks may allow both secondary feeding and feeding outside established periods in the day (Jorgensen and Jobling, 1992) but this is not feasible for open production systems where water currents and the settling velocity of feed (Chen *et al*, 1999) will dictate the time between the food pellet becoming available and it leaving the cage as waste.

Salmon take 15-25 minutes to achieve satiation, feeding initially at 0.3-0.5 kg t fish^{-1} min^{-1} (Talbot *et al*, 1999). Talbot *et al*, (1999) showed that feeding patterns change over short timescales, with initial surface feeding followed by feeding in deeper water at a reduced rate as the stomach becomes fuller. Various studies have been carried out to establish feeding patterns on a daily basis (Thorpe *et al*, 1990; Kadri *et al*, 1991, 1997; Jorgensen and Jobling, 1992; Talbot *et al*, 1999) and over longer periods (Blyth *et al*, 1993, 1999; Thomassen and Fjaera, 1996).

Broadly speaking Atlantic salmon are crepuscular feeders, with peaks shortly after sunrise and again before dark. This represents an evolutionary compromise between metabolic demands, vision capacity, predation risk and food availability (Eriksson and Alanara, 1992). In fish culture, feeding has to also coincide with the practicalities of operating staff. Boyard and Leatherland (1992) suggested that feeding regimes might be affected by this practical restriction in feeding time but other researchers have shown this is not the case. In longer term studies Blyth *et al*, (1999) showed that feeding was controlled by the light/dark cycle and temperature changes and not by specific feed delivery restrictions. As a consequence of reduced day length and the sea temperature decreasing in winter, feeding rate, feed consumption and growth are all reduced.

During feeding it is also important to assess satiation so that excess feed is not added. Many farms feed by hand or use a simple blower system, feeding the quantities recommended by the feed manufacturers, where satiation is assessed by observations of fish behaviour from the surface by the farmer. Appetite is affected by pollution, stress, water temperature and oxygen concentration (DeSilva and Anderson, 1995) and they act to complicate the observations. Ang and Petrell (1998) suggest that visual observation is an insufficient mechanism to assess when feeding should be stopped, partly because water clarity can affect the decision thus making it a subjective decision.

1.7.3 Feeding Technology

Increasingly, technological developments have enabled either direct sub-surface assessment of satiation (ie cessation of feeding) using video cameras (Foster *et al*, 1995; Ang and Petrell, 1997) or indirect assessment by hydro-acoustic detection (Juell, 1991; Juell *et al*, 1993) or by particle sensors (Blyth *et al*, 1993). For example, the Aquasmart AQ1 feeding system is programmed to automatically distribute a feed ration from a hopper and is calibrated with pellet size and sinking rate data. At a known depth below the surface, a detector system counts the number of pellets falling through the water column, the number used to assess whether feeding is continuing or has stopped. The loop is complete with feedback to the control unit, which finely adjusts subsequent feed output (see also Chapter 2). Importantly from a management point of view, the quantity of food delivered is computed and when used in conjunction with regular growth data can be used to calculate FCR on a regular basis. Ang and Petrell (1997) showed that sub-surface assessment of feeding activity and pellet detection, for judging satiation, reduces waste by improving feed conversion and growth. But in practice there has been insufficient study of the effect of these adaptive feeding systems on levels of waste output.

1.7.4 Feed Pellets

Adequate ration delivery that can be varied in time and space and the identification of both short and long term feeding strategies are not only important husbandry processes, but can also reduce waste. However, other factors such as pellet colour, texture, smell and hardness, size and shape will also influence the feeding behaviour (Mearns, 1985; Stradmeyer, 1992; Smith *et al*, 1995) and may affect the amount of wasted feed. Of these, pellet hardness and pellet size will have the largest influence on waste output.

Pellet hardness varies with different production processes (see Seymour and Bergheim, 1991) and between manufacturers (Chen *et al*, 1999). Subsequently, a pellet's stability in water will affect the leaching of nutrients and disintegration, potentially increasing waste levels.

As fish grow, mouth capacity increases and feed pellet size is increased accordingly. Salmon show a preference for pellets that are longer than they are wide and for a cylindrical shape. Large fish are unlikely to feed on smaller pellets because the increased energy required to capture many small pellets proves inefficient (Black, pers. Comm.). Manufacturing processes generally produce relatively uniform pellets but damage to material during the production process or subsequently through transportation and storage, or by the feed delivery system is likely to be lost as waste.

Improvements in feed formulation, feeding technology and our understanding of feeding behaviour and preferences have reduced the levels of waste entering and impacting the marine environment. Of great interest at the moment is what further improvements can be made, especially from the use of feeding technology.

1.8 Aim and Objectives of this thesis

Conflicting with the socio-economic importance of fish farming to the Scottish rural economy is the potential for intensive aquaculture to damage the environment. It has been shown, above, that developments in feed formulation and management practice have reduced the levels of waste being generated, but that in the main such changes have been driven by economics rather than particular concern for the environment.

The feeding of fish continues to be dominated by the hand feeding method with the quantity added determined from feeding tables and experience. However, new feeding technology is now available, one example of which is the Akvasmart UK CAS feeding system. Use of similar systems has been shown to reduce FCR and particulate waste at Sea Bream and Sea Bass farms (Huntingford, 2001). There is, however, little understanding of the environmental implications of utilizing this feeding system within salmon culture. The aim of this project, therefore, is to assess whether adaptive feeding systems confer any environmental benefit at salmon farms. Environmental benefit particularly refers to a reduction in waste particulates and an improvement in benthic habitat under and around fish cages. Environmental in this context does not include broader socio-economic or cost benefits of the system, which will be excluded from the analysis.

A series of studies will be conducted at or using data from two sites; Portavadie, where fish are fed using a CAS adaptive feeding system (Akvasmart UK Limited, Inverness) and Rubha Stillaig, where fish are fed by hand. Comparison between the different feeding regimes will be made using a physical, biological and modelling approach. Specific objectives and hypotheses are identified in respective chapters but general objectives are:

1) To deploy sediment traps at each site and a reference site to collect particulate material and quantify differences in the quantity and composition of deposited material and rate of sedimentation between sites, under the two feeding regimes (physical approach).

2) To collect sediment and benthic samples, using grabs, over a 2-year period that will allow quantification of species abundance and diversity at various distances from the cages and allow comparison of changes to the benthos under the two feeding regimes (biological approach).

3) To use a GIS-based particulate deposition model developed at the University of Stirling Institute of Aquaculture (IoA), plus feed and production data supplied by the fish farm company and sediment trap data collected during this study to compare the depositional area and sedimentation under each feeding regime (modelling approach).

Chapter 2

Site description and fish history

2.1 History of Portavadie

Portavadie is a small rural village located on the Cowal peninsula in Argyll on the west coast of Scotland (Figure 2.1 inset). A small collection of houses sits on the banks of Loch Fyne, at the open southern end, with views of the Kintyre Peninsula to the west and the Isle of Arran to the south. Locally there are few amenities with the nearest grocery shop situated in Tignnabruaich some 6 km distant and larger amenities located in Dunoon approximately 40 km by road. Transport links to the area consist of ferry access from Greenock to Dunoon and from Tarbert on the Kintyre Peninsula to Portavadie and by road, with many local roads being single track only.

The local economy derives the majority of its income from tourism during the summer months. However, the farming of sheep, logging and aquaculture also play an important role in maintaining the community. The whole area is surrounded by moorland and forest, owned and run by the Forestry Commission. Logs are transported both by road and by sea from the pier at Portavadie that was re-opened in October 2003 after refurbishment (Anon, 2004[a]). There has been a ferry running between Portavadie and Tarbert, on the Kintyre peninsula, since 1977 with hourly trips run by Caledonian Macbrayne during the summer months only (Anon, 2004[a]).

The largest facility in Portavadie is an unused concrete oil-rig platform production facility that was authorised for construction in 1975, by the then Scottish Secretary, on behalf of Sea Platform Constructors Limited (Scotland), at a cost of £4million (Kerr, 1975). A harbour and accommodation block was constructed but unfortunately no orders for production were forthcoming and the site has remained empty since its construction. The harbour currently provides anchorage for a few small boats but is generally rarely used.

2.2 History of Fish Farming at Portavadie

Portavadie and the nearby Rubha Stillaig fish farms are owned and run by Lighthouse of Scotland Limited (Cairndow, hereafter referred to as Lighthouse). It is a wholly-owned subsidiary of the Norwegian company Pan Fish Group, which is one of the biggest salmon producers in the world with fish farms in the USA, Canada, Norway, Japan and Scotland. Lighthouse is Scotland's third largest salmon farming company and owns 21 salmon-farming licences in Loch Fyne, with a total production capacity of approximately 10,000-12,000 tonnes. This corresponds to approximately 8 % of all salmon production in Scotland. Since 2002 Lighthouse has also owned and operated all of the farm sites previously owned by Highland Fish Farm Ltd sites in northern Scotland increasing their capacity by 50% (Panfish, 2004).

The open-water cage culture of Atlantic salmon at Portavadie fish farm commenced in 1984. The site had been owned by 3 companies prior to being purchased by Lighthouse in 1998. Biomass consent at the site has remained unchanged through this period at 300 tonnes. An application for biomass consent for Rubha Stillaig was granted to Lighthouse in 1998, with a biomass consent that has remained unchanged at 900 tonnes.

2.3 Site Description

The experimental site consisted of 2 farms, Portavadie and Rhuba Stillaig, both within the same embayment on the southern end of Loch Fyne on the western coast of Scotland. The two sites were approximately 1.2km apart (Figure 2.1) and a reference site was situated between the two farms in similar hydrographic and bathymetric regimes.

Portavadie consisted of 12-off 70m circumference (~ 22m dia.) Polar Circle cages in a block of 2 x 6. Relative to North the cages at Portavadie were orientated at 80° (see Figure 2.2). Rubha Stillaig was a slightly larger farm consisting of 20-off Polar Circle cages of the same size, in a block of 2 x 10 on an orientation of 30°

(see Figure 2.2). Each of the cages had a net depth of ~10m. Distances between the cage centres at both sites were 40m within a row (L) and 48m between rows (W) as shown in figure 2.2.

Water depth was measured at the start of the experimental period, using a 400KHz hand-held echo-sounder (Speedtech Instruments, USA) and was similar at each location, being 27m, 30m and 26m at Portavadie, Rubha Stillaig and the reference site respectively. Changes in tidal height were indicated by the hydrographic data collected at the site, shown in Chapter 3.

In addition to differences in farm size, the primary difference between the two sites was in the way the fish were fed (See 2.5 below). However, both sites were run by the same management and staff with subsequent husbandry and farming techniques applied uniformly between the two sites.

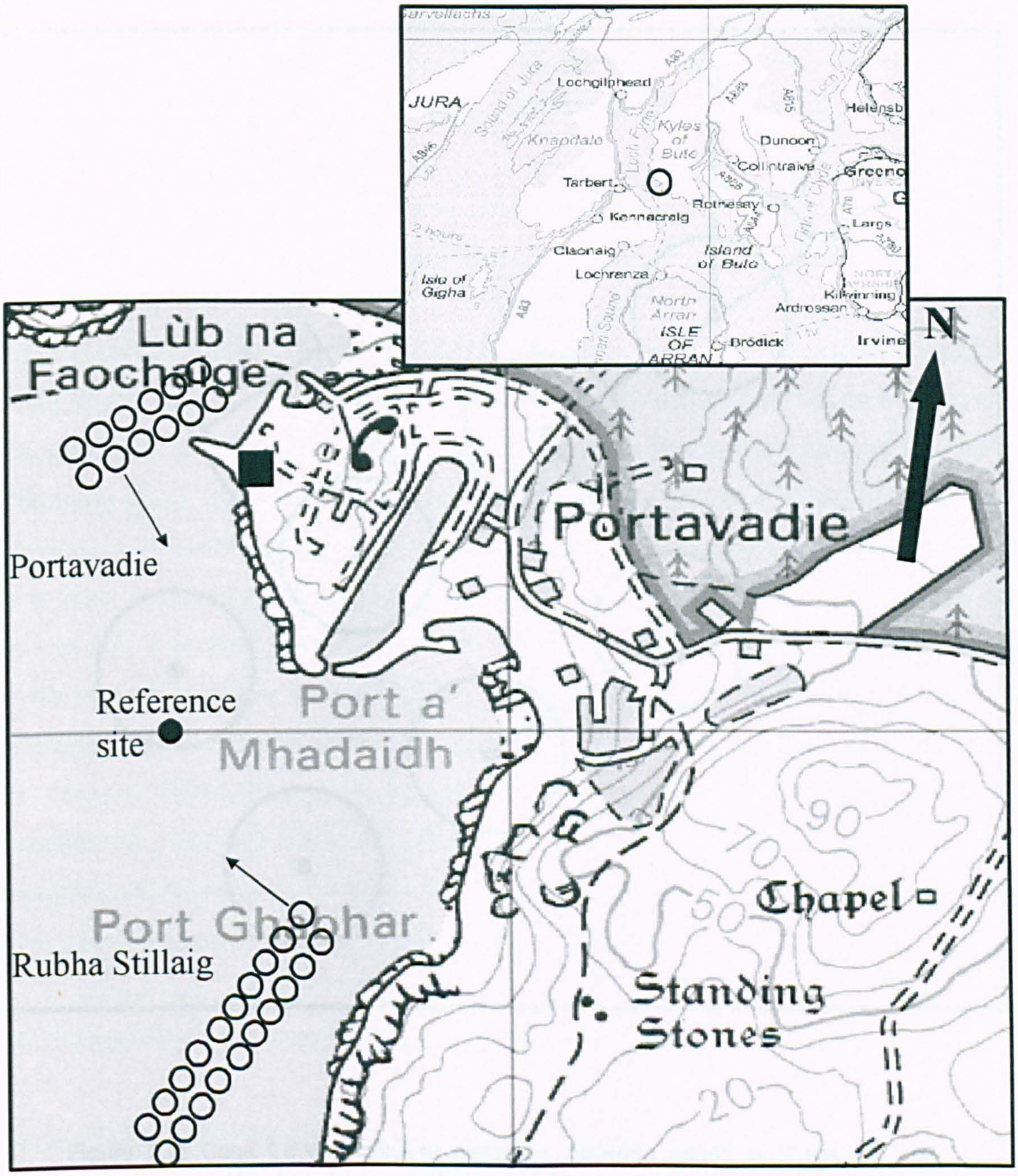


Figure 2.1 Layout and orientation of experimental sites. Arrows represent direction of transects for sediment trap studies and benthic grab collections at experimental cages. Arrows extend from cages identified as Cage 8 at Portavadie and cage 11 at Rubha Stillaig. ■ = location of site office, food store and CAS feeding unit. Inset shows site (circled) in relation to surrounding area. Maps from Ordnance Survey (2003).

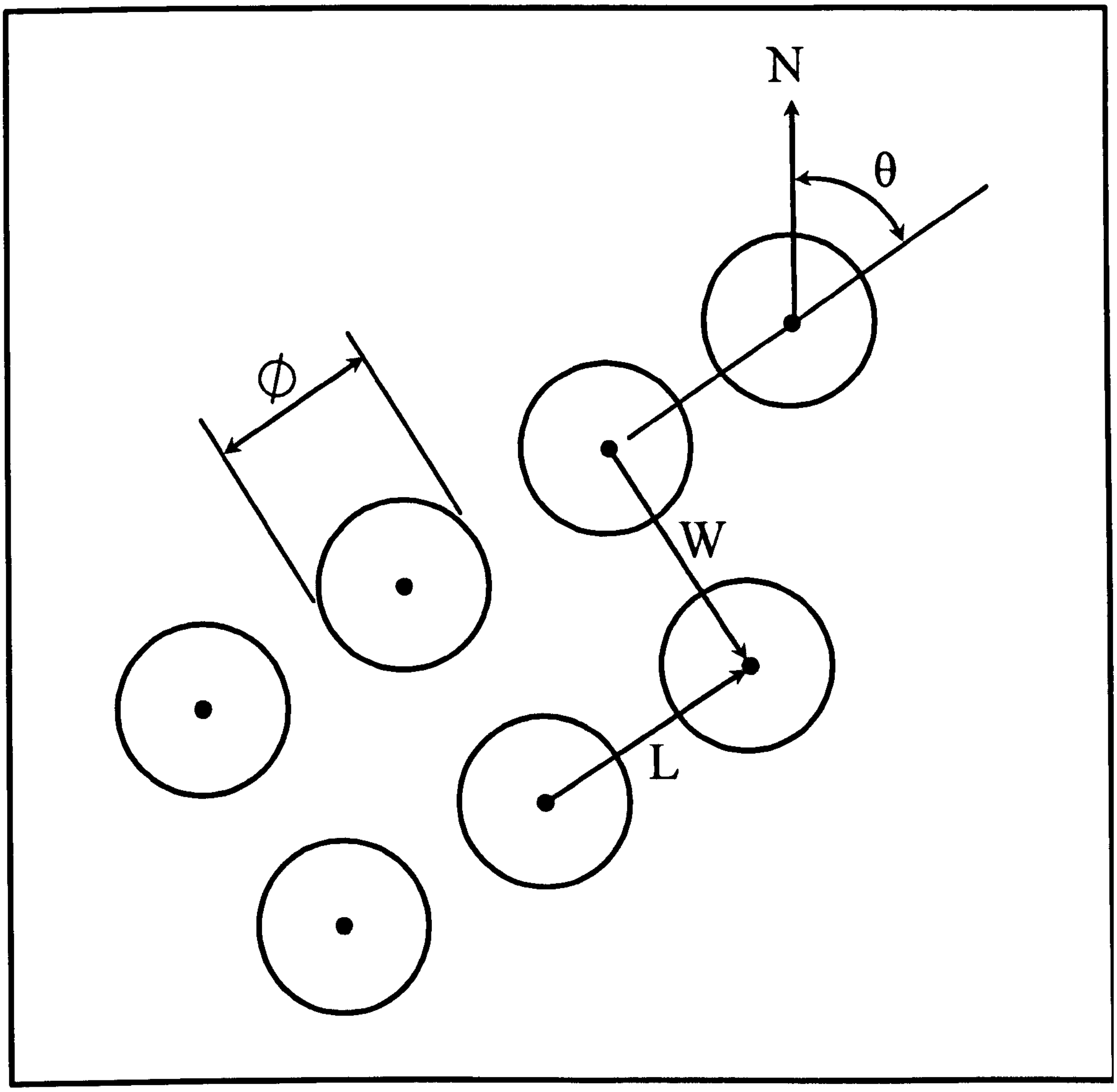


Figure 2.2: Cage Layout showing distances between cages in a row (L) and between rows (W). ∅ = diameter of cage. At Portavadie and Rubha Stillaig sites L = 40m, W = 48m and ∅ = 22m. θ = cage orientation from north in degrees. θ = 80° and 30° at Portavadie and Rubha Stillaig respectively. Not all cages shown.

2.4 History and movement of fish for present study

Atlantic salmon smolts arrived at the Portavadie site as S1's in June 2001. The overall biomass and stocking density at the site and subsequent moves to Rubha Stillaig remain confidential, although information on the specific experimental cages is reported below.

2.4.1 Experimental cage 8 at Portavadie

Cage 8 was stocked with 47,520 Atlantic salmon smolts with an average weight of 164g on arrival. The cage was specifically double stocked until the fish were large enough to be split between 2 cages. The fish were counted and graded on the 9th February 2002 and the stock was split. Of the original quantity 28,063 were transferred to Rubha Stillaig and took no further part in the experiments. 10,491 fish were retained in cage 8 at Portavadie with 8,966 mortalities between first arrival and the stock split. The 10,491 fish that remained at Portavadie had an average weight at the time of 2.85kg. The fish remained at Portavadie until 10th July 2002 when they were temporarily moved to Rubha Stillaig and made ready for harvest approximately 1 month later. A further 177 mortalities occurred between 9th February 2002 and 10th August 2002 with 10, 685 fish harvested at an average gutted weight of 5.6kg. The increased number of fish harvested resulted from minor fish movement between cages at the site.

2.4.2 Experimental cage 11 at Rubha Stillaig

Cage 11 at Rubha Stillaig was created from double stocked fish kept in cage 11 at Portavadie. Cage 11 at Portavadie was counted and graded on 11th December 2001 and 18,900 fish at an average weight of 1.9kg were moved to Rubha Stillaig to become the experimental cage at this site. Prior to this no fish had been grown at Rubha Stillaig for 12 months. The site had been fallowed for the period 27th December 2000 to 11th December 2001 but cage infrastructure (cages and buoy grid, but not nets) remained moored at the site. Cage 11 was graded on 4/5th February 2002 and again on 30th May 2002 when 6,814 fish were removed and transferred to another site. The remaining fish at Rubha Stillaig were harvested on 5th November 2002 and again as a result of minor fish movements at the site, 17,597 were harvested at an average weight of 4.7kg.

2.5. Feeding at the sites

The primary difference between the experimental sites was the method used for feeding the fish. A CAS adaptive feeding system was used at Portavadie and hand feeding was employed at Rubha Stillaig.

2.5.1 CAS Adaptive Feeding System – at Portavadie

The CAS (Centralized Adaptive System) (figure 2.3) system is a derivative of the AQ1 technology originally developed by Blyth *et al* (1993) and supplied by Akvasmart UK Ltd (Inverness, Scotland) for a centralized hopper system.

The feeding system is programmed to automatically distribute a feed ration from a hopper based on the shore and is calibrated with pellet size and sinking rate data. At a known depth below the surface, a detector system counts the number of pellets falling through the water column with the number used to assess whether feeding should be continued, at either an increased or decreased rate, or stopped. The loop is complete with feedback to the control unit, which finely adjusts subsequent feed output. Full technical information is available from Akvasmart UK Limited, Inverness, but briefly the system consists of the following:

- The CAS system components include a Windows based software package located on a personal computer (PC) held in an office facility on site, a centralised feeding unit based on shore at Portavadie and a CAS-1 control unit mounted on the cage. The key equipment also includes a sensor, cone (3m diameter) and cable located in the centre of the feed spread at an appropriate depth (~5m) within the cage and radio transmission between the CAS-1 unit and the PC.

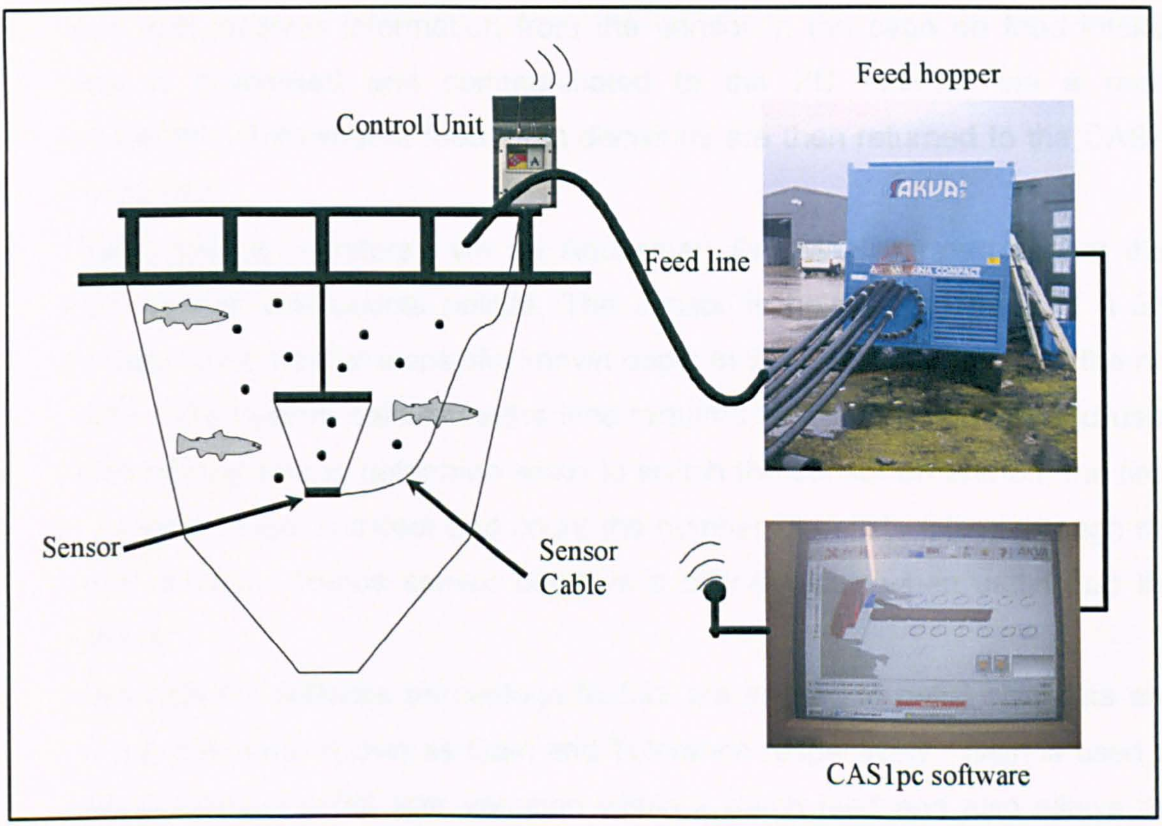


Figure 2.3: Schematic of Centralized Adaptive System (CAS) feeding system.

- The Windows based CAS software is used to remotely monitor and control the quantity of feed added and rate of feeding. Specifically this is done in real time but also all data is stored for historical monitoring.
- The CAS-1 control unit contains software in an EPROM, which controls feed input and receives information from the sensor in the cage on feed intake. Data is processed and communicated to the PC receiver via a radio transmitter. Subsequent feed input decisions are then returned to the CAS-1 control unit.
- Feed intake is monitored via an Aquasmart PACMAN infrared sensor that discriminates and counts pellets. The sensor is held at the base of a 3m diameter cone held at a specific known depth in the water, typically half the net depth. The system calculates the time required for dispensing feed and uses pellet sinking rate to determine when to switch the sensor on and off, the time in between used to detect and count the number of pellets falling through the water column. Hence sensor depth is a critical factor when setting up the system.
- Within the PC software percentage factors are applied to pellet size data and sinking rate data, known as Gain and Tolerance respectively. Gain is used to take account of pellet size variation within a batch feed and also allows the pellet sensor to distinguish between a pellet sinking past the sensor and other particulates such as plankton, algae and faeces which should not be counted. Pellets of slightly different size and composition will sink through the water column at a slightly different speed and a further value, called Tolerance, is also applied to allow for this variation.
- Finally, the operator inputs meal settings into the PC. Initial data includes meal duration via a start and finish time. Feed activity data controls the rate at which feed is delivered to the fish. Generally feed is added at the minimum rate initially and the rate increased to a maximum rate. Pellet count is monitored against pre-determined lower and upper threshold settings within the PC. When a pellet count below the lower threshold is detected, feeding rate is increased in stages until the maximum rate is reached. Feeding continues until a pellet count higher than the upper threshold is detected.

Then either the feeding rate is reduced until the pellet count is reduced or if the pellet count remains high the system sleeps and feeding is stopped for a pre-determined period. The days feeding is halted at satiation, determined by the pellet count remaining high or the maximum feed input being reached. The maximum feed input is determined from feed tables.

2.5.2 Hand feeding – Rubha Stillaig

Hand feeding remains the commonest form of feeding on fish farms in Scotland. The term "hand feeding" is somewhat expanded from the notion of using a scoop to throw feed into a cage. Hand feeding also includes the use of air or water blowers, with feed distributed from a hopper under the manual control of the fish farmer.

At Rubha Stillaig fish were fed via an air-based blower from a hopper aboard a feeding vessel. Blower feeding was deemed to be an advanced form of hand feeding as the determining measure of satiation and decision to stop feeding was through visual observation by the fish farmer.

2.5.3 Feeding Data

Information critical to an analysis of sedimentation rate and for mass balance calculations in a Waste Dispersion Model was supplied by Lighthouse of Scotland Limited. Specifically this included quantity of feed added on a daily basis through the sediment trap deployment periods (Table 2.1). The farm manager completed a monitoring form (Appendix 1) each day that comprised feed size, feed quantity, sea temperature, and general weather conditions. Comments could also be added to explain anomalies in feeding, including reasons for non-feeding. The farm manager gained his information for Portavadie via the CAS PC software and for Rubha Stillaig by notification from the fish farm workers.

Table 2.1: Daily feed input (Kg) to cage 8 at Portavadie and cage 11 at Rubha Stillaig for each day of the sediment trap trials. Zeros represent non-feeding due to prevailing weather conditions.

Day	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Date															
Portavadie															
16 - 31 August 2001	260	282	262	253	288	144	363	355	248	319	315	329	285	459	74
14 - 28 February 2002	254	255	256	257	258	252	263	256	0	265	267	245	271	223	137.5
16 - 30 April 2002	245	246	248	175	229	189	209	217	193	164	186	246	259	215	130.5
Rubha Stillaig															
14 - 28 February 2002	440	340	280	280	0	350	0	250	0	0	290	280	300	320	150
15 - 30 April 2002	190	300	270	350	390	370	340	380	380	380	390	0	350	340	0
3 - 18 September 2002	450	500	250	630	425	550	450	700	675	750	550	650	650	550	500

Chapter 3

General materials and methods

3.1 General introduction

This chapter describes methodology that is commonly used or those methods that would have been described in more than one of the forthcoming chapters. Remaining methods are described in the appropriate chapter.

3.2 Sampling locations

Samples for the benthic study (Chapter 4) and the sediment trap study (Chapter 5) were collected relative to specific cages identified as cage 8 at Portavadie and cage 11 at Rubha Stillaig. The position of the two cages outer edge was fixed using a handheld Global Positioning System (GPS) (Lowrance Globalmap™ 100 12-Channel Receiver). At Portavadie the position was (WGS84) 55° 52' 33.6"N, 5° 19' 21.9"W (OSGB36; NR 192168E, 669747N) and at Rubha Stillaig (WGS84) 55° 51' 53.4"N, 5° 18' 5.94"W (OSGB36; 192500E, 668487N) with 8 and 9 satellites detected respectively. In addition the reference site was at position (WGS84) 55° 52' 18.0"N, 5° 19' 451"W (OSGB36; NR 182023E, 669111N) with 8 satellites detected

3.3 Carbon and nitrogen analysis

Collection of specific samples is described in the appropriate chapter. Dried samples were crushed to a powder, using an agate pestle and mortar, to create a homogenous mix and stored in sample containers in a dessicator until analysis.

Total carbon and total nitrogen were analysed using a standard combustion method on a Perkin Elmer 2400 Series II CHNS/O auto-analyser with integrated AD-4 auto-microbalance (Perkin Elmer Corp., Norwalk, USA). Triplicate samples weighing between 4 and 6mg were used as homogeneity in samples containing feed and faeces is often difficult to achieve and the triplicate samples confirmed the precision of the method. Means were then used for further analysis. Samples were weighed out into pressed ultra-clean tin capsules 6 x 4mm (Elemental Microanalysis Limited) that had been tared to zero prior to the sample being added. After downloading the weight of the sample into the auto-analyser the tin capsules were sealed by folding and placed into the auto-sampler. Carbon and nitrogen were determined simultaneously, with up to 120 samples per day run automatically from the auto-sampler.

3.4 Particle Size Analysis

The method of collecting samples and collection locations for analysis of particle size is described in chapter 5. Once collected, samples were stored deep frozen. After defrosting the whole sample was dried in an oven at 90°C and stored in separate sample containers. Analysis for particle sizing was conducted in 2 stages, wet sieving and dry sieving.

This particle size method uses an aqueous solution of sodium hexametaphosphate $(NaPO_3)_6$ to prevent clumping and concretion of the fine particles of sediment. The solution was made using 6.2g of crystalline $(NaPO_3)_6$ dissolved in one litre of water; warmed under the hot tap to ensure all the $(NaPO_3)_6$ had dissolved.

Approximately 25g of dried sample was weighed accurately on a 4 decimal place (dp) analytical balance (Mettler AJ100, Mettler-Toledo Ltd, Leicester, UK) and was placed into a 500ml glass beaker, to which 10ml aqueous sodium hexametaphosphate and 250ml of tap water was added. The contents of the beaker were stirred using a glass rod for 6 minutes and then allowed to stand over night. After 24 hours the sample was re-stirred for a further 6 minutes before being washed through a 63µm sieve. Washing consisted of puddling the sieve in a white tray until the water ran clear. After sieving both the sieve and sample were dried for 1 hour, or until completely dry, in an oven at 90°C. After drying and cooling at room temperature, the sample was gently brushed from the sieve into a plastic weighing pan and accurately re-weighed on a ±0.0001g analytical balance. The difference in weight was the mass of particles less than 63µm in size removed through wet sieving.

After re-weighing the sample was placed in a series of eight stacked sieves (2mm, 1mm, 500μm, 250μm, 180μm, 125μm, 90μm and 63μm) plus a base pan for dry sieving. The use of these specific sieve sizes ensured the sediment sample was divided according to the Wentworth Phi scale (Wentworth, 1922). Samples were placed onto an Analysette 3 SPARTAN pulverisette 0 automatic shaker (Fritsch, Oberstein, Germany) and shaken for 10 minutes at an amplitude of 1.5. The content of individual sieves was weighed in a tared plastic weighing pan on a ±0.0001g analytical balance. The contents of the base pan were included and added to the fraction removed through wet sieving to give the total weight of particles less than 63μm, identified in the Wentworth scale as a mixture of silt and clay. No further analysis of this fraction was required for this work. Data were converted to the Wentworth phi scale (Table 3.1) and the percentage of each size fraction was calculated. Grain size parameters (median grain size, quartile deviation and skewness) were estimated using a graphical approach (after Inman, 1962).

Table 3.1: Relationship between particle sizes and Wentworth phi units under the Wentworth classification of sediment.

Particle Size Range (mm)	Phi Units	Grade name
> 256	< -8.0	Boulder
256 to 64	-8.0 to -6.0	Cobble
64 to 4	-6.0 to -2.0	Pebble
4 to 2	-2.0 to -1.0	Granule
2 to 1	-1.0 to 0.0	Very course sand
1 to 0.5	0.0 to 1.0	Coarse sand
0.5 to 0.25	1.0 to 2.0	Medium sand
0.25 to 0.125	2.0 to 3.0	Fine sand
0.125 to 0.0625	3.0 to 4.0	Very fine sand
0.0625 to 0.0039	4.0 to 8.0	Silt
< 0.0039	> 8.0	Clay

3.5 Hydrographic data collection

3.5.1 Deployment and recovery

Valeport BFM106 recording current metres (Valeport, Dartmouth, Devon) were deployed on a u-shaped mooring. Meters were configured as either one meter held at the depth of the cage net or a combination of two meters, one 3m below the surface (at low spring tide water level) and the other 3m above the seabed, the decision being dependent upon the availability of current meters.

Meters were deployed not more than 100m from the experimental cages in positions outside of the cage block so that the cages and cage moorings did not interfere with normal current flows and deployment. Current speed and direction were measured and averaged over 60 seconds every 20 minutes throughout the tidal cycle.

Current meters were deployed coincident with sediment trap data collection (Chapter 5) with the meters placed into the water prior to sediment trap deployment and recovered after the last sediment trap was collected. Hydrographic data was therefore collected for 1 tidal cycle (15 days). The position of the current meter was determined using a hand-held Global Positioning System (GPS) (Lowrance Globalmap™ 100 12-Channel Receiver) at each deployment. Position was recorded as latitude and longitude and British National Grid Reference (BNG), along with the number of satellites. The accuracy of the GPS varies with the number of satellites present at the time of recording. A minimum of seven satellites are required to give an accuracy of approximately 10m.

3.5.2 Data format

Valeport BFM106 current meters are mechanical recording instruments that measure current speed via a calibrated impeller. Current speed is measured in m s^{-1} and current direction is measured in degrees from North. As a mechanical recorder of current speed, the impeller has a minimum start up speed of up to 2cm s^{-1} due to friction. This is normal and no account of this is taken in the calculations used. It is generally accepted that when readings of 0.000 m s^{-1} were recorded the actual speed would have been between 0.00 and 0.02 m s^{-1} (Telfer, Pers.comm.).

3.5.3 Current speed and direction

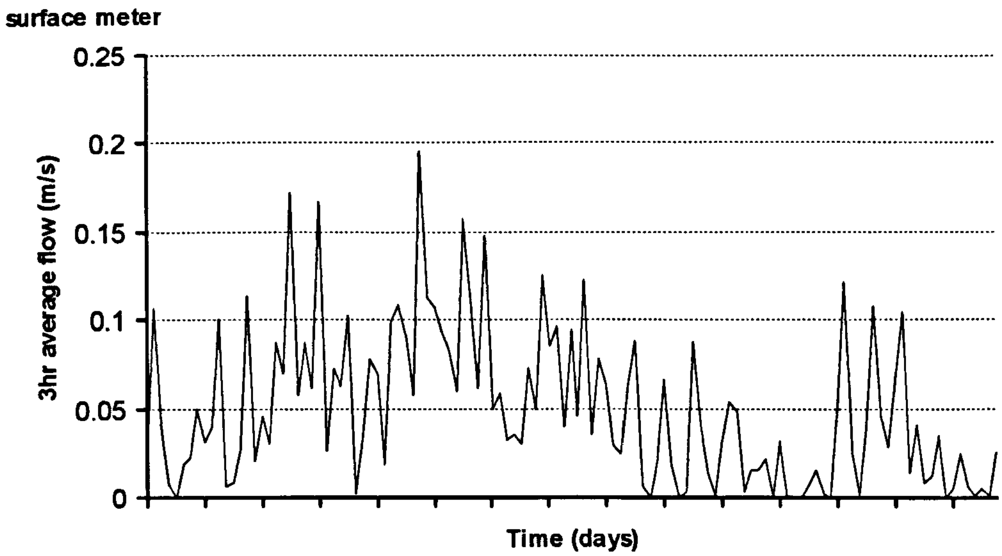
Of the three deployments of current meters made at the experimental site, collection numbers two (at Rubha Stillaig in February 2002) and three (reference site in April 2002) suffered from a failure. This was due to water ingress into the electronic system of the equipment, owned by the Institute of Aquaculture and used here, resulting in no data being recorded. Therefore, all analysis in this thesis is based on the first collection of data made in August 2001.

This first deployment of two current meters on a u-shaped mooring was sited within 100m of the Portavadie site but outside the influence of the cages themselves at (WGS84) 55° 52' 54.1" N and 5° 19' 39.8" W (OSGB36; NR 192120 E, 669781 N), with 7 satellites detected. Given the nature of the bay in which both experimental sites were located; that is similar water depth, surrounding hill structure and exposure to wind; it was assumed that this collection was representative of the current speed and direction of water movements for the entire bay. It was deemed that any differences between locations were unlikely to be sufficient to alter the overall effects of current speed and direction, to those recorded, on the subsequent data analysis.

The surface current meter was deployed at an average depth of 4.7m and varied between 2.7m and 6.5m over the spring/neap tidal cycle. The average depth of the seabed meter was 21.8m, varying between 19.8m and 23.7m. Minimum and maximum tidal range through the 15 day deployment was 1.5m and 3.9m respectively.

The 3-hour average current speeds are shown in Figure 3.1. The highest speeds occurred during the spring cycle between days 3 and 9. The average current speed during this period was 7.3cm s^{-1} and 5.0cm s^{-1} in the surface and bottom waters respectively. The highest recorded current speed was in surface waters at 23.6cm s^{-1} on day 6, coincident with the maximum tidal range. During the remaining (neap) tides current speed was reduced in both surface and seabed waters with averages of 2.9cm s^{-1} and 1.7cm s^{-1} respectively. Overall the mean current speed in surface and seabed waters was low to moderate at 4.7cm s^{-1} and 3.2cm s^{-1} respectively. Water at depths greater than 20m is not generally affected by wind induced currents and there were a high number of readings (52.4%) less then 3cm s^{-1} on the seabed compared to only 24.9% in surface waters, as shown in Figure 3.2. Speeds less than 3cm s^{-1} are regarded by SEPA as quiescent water (SEPA, 2002).

Current direction was similar at both depths, as shown in Figure 3.3, with a higher number of readings in a north-south direction than east-west. In surface waters the readings were skewed to the left of 0° and 180° showing that the overall direction of water movement was north-north-westerly to south-south-easterly and this was confirmed by the surface meter scatter plot of current speed and direction shown in Figure 3.4. The opposite was true for the deeper water where the direction NNE-SSW is more prevalent. Overall the residual currents would have a tendency to move particulate material to the north initially then SSE as the particles fell through the water column, as shown in Figure 3.5.



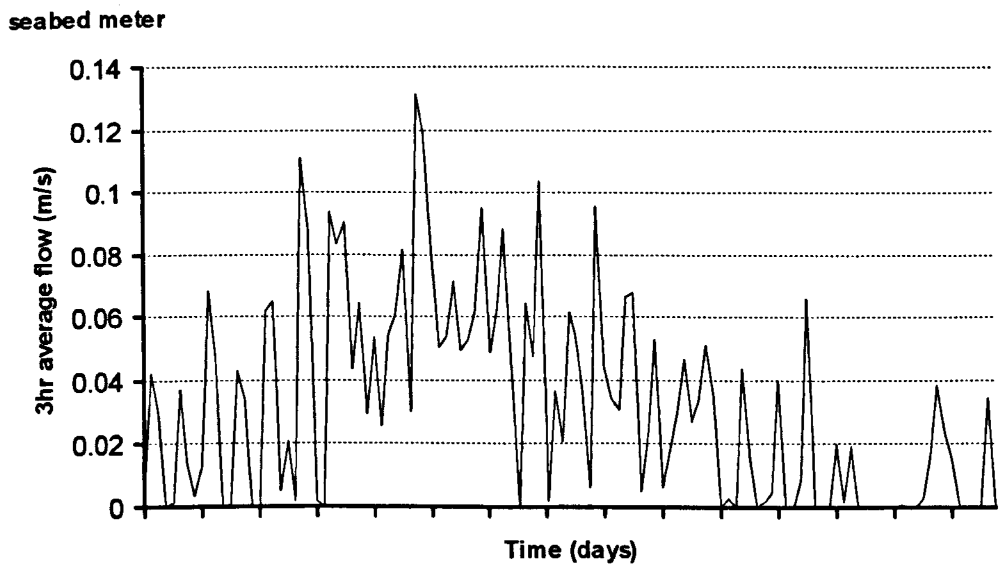
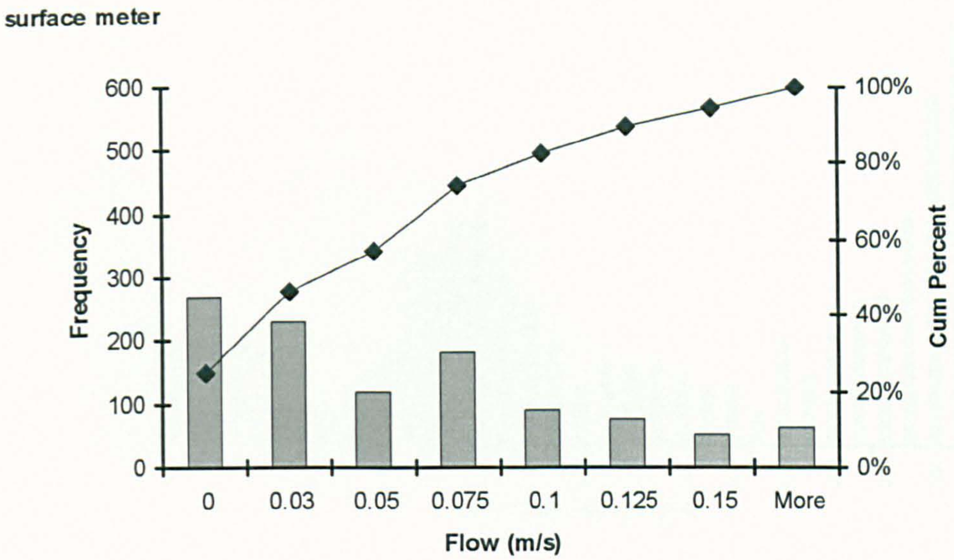


Figure 3.1: 3-hour average current speed measured at Portavadie on Loch Fyne in Scotland, over 1 tidal cycle (15 days) in August 2001, using a Valeport BFM106 direct recording current meter. Surface meter was at a mean depth of 4.7m, seabed meter mean depth was 21.8m. Note the different scales.



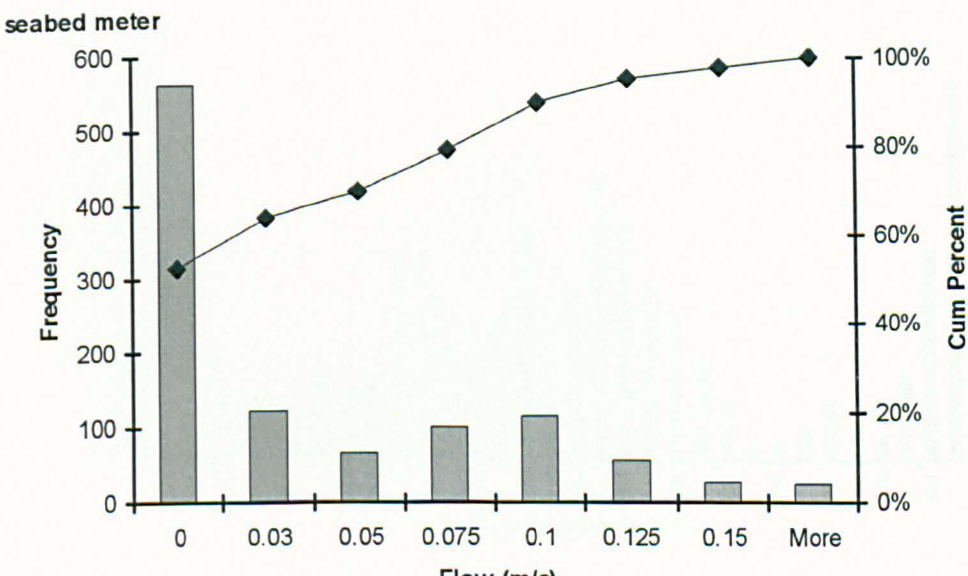
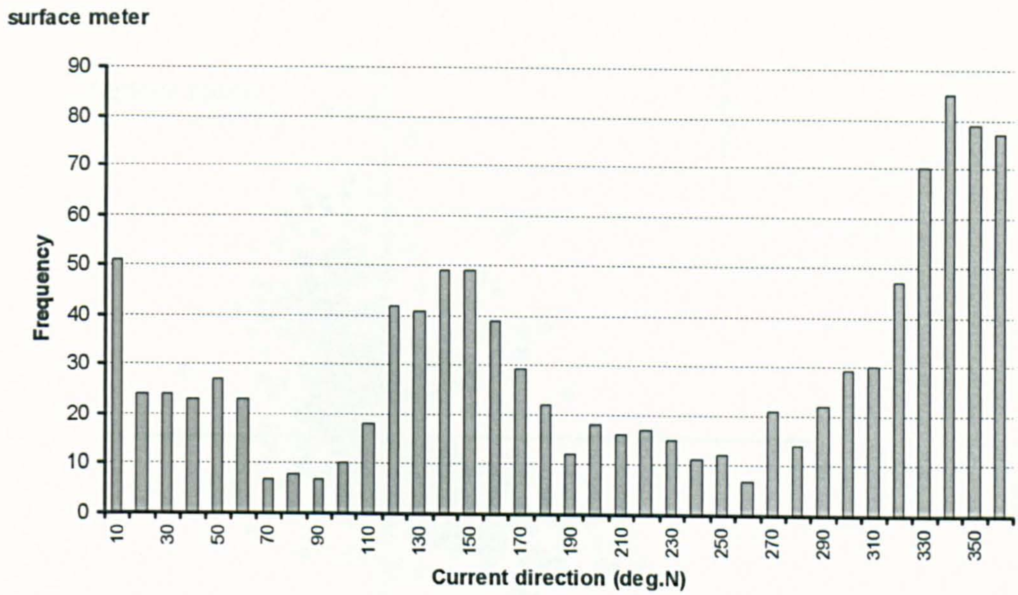


Figure 3.2: Frequency histogram and cumulative frequency of current speed measured at Portavadie on Loch Fyne in Scotland, over 1 tidal cycle (15 days) in August 2001, using a Valeport BFM106 direct recording current meter. Surface meter was at a mean depth of 4.7m, seabed meter mean depth 21.8m.



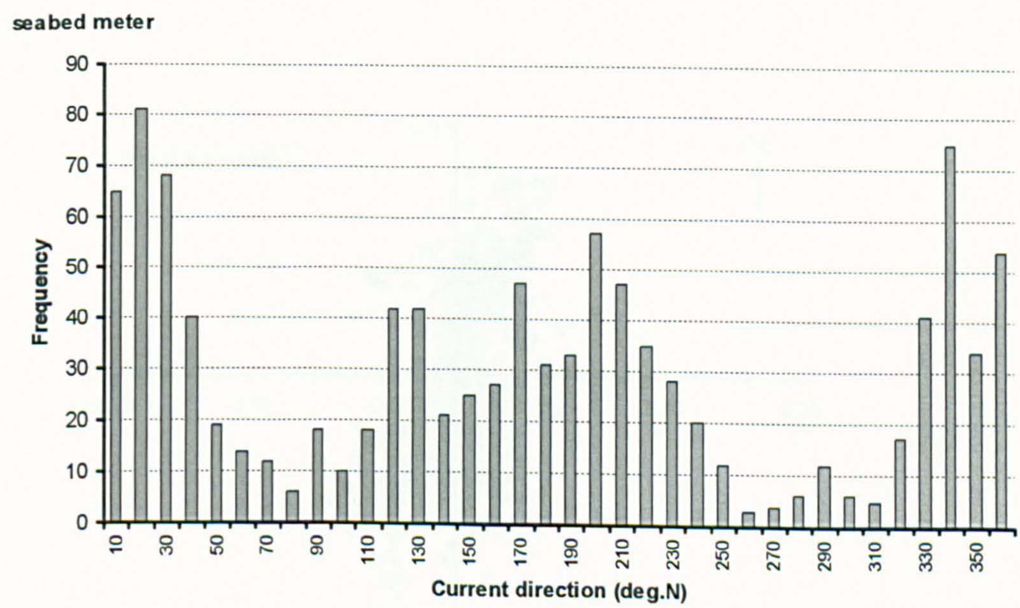
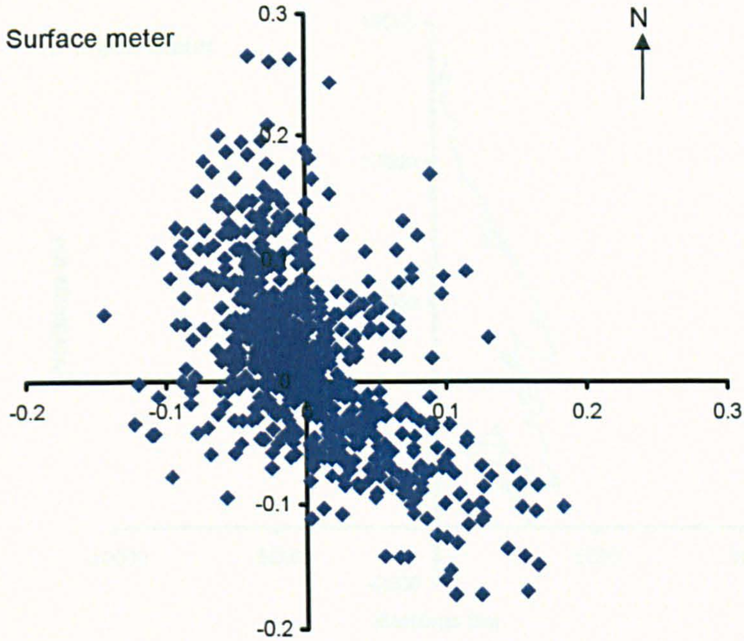


Figure 3.3: Frequency of direction of current flow at Portavadie on Loch Fyne in Scotland, over 1 tidal cycle (15 days) in August 2001, using a Valeport BFM106 direct recording current meter. Surface meter was at a mean depth of 4.7m, seabed meter mean depth 21.8m.



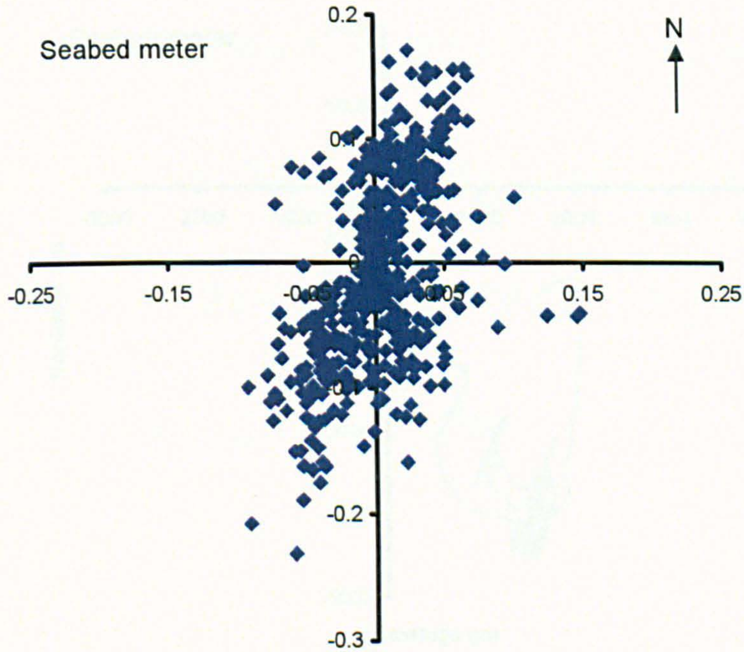


Figure 3.4: Scatter plot of current speed (ms^{-1}) and direction at Portavadie on Loch Fyne in Scotland, over 1 tidal cycle (15 days) in August 2001, using a Valeport BFM106 direct recording current meter. Surface meter was at a mean depth of 4.7m, seabed meter mean depth 21.8m.

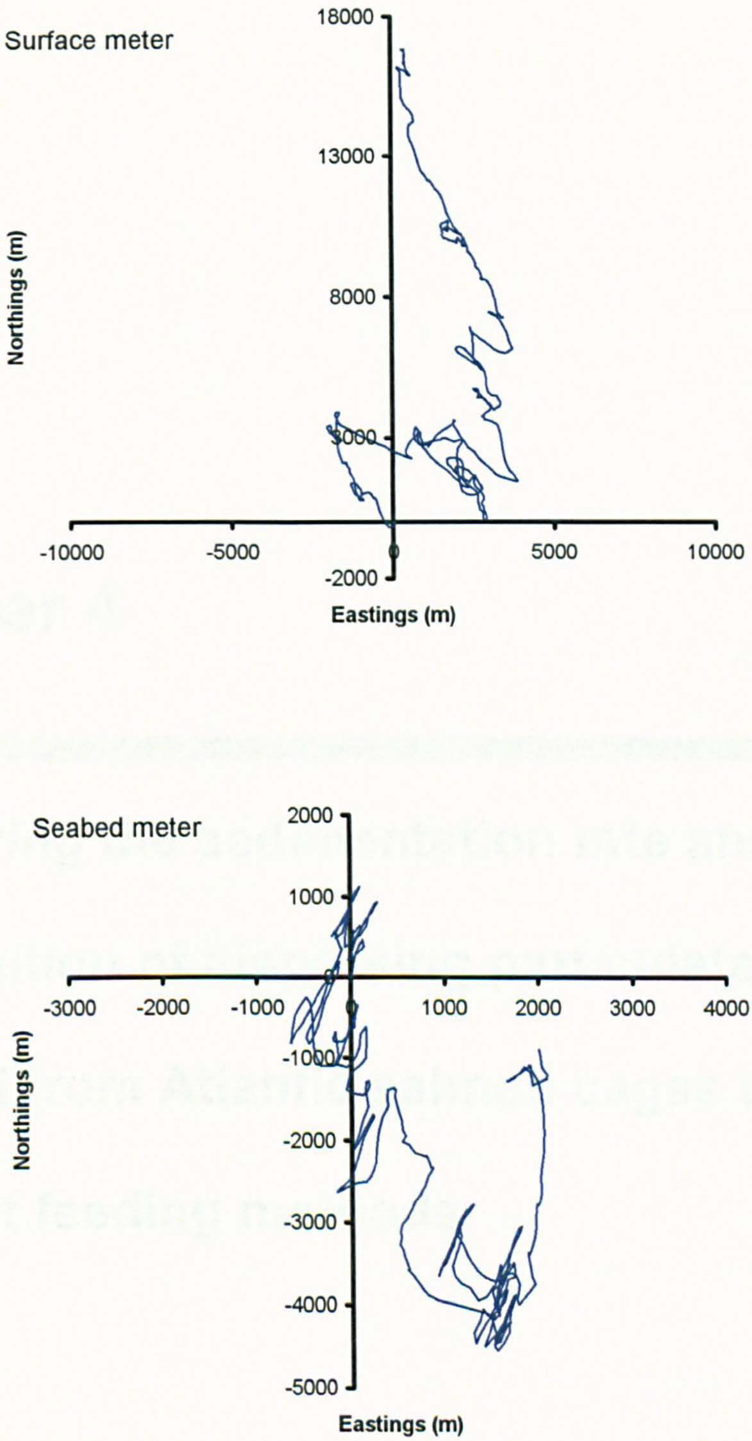


Figure 3.5: Residual current flow at Portavadie on Loch Fyne in Scotland, over 1 tidal cycle (15 days) in August 2001, using a Valeport BFM106 direct recording current meter. Surface meter was at a mean depth of 4.7m, seabed meter mean depth 21.8m.

Chapter 4

Comparing the sedimentation rate and composition of dispersing particulate material from Atlantic salmon cages using different feeding methods

4.1 Introduction

Measuring rates of sedimentation and composition of particulate material is a commonly used technique of assessing fluxes to the seabed. Typically, sediment traps are employed to capture material falling through the water column. They are used especially by oceanographers to measure nutrient and element cycles on an oceanic scale and over long time periods where sophisticated technology is used to capture and fix the particulates *in-situ* (Siegel and Deuser, 1996; Buesseler *et al*, 2000). Sediment traps are also used where output from specific point sources, such as aquaculture, requires quantification.

4.1.1 Sediment trap use in the marine environment

The particular design of sediment trap used and way in which the deployment is conducted depends upon the question being asked but typically oceanographers deploy traps at depths measuring many hundred metres and over extended periods. Despite the ability to collect time series data using traps with pre-programmed closure of the tubes, Buesseler *et al* (2000) described sediment traps as "passive 'rain gauges' used to assess the flux of material in time, space and depth" and as collectors of material for later analysis.

In an assessment of a number of sediment trap designs Bale (1998) suggests that the configuration of the trap and their hydrodynamic response can influence the amount of particulate material captured by the trap in flow velocities of <10 cm s^{-1}. Hence, Bale (1998) argues, sediment traps are not passive collectors. Sediment trap design is, therefore, an important consideration when attempting to collect particulate material falling through the water column. Butman *et al* (1986) assessed a number of characteristics including trap geometry, trap Reynolds number (a dimensionless quantity that is used to define whether the flow of a fluid through or around an object is laminar or turbulent, varying with current speed, fluid viscosity and size of object – Anon, 2004[b]) and the relationship between flow velocity and particle settlement rate. Horizontal and spatial variation in water movement (Siegel and Deuser, 1997), levels of general turbulence and turbulence experienced around the trap opening (Blomqvist and Hakanson, 1981; Butman, 1986; Siegel *et al*, 1990) in combination with geometry (Butman, 1986) act to bias the collection efficiency of the trap. For example, Hargrave and Burns (1979) determined that a height to diameter ratio, also called the Aspect Ratio, of 5:1 should be used to avoid the sample being disturbed by the flow of water across the open tube. When collections are being carried out in unstable water bodies (Hargrave and Burns, 1979), such as those experienced in coastal habitats, collection efficiency is improved when this ratio is increased, by lengthening the tube and/or reducing the diameter.

Gardner *et al* (1983) have assessed the accuracy of using sediment traps to collect particulate samples. They showed that degradation of material in the traps ranged from 0.1-1.0% d^{-1} in deep-sea traps deployed for extended periods. It is likely, therefore, to be higher in traps deployed for similar periods in shallower zones, where productivity is higher and grazing animals are found in increased numbers. In shallow seas grazing zooplankton, collectively known as swimmers (Banse, 1990), is a source of error in measured fluxes by consuming the particulate material collected in the trap but subsequently contributing to the carbon and nitrogen levels in the sample (Banse, 1990; Michaels *et al*, 1990). Levels of consumption by swimmers and microbial degradation can be reduced by the *in situ* use of preservatives, such as formalin, (Hedges *et al*, 1993; Wakeham *et al*, 1993) although such use is generally restricted to longer-term studies.

4.1.2 Particulate waste from aquaculture

All aquaculture operations deposit particulate material onto the seabed. For example, shellfish farms, whilst not explicitly adding feed during the growth period, do concentrate naturally occurring food and increase deposition of pseudofaecal material over a small area. Hayakawa *et al* (2001) used sediment traps to assess the sedimentation flux at an Oyster farm in Japan. Measured fluxes of particulate material ranged from 5 – 390g m^{-2} d^{-1}, with mean Total particulate Carbon (TC) of 2200mg m^{-2} d^{-1} and mean Total particulate Nitrogen (TN) of 290mg m^{-2} d^{-1}.

Deposition around bivalve farms is increased because of the concentration of animals in a relatively small area.

In aquaculture terms, however, the intensive culture of caged marine fish is thought to be more detrimental, with deposition of a significant amount of waste particulate material, such as waste feed and faeces, on the localised seabed. The extent of the deposition is primarily dependent upon water depth, hydrodynamic conditions and particle settling velocity (Chen *et al*, 1999[a]; Cromey *et al*, 2002; Carroll *et al*, 2003). Sediment traps are not routinely used to collect aquaculture waste for analysis during environmental impact studies. However, they are increasingly being used to measure sedimentation rates, to assess the distribution of the settling material and for collection of data to validate models (Cromey *et al*, 2002).

The primary components of waste are carbon and nitrogen and the amount of solid waste that settles on the seabed from fish farms results from a complex interaction between fish biomass, husbandry, feed conversion and seasonality. Salmon growth is temperature dependant (Silvert and Sowles, 1996) with metabolic processes varying over the course of a year as sea temperature changes (Blyth *et al*, 1999). It is therefore imperative that feeding regimes reflect this, so that excess feed is not wasted and sedimentation increased. Faecal material is an amalgam of undigested feed, mucous, intestinal cells and bacteria (Beveridge *et al*, 1991). The amount of faecal matter is a function of the metabolic rate and so varies seasonally and is proportionately lower as fish body size increases (Bergheim *et al*, 1984). Table 4.1 details the total amount of waste estimated to be produced from typical salmonid farms.

Table 4.1: Percentage losses of nitrogen, phosphorous and carbon estimated to be leaving from salmonid fish farms. [1]= assumes % loss to environment = 100%. [2]= Data from Hall *et al*, 1990, 1992; Holby and Hall, 1991. [3]= data from Enell and Lof, 1983; Brattan, 1990; Hall *et al*, 1990, 1992; Holby and Hall, 1991; Beveridge *et al*, 1991; Strain *et al*, 1995. ND = no data. Note that all figures do not add up to 100% due to differing sources of data.

	Lost to Environment[2] (%)	Dissolved form[1,2] (%)	Particulate form[1,2] (%)	Particulates entering sediment[3] Kg t fish prod^{-1}
Nitrogen	67 – 71	72 – 83	12 – 20	71 – 102
Phosphorous	78 – 82	34 – 41	59 – 66	1 – 22
Carbon	65 - 75	4 – 49	29 – 71	ND

Cho (1991) estimated that 205 kg solids are lost per tonne fish production from trout farms though Ackerfors and Enell (1994) suggest a figure nearer 2500 kg. It should be noted that variation in these figures (and those in Table 4.1) arise from species differences, improvements in waste management over time, differences in loss calculations, feed composition, settlement plate collection methods and FCR. Silvert (1994) showed that variations in parameters used to calculate feed losses from growth data, such as FCR, can have significant effects on the calculated waste and are highly significant in sensitivity analysis of waste dispersion models (Brooker, 2002).

The present salmon farming industry's average FCR is 1.1 - 1.3 (Beveridge, pers. comm.) using high-density nutrient (HND) diets. Comparing two theoretical farms, each with consent to grow 500 tonnes of fish, this relatively small difference of 0.2 equates to 100 tonnes of additional feed being added over the growth cycle by the farm achieving an FCR of 1.3 compared to the farm with an FCR of 1.1. The effects of feed conversion on environmental sustainability cannot therefore be underestimated. The said parameters are very difficult to measure with any precision in the environment, however, (Cho, 1991) where the final calculation of FCR has to include mortalities (where fish have been fed but have died before harvest), has to recognise that within a single cage the feeding and growth rates will vary and that there is physical difficulty of collecting waste feed and faecal material in the environment. Talbot and Hole (1994) show there is no *a priori* relationship between FCR and the amount of waste because it is not merely a function of how much food is added but also how the fish use that food, which will vary with digestibility of the feed, fish size and appetite as well as abiotic factors such as temperature.

Data in Table 4.1 do not include the amounts of waste material re-entering the water column in dissolved form from benthic metabolism, which may be as high as 10% (Hall *et al*, 1990, 1992; Holby and Hall, 1991) or the losses from leaching (Chen *et al*, 1999). However, considerable amounts of particulate nutrients are deposited on the seabed under cages (see Chapter 4) and consequently have received much attention in the literature (Gowen and Bradbury, 1987; Lumb, 1989; Beveridge *et al*, 1991; Findlay and Watling, 1994; Hargrave, 1994; Henderson and Ross, 1995).

4.1.3 Sediment trap studies in aquaculture

The use of sediment traps in aquaculture has been fairly limited (Chen, 2000; Cromey *et al*, 2002; Kempf *et al*, 2002) and Gowen *et al* (1991) explains some of the reasons for this. Prior to its employment by Scottish Environment Protection Agency (SEPA) for Environmental Impact Assessments and Discharge Consent Applications, the particulate dispersion model DEPOMOD (Cromey *et al*, 2002), which has been specifically developed for aquaculture application, underwent validation involving deployment of sediment traps around an Atlantic salmon cage. Cromey *et al* (2002) successfully deployed 5 sediment traps but for only a limited period (48-hours) and over a relatively small area. They found a strong similarity between modelled and observed data and regarded the model as validated. However, analysis of the trap layout reveals that the 5 traps used in the Cromey *et al* (2002) study were distributed directly under a cage, to the cage edge only, and thus accounted for a small proportion of the potential depositional area. The estimated deposition at positions away from the cage area within the model presented by Cromey et al (2002) must therefore not be validated. Kempf *et al* (2002) collected particulate material for an equally short period, although over a wider area, but both studies may have encountered a further problem, with the collection of re-suspended material.

When deploying sediment traps around fish farms the depth at which the sediment traps are set is critical. The primary aim is to collect material that has fallen from the cage and to capture it as near to the seabed as possible to represent the point of deposition, but importantly whilst avoiding the collection of re-suspended materials. Kempf *et al* (2002) positioned their sediment traps directly on to the seabed with a trap height of 0.8m and they suggested a "significant" proportion of the material collected was due to re-suspension. Cromey *et al* (2000) used a simple design with a single tube with the top of the trap set at 0.55m above the seabed and may have also suffered from the same problem. In addition, neither study repeated measurements at different stages of the fish growth cycle to assess variation in settling particulate material over time.

4.1.4 Aim of this study

To date no published literature has assessed the environmental consequences of using adaptive feeding technology in salmon culture, in terms of the potential to reduce particulate waste that deposits on the seabed around farms. Small scale studies have indicated the potential under defined and controlled conditions (Chen, 2000; Huntingford, 2001), but no study has been carried out under normal production and management conditions.

The aim of this study is to assess whether the quantity and nutrient composition of particulate material deposited on the seabed under a farm that uses adaptive feeding is different to that deposited on the seabed at a hand-fed site.

Specific objectives of this study are:

a) To assess the quantity, nutrient composition and sedimentation rate of particulate material deposited from a farm that uses an adaptive feeding system, using sediment traps.

b) To assess the quantity, nutrient composition and sedimentation rate of particulate material deposited from a farm that uses traditional hand feeding, using sediment traps.

c) To compare the two feeding methods. The null hypothesis is there is no significant difference in the quantity and composition of material deposited on to the seabed, under each feeding system.

d) To carry out repeated sediment trap sampling to determine whether fish size varies the amount of material deposited. The null hypothesis is that there is no significant difference in the quantity and composition of material deposited on the seabed over time.

4.2 Materials and Methods

The farm sites used were subject to normal farm management practice (such as feeding, boat movement and husbandry) with the fish undergoing typical patterns of production, grading and movement between cages. No farm activities interfered with the deployment and recovery of sediment traps. Collection of particulate material falling from fish farm cages was made using existing sediments traps (Figure 4.1) fabricated at the University of Stirling but based on an original concept by Dunstaffnage Marine Laboratory (Leftey and MacDougall, 1991).

4.2.1 Sediment trap design

Each trap consisted of 4-off PVC tubes of length 60cm and diameter 8cm, giving an Aspect Ratio of 7.5:1 and an effective collection area of $0.05m^2$ per trap. The 4-off PVC tubes were held at 90° from each other on a central ungimballed spigot. The distance between tubes on opposing legs was 43cm. Particulate material falling through the water column and captured in the sediment trap were collected in 150ml (100ml at the reference site) Sterilin polystyrene metal capped containers (Scientific Laboratory Supplies Ltd, Nottingham, England) screwed into the end of each tube. The containers were easily removed on site and were replaced with new containers through the experimental period. No preservative was added to the samples during *in situ* collection but losses due to swimmers and degradation were assumed to be negligible.

4.2.2 Sediment trap deployment positions

Sediment traps were deployed to assess deposition of particulate material from a single cage at the site whilst avoiding interference from remaining cages and interference with the day to day operation at the sites. Four traps were deployed at both Portavadie and Rubha Stillaig, across the main current direction at 90° to the orientation of the cages as indicated by the arrows in Figure 2.1. The GPS position of the respective cage edges are as described in Chapter 3.2. The direction from the cages in which the sediment traps were deployed was the same as for the benthic collections and videographic survey described in Chapter 5.

Four sediment traps were deployed at each site, as shown in Figure 4.2, at distances A, B, C and D. These were, respectively, underneath the cage and 5m, 15m and 25m from the cage edge. Within SEPA's quality standards (SEPA, 2002) 25m is the present limit of the Allowable Zone of Effect (AZE), the mixing depositional area around cages were some degradation of sediment conditions is acceptable.

The mooring of all sediment traps was made using 14mm split film polypropylene rope. The position of each trap was fixed on deployment by a taught surface line with loops at the set distances. One end of the surface line was attached to the circular cage, the other held in position with a 100cm Scanmarin Dhan surface marker buoy (Gael Force, Inverness, Scotland) and a line to an anchor with riser. Sediment traps under the cage where deployed from one side of the cage and pulled into position under the cage, using a second attached line, from the opposite side of the cage. The resting position of this sediment trap was fixed using markers on each line. All traps were anchored to the seabed in fixed positions using concrete blocks of 25-30kg. The top of each sediment trap was set at 3m above the sediment surface by measured rope to reduce the likelihood that re-suspended sediment would interfere in the collection. Sediment traps were maintained in a vertical orientation by 30cm diameter trawl floats (Gael Force, Inverness, Scotland) that were held 3m above the sediment traps, so as not to interfere with the settling of material. As the 4 tubes per trap was an effective square no attempt was made to orientate the direction of the trap, using fins, relative to the current. It was also recognised that the trap would be subject to tilt as a result of current movement, pivoting around the weight on the seabed (after Bonnin *et al*, 2002), but no account was taken of this in subsequent calculations. At the surface sufficient rope was included to take account of the tidal range, the rope was inserted through the loops in the surface line and was marked using a 20cm diameter trawl float (Gael Force, Inverness Scotland).

Figure 4.1 Sediment trap design used to collect particulate material around fish farms. 4-off PVC tubes of length 60cm and diameter 8cm (Aspect Ratio 7.5:1). Tubes held at 90° from each other on a central ungimballed spigot. Distance between tubes on opposing legs was 43cm. Samples accumulate in 150ml Sterilin polystyrene metal capped containers, 100ml at the reference site (inset).

In addition one further trap (total 9 traps in all) was deployed half way between the two sites (see Chapter 3) as a reference. The reference position was at least 600m from either site and was used to collect and assess background particulate settlement. The distance from each of the cage blocks ensured the sediment trap was outside of the area of influence of the cages.

Where appropriate each station was denoted by "P" for Portavadie and "R" for Rubha Stillaig with subscripts representing the distance from the cage edge, except under the cage, which were designated P_0 and R_0 respectively. The reference site was denoted "Ref.".

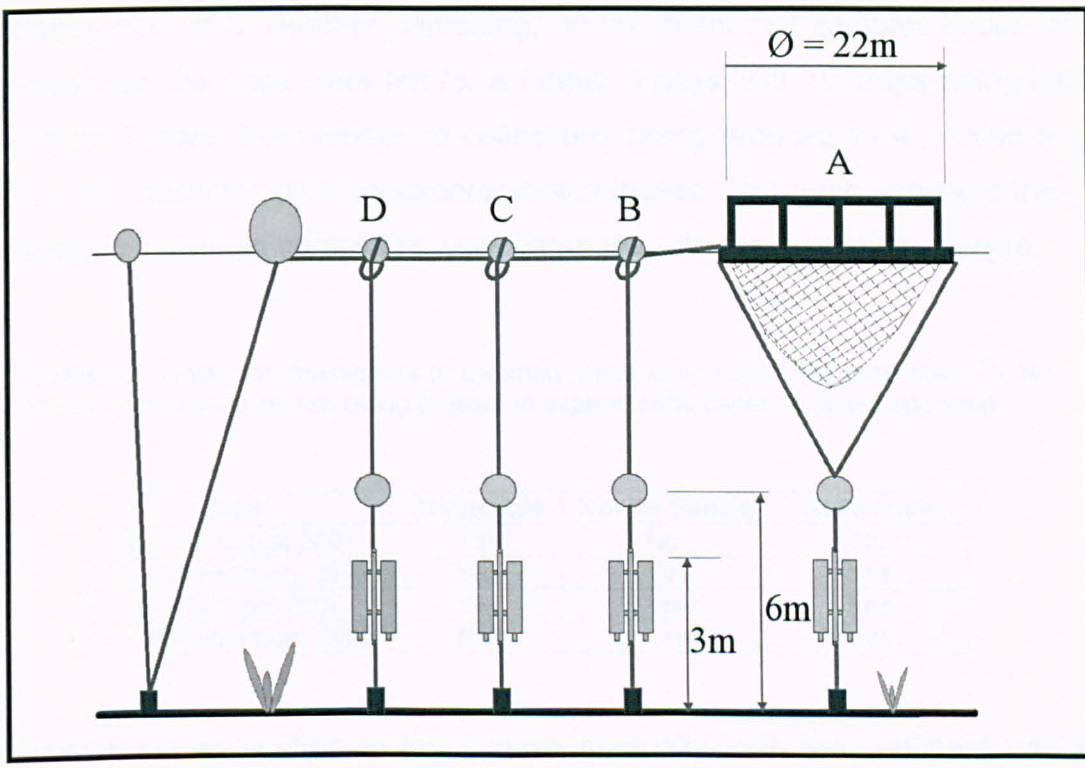


Figure 4.2 Layout of sediment traps in a transect from a 22m-diameter Polar Circle fish farm cage. Sediment Traps were deployed at distances A, B, C and D that were under the cage centre and 5m, 15m and 25m from cage edge respectively. Figure not to scale.

4.2.3 Sediment trap deployment and recovery

The traps were deployed for 1 tidal cycle (15 days) as specified in Table 4.2. In August 2001 there were no fish at the Rubha Stillaig site and in October 2002 the fish held at Portavadie had either been moved or had been harvested and therefore no sediment traps were deployed at these sites on these dates. Samples were collected every 3 days, giving a maximum number of collections per deployment of 5, weather permitting. In the event that samples could not be collected then the traps were left for a further 3 days, with no traps being left for more than 6 days, the number of collections being reduced to 4. Also at the majority of collections all 4 containers were retrieved from each sediment trap but on occasion individual containers were damaged with the loss of the sample.

Table 4.2: Dates for deployment of sediment traps at specified fish farm sites. [1] = No deployment due to no fish being present in experimental cages on dates specified.

Date	Portavadie	Rhuba Stillaig	Reference
16 – 30 August 2001	Yes	No[1]	No
14 – 28 February 2002	Yes	Yes	Yes
16 – 30 April 2002	Yes	Yes	Yes
3 – 18 September 2002	No[1]	Yes	Yes

Sediment traps were lifted to the surface manually, with the sediment trap and attached weight brought onto the boat. The sediment trap was maintained in a vertical orientation to avoid loss of the sample and to allow drainage of excess water in each tube. Polystyrene containers were unscrewed and new containers added prior to redeployment. Recovery and re-deployment of the 9 traps took approximately 2hrs to complete and were collected at the same time ±1hr on each of the collection days.

4.2.4 Laboratory manipulation

After transportation collected samples were placed into a fridge (4°C) overnight and allowed to settle. Water was decanted from the samples using a pipette, without disturbing the particulate material. Samples were washed out, using distilled water, into pre-weighed aluminium trays and dried to a constant weight in an oven at 60°C. To show that distilled water did not add weight to the samples 5 replicates of 15ml of distilled water were placed into pre-weighed aluminium trays (3 d.p. balance) and dried overnight in an oven held at 60°C. All trays showed no increase in weight (one-way ANOVA; $F = 0.14$, $n = 2$, $p = 0.719$). After drying the samples were weighed on a 4dp balance (Mettler AJ100, Mettler-Toledo Ltd, Leicester, UK) and dry weight calculated. Samples were crushed to a powder, using an agate pestle and mortar, to create a homogenous mix and stored in sample containers in a dessicator until analysis for carbon and nitrogen content. Sediment trap samples were analysed for total carbon (TC) and total nitrogen (TN) as described in Chapter 3. Differentiation between organic and inorganic carbon and nitrogen was not required.

4.2.5 Salt content in sediment trap samples

The salt content of sedimented samples is rarely given consideration (Black, pers. comm.) when the amount of material deposited is sufficiently high that the amount of salt is deemed insignificant. However, when sedimentation rates are low the salt content of a relatively low volume of seawater (Figure 4.3) can have a significant overall effect on the weight of the sample and lead to an over-estimation of deposition.

In addition to the particulate material collected, sediment trap samples brought back to laboratory contained 150ml (100ml in reference samples) of seawater representing the size of container used. The majority of this seawater was decanted prior to drying but there remained a small volume (3-10ml) that, when dry, added to the overall weight of the sediment. When calculating sedimentation rate the salt content is irrelevant but to assess the absolute amount of solids and percentage carbon and nitrogen deposited the amount of salt in the samples was critical and account was taken of this in the subsequent calculations.

The water content of 9 samples after decantation, 1 from each of the sampling stations, was assessed by filtering. The mean water volume was 7.4ml for the farm stations and 3.2ml for the reference stations (by virtue of the smaller sampling pot used and the lower volume of sedimenting material that meant more water could be removed prior to drying). The regression line in figure 4.3 shows that salt weight (g) equals 0.0392x water volume (ml) (r^2 = 99.99%) and thus 0.290g (7.4ml) was subtracted from the sediment weight at all farm stations and 0.125g (3.2ml) for the reference station, to adjust for the salt content in the calculation of solids deposited.

The CHNS/O autoanalyser calculates percentage carbon and nitrogen based on the weight of the sample, including salt. To adjust for the new lower weights (excluding salt) the percentages calculated were adjusted (increased) as shown in table 4.3. All samples taken and reported in the results (section 4.3) were adjusted by the factors specified in Table 4.3. To ensure consistency sedimentation rate was calculated using the adjusted figures, although the revised larger carbon and nitrogen percentages of adjusted smaller weights, equate to using the original CHN/O autoanalyzer outputs on the original samples (including salt).

Table 4.3: Adjustment Factors applied to % carbon and % nitrogen measured by CHN/O Autoanalyser to account for removal of salt content. Original weights reduced by 0.29g for all stations except reference (0.125g). Means based on collections made in August 2001 at Portavadie, except Reference collected in February 2002. Proportion = new weight as a % of original weight. n = number of samples.

Station	n	Mean original weight (g)	Mean new weight (g)	Proportion	Adj Factor
Under	20	1.2751	0.9851	77.26	1.29
5m	19	0.7033	0.4133	58.77	1.70
15m	15	0.5466	0.2566	46.94	2.13
25m	19	0.4373	0.1473	33.68	2.97
Reference	15	0.4222	0.2972	70.39	1.42

It was not possible to maintain an absolute amount of seawater in the samples due to the variation in volume of particulate material in the tubes of the sediment trap. The adjustment factor applied to the samples therefore varies with distance from the cage to reflect this variability. The "sticky" nature of the deposited material and poor rates of recovery meant experimental samples could not be filtered prior to analysis.

4.2.6 Analysis

Data were analyzed using the statistical package Minitab v13. The amount of solids deposited in sediment traps was expressed per tonne of production as g m^{-2} t^{-1} d^{-1}; carbon and nitrogen content was expressed as a percentage; and carbon and nitrogen sedimentation rate was defined as "the total amount of material sampled in a sediment trap with a known cross sectional area over a known length of time" (Charles *et al*, 1995) expressed per tonne of production in g C m^{-2} t^{-1} d^{-1} and g N m^{-2} t^{-1} d^{-1} respectively. All data was calculated and expressed in terms of dry weight.

Differences within stations, between collections, were analyzed using one-way ANOVA on data that conformed to normality and equality of variance tests (Bartlett's Test) and differences assessed using Tukey's Pairwise Comparison. Data that was not normally distributed was transformed using standard transformations or using lambda values (λ) generated using the Box-Cox transformation method (Box and Cox, 1964 in Krebs, 1999), where appropriate.

In parametric statistical analysis, data transformations involve changing the scale of the measurement in order to comply with the requirements of the tests being carried out, specifically that the data is normally distributed and that variances between the data are not statistically different. Specific transformations can be applied, such as none, square-root or log, but when these are insufficient (to normalize data) or where a standard transformation is not required then the Box-Cox method provides a more general approach. Using the Box-Cox method, leaving the data untransformed equates to a lambda value of 1, whilst square-root equates to 0.5 (Krebs, 1999). Using Minitab, the Box-Cox method estimates the most appropriate value for lambda for the data presented, transforms the data and stores the converted data for subsequent analysis.

In the event that data remained non-normal or variances where not equal then differences were assessed using the non-parametric Kruskal-Wallis Test. Data within site was pooled and differences between sites were compared using 2-sample t-tests. All transformations are specified in the text.

Sedimentation rate curves, for carbon and nitrogen, were compared using a Factorial Analysis of Variance, comparing the rate of change in sedimentation with distance from the cage centre (regression) across the different collection dates. Data were transformed using the most appropriate method, in this case using natural logarithms, prior to analysis.

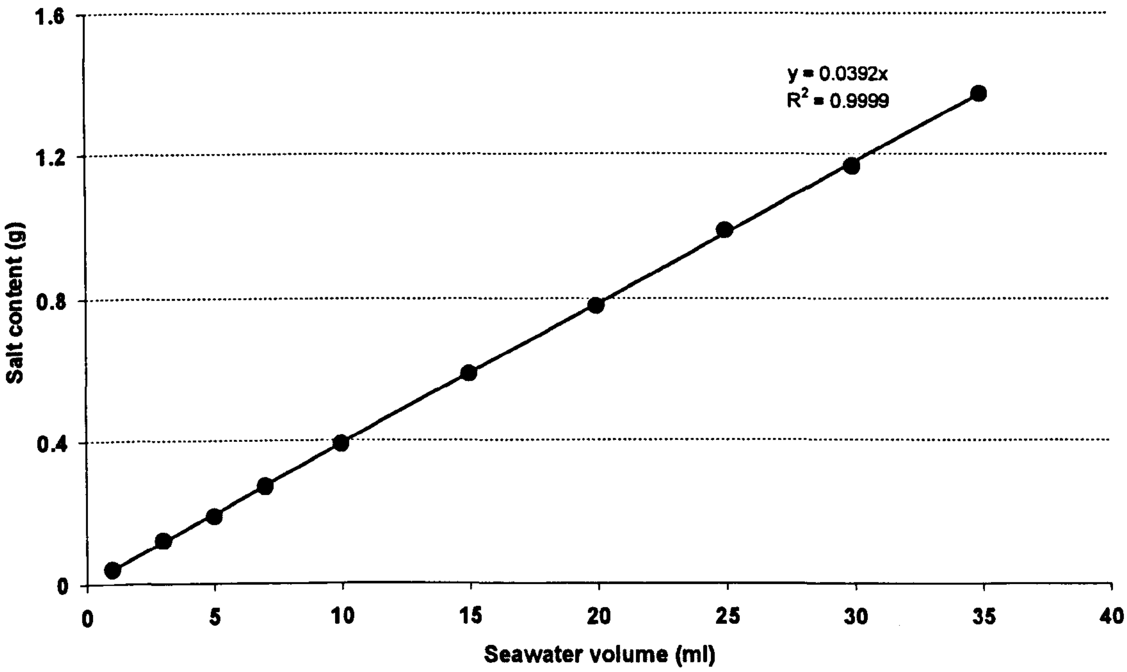


Figure 4.3: Relationship between seawater volume and salt content. Seawater collected from study sites at Loch Fyne.

4.3 Results

4.3.1 Solids Deposition

The maximum mean deposition measured was 320.41 ± 54.01 g m^{-2} d^{-1} under the cage at Portavadie during collection 4 in February 2002 (Appendix 2, Table A2.1) with all 4 tubes of the sediment trap containing identifiable large feed pellets. Smaller medium sized pellets were also found at R_0 at the same collection. At Rubha Stillaig the maximal deposition was 248.12 ± 23.18 g m^{-1} d^{-1}, during collection 3 in April 2002, when large pellets were also identified. Analysis of a sample of unused pellets taken from the fish farm at the time showed the mean dry weight of a single large pellet was 0.94g compared to 0.33g for medium sized pellets (n = 10), with a few large pellets making a large difference to the level of TS deposited. These were the only occasions when whole feed pellets were identified in the sediment traps, with all other occasions' having faecal material only. It was deemed that the use of sediment traps was a poor method to fully assess the deposition of feed pellets at the sites, given the scale of the cages and assessment of feed deposition was abandoned. The weights of feed pellets collected on occasions specified above were removed from subsequent analysis, and the following analysis based on faecal solids (FS) per tonne of production only.

Across both sites there was a general trend of reducing amounts of FS being deposited with increased distance from the cage centre and this is most clearly shown in Figure 4.4. In August 2001 at Portavadie (4^{th} root transformed) and both April (square-root transformed) and September (Box-Cox transformation method λ = -0.562) collections at Rubha Stillaig, FS under cages was significantly higher than was deposited at the remaining farm stations (one-way ANOVA; p = <0.05). At the remaining collections P_0 and R_0 differed from outlying stations only (P_{25} and R_{25}), except February 2002 at both sites where all farm stations had statistically similar FS deposited (one-way ANOVA, untransformed data, p = > 0.05), although P_0 and R_0 encountered higher FS settlement than remaining stations. Stations at 5m, 15m, and 25m did not vary from each other, except in August 2001 at

Portavadie where P_5 differed from P_{25} only. However, lower amounts of material were recorded as the distance from the cage increased. The lack of a distinct difference on some occasions was the result of the wide variation between settlement of FS in the different tubes of the sediment trap at the same station and variations over collection periods.

4.3.1.1 Within-site variation – Portavadie

Figure 4.4, 4.5 and 4.6 shows the quantity of FS deposited at the four Portavadie stations in August 2001, February 2002 and April 2002 respectively. The data presented has had the background deposition, measured at the reference site, removed and is converted per tonne of production, based on the production figures shown in Table 4.4. This enables sites to be compared despite slight differences in production biomass. Production represents the mean fish growth per day, estimated from the food input and FCR (Table 4.4).

Table 4.4: Total and average feed input, FCR and estimated production at Portavaide and Rubha Stillaig fish farms during specified trial periods

Site/Trial Period	Total feed input (Kg)	FCR	Production (Tonnes day^{-1})
Portavadie			
August 2001	4236	1.10	0.256
February 2002	3460	1.16	0.180
April 2002	3152	1.12	0.188
Rubha Stillaig			
February 2002	3280	1.64	0.106
April 2002	4430	1.48	0.180
September 2002	8280	1.20	0.415

All FS deposited at Portavadie were normally distributed and variances were equal within each station and comparisons were made using one-way ANOVA on untransformed data. Over the 3 collections at P_0 the highest variability in FS settlement occurred in April 2002 (Figure 4.6). However, the highest deposition occurred in August 2001, when the experimental cage was double stocked with 47,520 small fish, prior to splitting stock between two cages in December 2001, and may account for the higher deposition recorded during this period. The higher settlement at P_0 may also explain why settlement at remaining stations was lower in August 2001. Overall settlement at P_0 did not differ significantly over the course of the 3 collection periods ($p = 0.237$, $F = 1.67$, $df = 2$).

At P_5, the collection on 24th February was the highest recorded, per tonne of production, of all collections at the site (Figure 4.5). At the time of collection it was noted that the cage had moved and was positioned over the P_5 station as a result of the current flow. Although the collections were averaged over 3 days it is thought that cage movement towards and over the P_5 station may be responsible for the higher settlement on this occasion. As a result of this cage positioning there was a wide variation in settlement at P_5 (193.12 ± 117.11 (SD) g FS m^{-2} t^{-1} d^{-1}) and no overall significant difference in mean settlement at that station ($p = 0.131$, $F = 2.51$, $df = 2$). This movement also resulted in slightly higher settlement at P_{15}, though this was not observed at P_{25} (Figure 4.5).

At P_{15} FS was significantly lower in August 2001 ($p = 0.046$, $F = 4.42$, $df = 2$) as a result of the increased settlement under the cage (P_0), identified above, although a Tukey's pairwise comparison showed no difference in settlement between February and April collections. Despite the lower settlement at P_{25} in August 2001 (Figure 4.4) compared to remaining dates (Figure 4.5 and 4.6) there was no significant difference in FS deposited at the P_{25} station ($p = 0.137$, $F = 2.45$, $df = 2$). Combining data for each site (square-root transformed) for a comparison between collections at Portavadie showed there was no significant difference between the collections made in August 2001, February 2002 and April 2002, with the amount of faecal material deposited per tonne of production being similar ($p = 0.178$, $F = 1.79$, $df = 2$).

There was, however, a general trend noted, with higher settlement (g FS t^{-1} d^{-1}) during the August collection when fish size was small, with progressively lower settlement during later collections in February and then April, as the size of the fish increased.

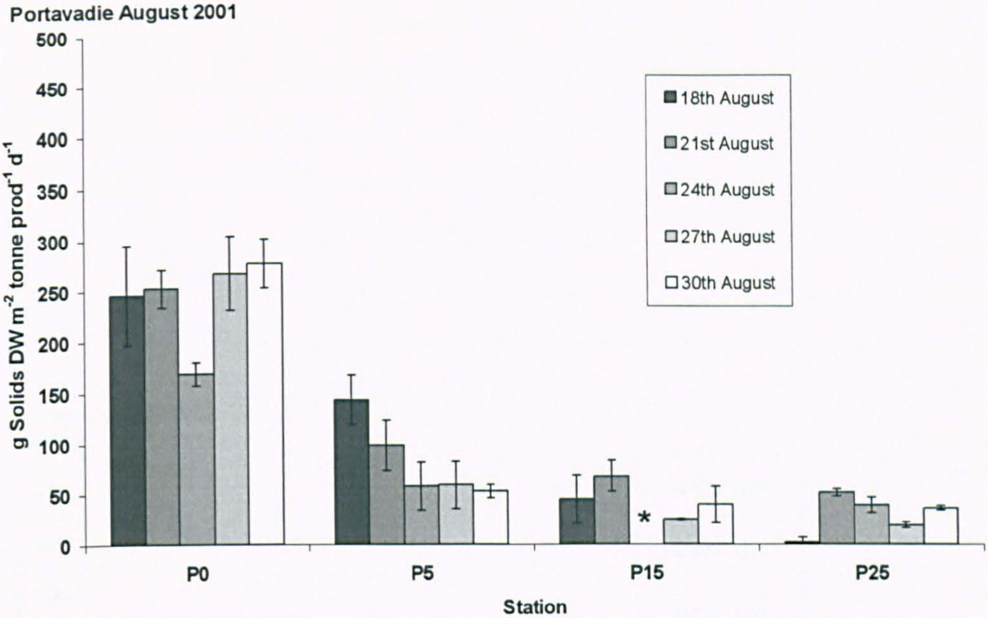


Figure 4.4: Mean deposition of faecal solids with distance from Portavadie fish farm in August 2001. Data collected using sediment traps and adjusted for background deposition. Error bars = standard error where n = 16-20 samples at each collection. Collection was every 3 days. * = missing data due to failure of collection, with mean of subsequent collection based on an increased number of days.

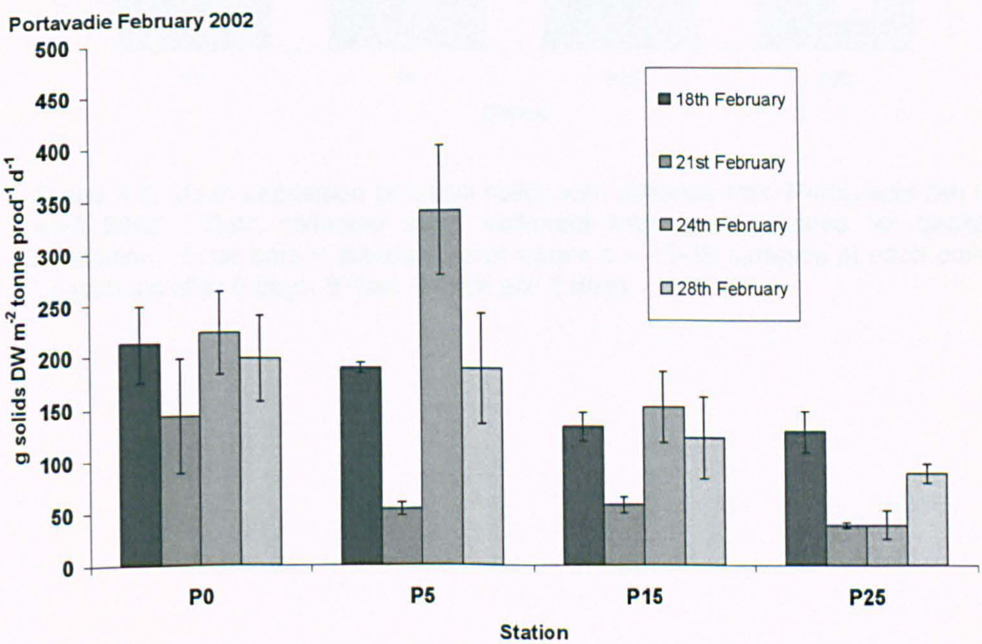


Figure 4.5: Mean deposition of faecal solids with distance from Portavadie fish farm in February 2002. Data collected using sediment traps and adjusted for background deposition. Error bars = standard error where n = 15-16 samples at each collection. Collections after 4 days, 3 days, 3 days and 5 days.

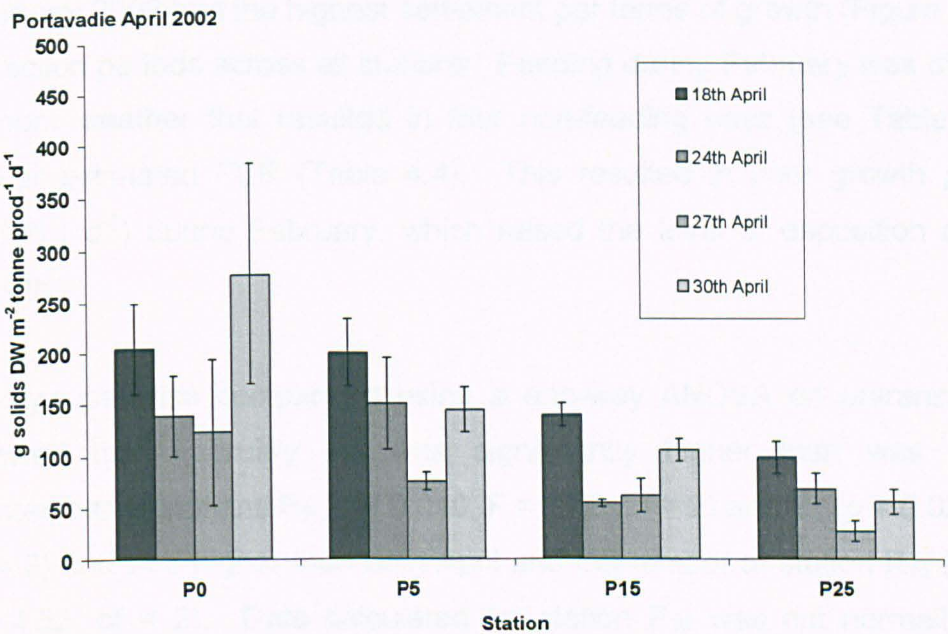


Figure 4.6: Mean deposition of faecal solids with distance from Portavadie fish farm in April 2002. Data collected using sediment traps and adjusted for background deposition. Error bars = standard error where n = 15-16 samples at each collection. Collections after 3 days, 6 days 3 days and 3 days.

4.3.1.2 Within-site variation – Rubha Stillaig

Figures 4.7, 4.8 and 4.9 show the quantity of FS deposited at the four Rubha Stillaig stations in February 2002, April 2002 and September 2002 respectively. The measured quantities of faecal settlement (in g m^{-2} d^{-1}) in September 2002 were higher than recorded in the remaining collections at Rubha Stillaig. However, the higher levels of growth (Table 4.4) during this period resulted in a lower overall quantity of FS per tonne of production, as shown in Figure 4.9. February 2002 had the highest settlement per tonne of growth (Figure 4.7) of the 3 collection periods across all stations. Feeding during February was disrupted due to poor weather that resulted in four non-feeding days (see Table 2.1) and a higher estimated FCR (Table 4.4). This resulted in poor growth performance (0.106 t d^{-1}) during February, which raised the level of deposition per tonne of growth.

Tukey's pairwise comparison using a one-way ANOVA on untransformed data showed that February FS was significantly higher than was collected in September at stations R_0 ($p = 0.010$, $F = 7.48$, df = 2) and R_5 ($p = 0.027$, $F = 5.91$, df = 2) and was higher than both April and September at station R_{15} ($p = 0.003$, $F = 14.52$, df = 2). Data calculated for station R_{25} was not normally distributed despite transformation and was compared using a Kruskal-Wallis test. Analysis suggested that settlement in February was higher than remaining collection periods (varying most from the average rank order) but showed that median settlement of FS at this station was statistically similar across all dates ($H = 5.96$, df = 2, $p = 0.049$).

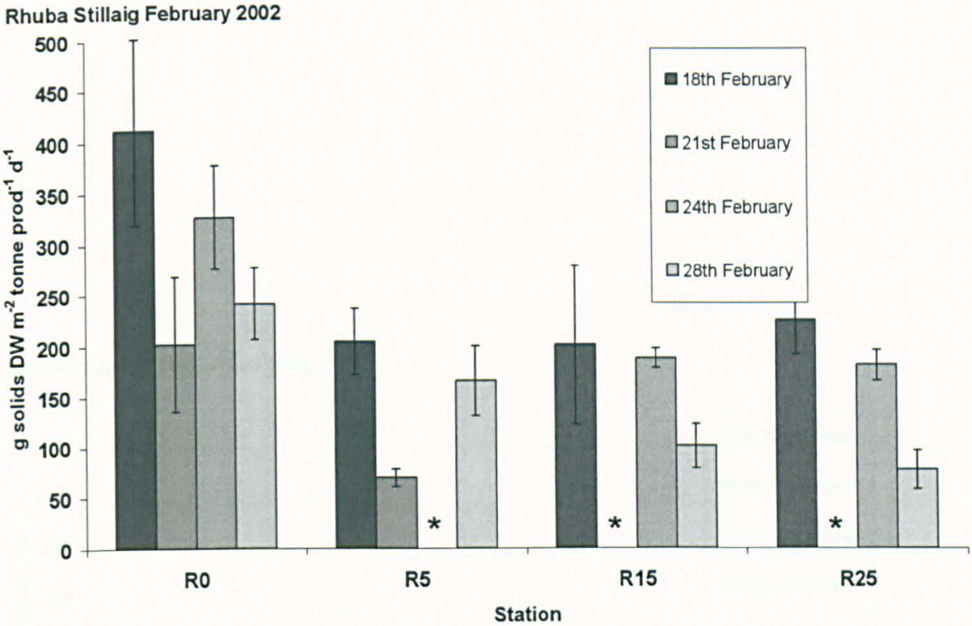


Figure 4.7: Mean deposition of faecal solids with distance from Rubha Stillaig fish farm in February 2002. Data collected using sediment traps and adjusted for background deposition. Error bars = standard error where n = 15-16 samples at each collection. Collections after 4 days, 3 days 3 days and 5 days. * = missing data due to failure of collection, with mean of subsequent collection based on an increased number of days.

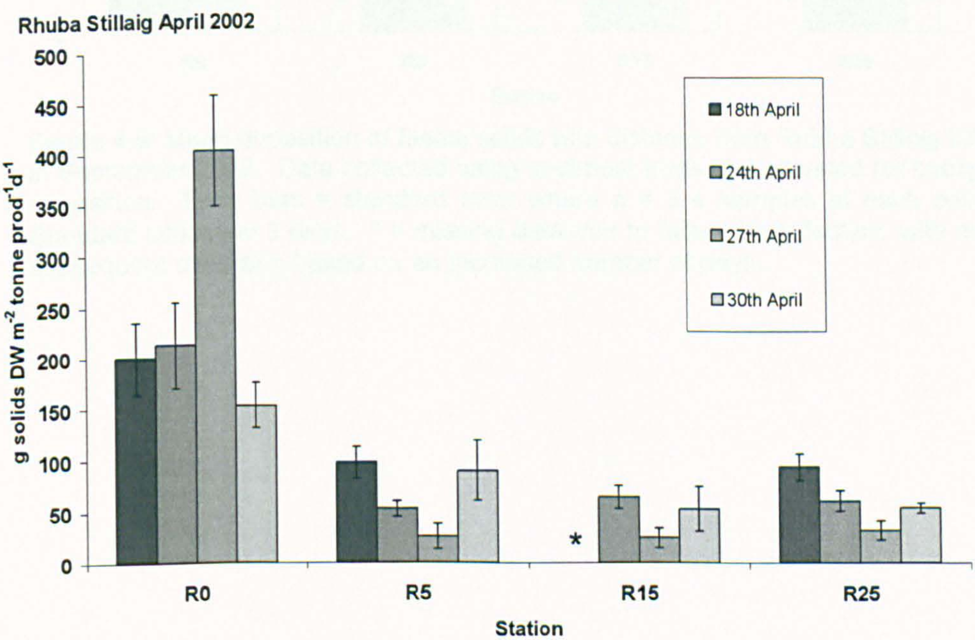


Figure 4.8: Mean deposition of faecal solids with distance from Rubha Stillaig fish farm in April 2002. Data collected using sediment traps and adjusted for background deposition. Error bars = standard error where n = 12-16 samples at each collection. Collections after 3 days, 6 days 3 days and 3 days. * = missing data due to failure of collection, with mean of subsequent collection based on an increased number of days.

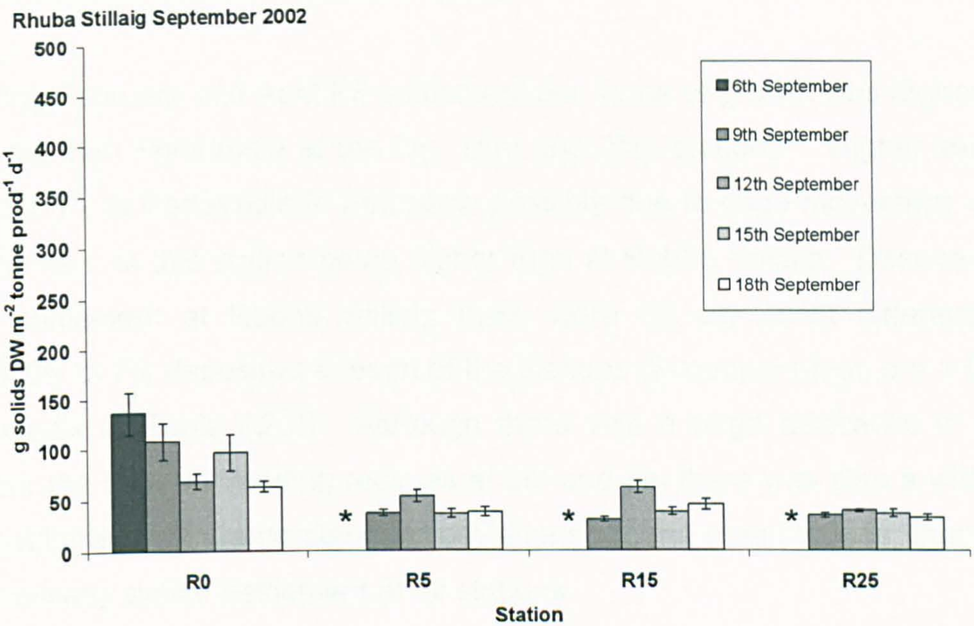


Figure 4.9: Mean deposition of faecal solids with distance from Rubha Stillaig fish farm in September 2002. Data collected using sediment traps and adjusted for background deposition. Error bars = standard error where n = 3-4 samples at each collection. Standard collection 3 days. * = missing data due to failure of collection, with mean of subsequent collection based on an increased number of days.

4.3.1.3 Between-site variation

Although the data for Rhuba Stillaig varied between collection periods (section 4.3.1.2 above), the quantities of faeces collected during each period represents the range of settlement per tonne of growth at each site. Therefore, data for February and April at each site (when sediment trap studies were conducted simultaneously) was pooled (Table 4.5) and comparison made between stations at each site and between sites as a whole.

During February and April FS settlement per tonne of growth was higher at Rubha Stillaig than Portavadie at the 0m, 15m and 25m stations. Higher settlement at station P_5 at Portavadie in February, possibly due to cage movement, resulted in settlement at this station being higher than at Rubha Stillaig. Despite the higher FS settlement at Rubha Stillaig there were no significant differences in the quantity of FS deposited at each of the stations (2-sample t-test, $p = > 0.05$ – See appendix 2, Table A2.2). Although there was a large difference in settlement under the different feeding regimes at 0m and 5m there was also a wide variation in settlement with large standard deviations around mean values, that resulted in statistically similar settlement at all stations.

Table 4.5: Mean settlement of faecal solids (FS) (g FS t^{-1}) for stations at Portavadie and Rubha Stillaig fish farms for collections in February and April 2002 respectively. SE = standard error.

	Stations	0m	5m	15m	25m
Portavadie	Mean	190.0	167.7	102.6	67.6
	SE	18.3	31.1	14.1	12.2
Rubha Stillaig	Mean	268.8	100.7	105.0	103.0
	SE	34.9	23.9	30.1	27.3

There was also a lack of correlation between feed input and the amount of FS deposited in sediment traps under the cages at each site (Table 4.6), which suggests that there is not a simple relationship between feed input and the amount of faeces produced by the fish (as represented by the contents of sediment traps) at a temporal scale of one day.

Table 4.6: Pearson correlation coefficients between the mean faecal solids deposited under cages (g m^{-2} t^{-1} d^{-1}) and average weight of pellet feed added to cages (kg d^{-1}). * = significant.

Date	Site	Coefficient	p
Aug-01	Portavadie	-0.449	0.449
Feb-02	Portavadie	-0.500	0.500
Feb-02	Rhuba Stillaig	0.210	0.790
Apr-02	Portavadie	0.478	0.522
Apr-02	Rhuba Stillaig	-0.038	0.962
Sep-02	Rhuba Stillaig	-0.724	0.167

4.3.2 Percentage carbon and nitrogen

There was wide variation in the measured percentages of total carbon (%TC) and total nitrogen (%TN) (Figures 4.10 to 4.14) in sediment trap samples depending upon the amount of feed pellets, faecal material and general background material deposited. In general the percentage carbon present was mirrored by the percentage nitrogen as shown by the strong correlation between carbon and nitrogen content of settled particulates ($p = <0.001$ – see Appendix 2, Table A2.3). High %TC and %TN content were observed when the amount of FS (FS in g m^{-2} t^{-1} d^{-1}) was high and *vice versa*. This occurred predominantly under the cages, with strong positive correlations (Spearman's coefficient; $p = <0.05$) at all collections from R_0 and P_0, except August 2001 at Portavadie (see Appendix 2, Table A2.4). In general correlation values reduced with increased distance from the cage ($p = >0.100$), indicating that the lower %TC and %TN in background material (Figure 4.14) became increasingly prominent.

There were no significant correlations between FS, %TC and %TN at the reference site for collections in April and September, but there was a strong negative correlation between FS and %TC in February 2002. There was an increased deposition of solids at the April Reference collection, which was mirrored with an increase in %TN through the collection period and higher then normal TC (Figure 4.14), as a result of an algal bloom that increased settlement during April over the remaining collections.

Maximum %TC and %TN content occurred under the cages (Table 4.7). At Rubha Stillaig the quantity of material deposited in April 2002 was commensurate with feed pellets being present as specified in 4.3.1 above. Overall the minimum and maximum %TC and %TN content of the settled particulate material were similar under the two feeding regimes as shown in Table 4.5. At Portavadie the minimum %TC and %TN was broadly similar at P_0 and P_5 and at P_{15} and P_{25} but the maximum values peaked under the cage, where deposition is known to be highest. There was a similar deposition of %TC and %TN at R_0 and P_0 stations (2-sample t-test $t = -0.76$, $df = 59$, $p = 0.451$ and Mann-Whitney U-test $W = 856.5$, $p = 0.215$ for TC and TN respectively). The remaining stations at Rubha Stillaig contained a lower percentage of both carbon and nitrogen than the equivalent stations at Portavadie (Mann-Whitney U-tests, $p = <0.001$, see appendix 2, Table A2.4). This was unusual as faecal settlement has been shown to be similar between sites and its carbon and nitrogen content would be expected to be similar also.

The difference would in part be due to differences in carbon and nitrogen content of feed pellets (Portavadie, carbon 49.5 – 52.5% [n=10], nitrogen 6.25 – 6.55% [n = 10]; Rubha Stillaig, carbon 49.5 – 51.1% [n = 10], nitrogen 7.5 – 7.6% [n = 20]), where the 1% difference in Nitrogen equates to a 6.25% difference in crude protein (assuming protein is 16% nitrogen). This difference affected the carbon and nitrogen content of the faeces. There may have been variations in the proportion of background settlement at each site, which would contain a lower carbon and nitrogen content and therefore vary the composition. It was not possible to differentiate, in percentage terms within a single sample, between that material originating from the cage and that from general background material.

Table 4.7: Minimum and maximum mean percentage total carbon (%TC) and total nitrogen (%TN) measured from sediment trap samples collected at experimental fish farm sites at stations under and at specified distances from cage edge.

	Portavadie				Rhuba Stillaig			
Station	MinTC	MaxTC	MinTN	MaxTN	MinTC	MaxTC	MinTN	MaxTN
Under	11.36	51.11	0.76	6.44	8.89	56.78	0.65	7.51
5m	10.35	29.82	0.91	2.78	3.63	15.83	0.37	2.68
15m	5.16	33.76	0.54	3.41	2.31	14.68	0.43	1.17
25m	5.02	29.33	0.59	3.39	3.68	11.23	0.47	1.91
Reference	0.95	4.03	0.26	1.96				

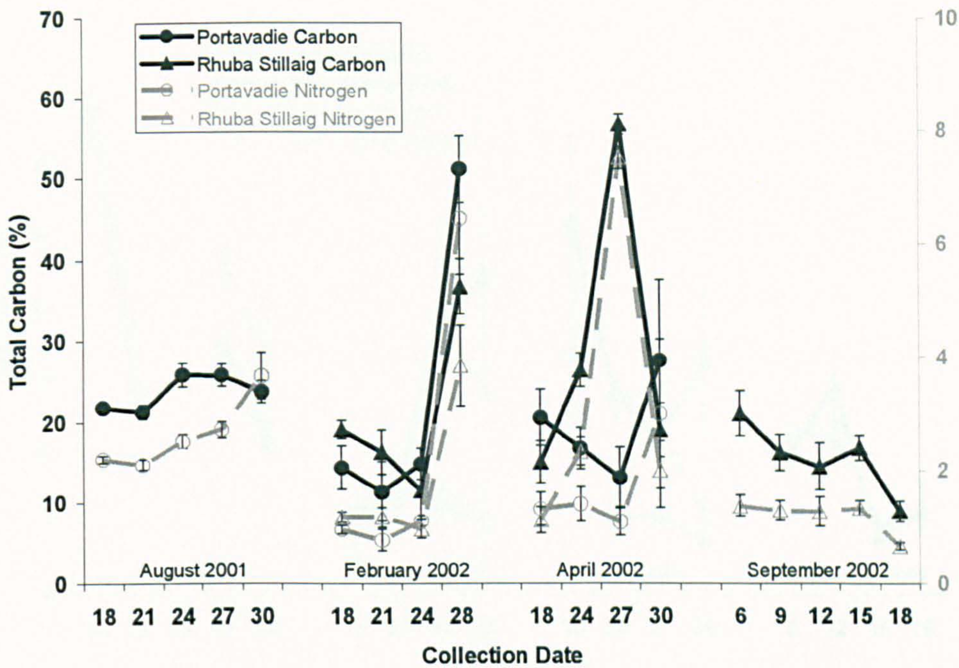


Figure 4.10: Mean percentage total carbon and total nitrogen measured using CHN/O autoanalyser combustion method in samples collected from underneath experimental cages using sediment traps. Values not adjusted with background levels. Error bars = standard error where n = 3 or 4 samples per collection.

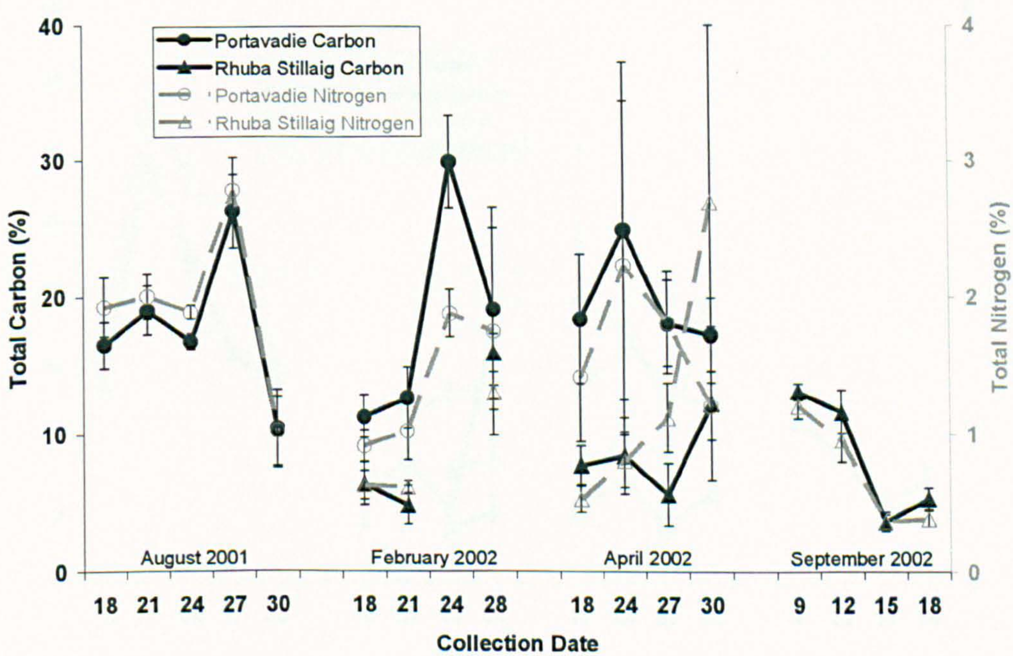


Figure 4.11: Mean percentage total carbon and total nitrogen measured using CHN/O autoanalyser combustion method in samples collected at 5m distance from experimental cages using sediment traps. Values not adjusted with background levels. Error bars = standard error where n = 3 or 4 samples per collection.

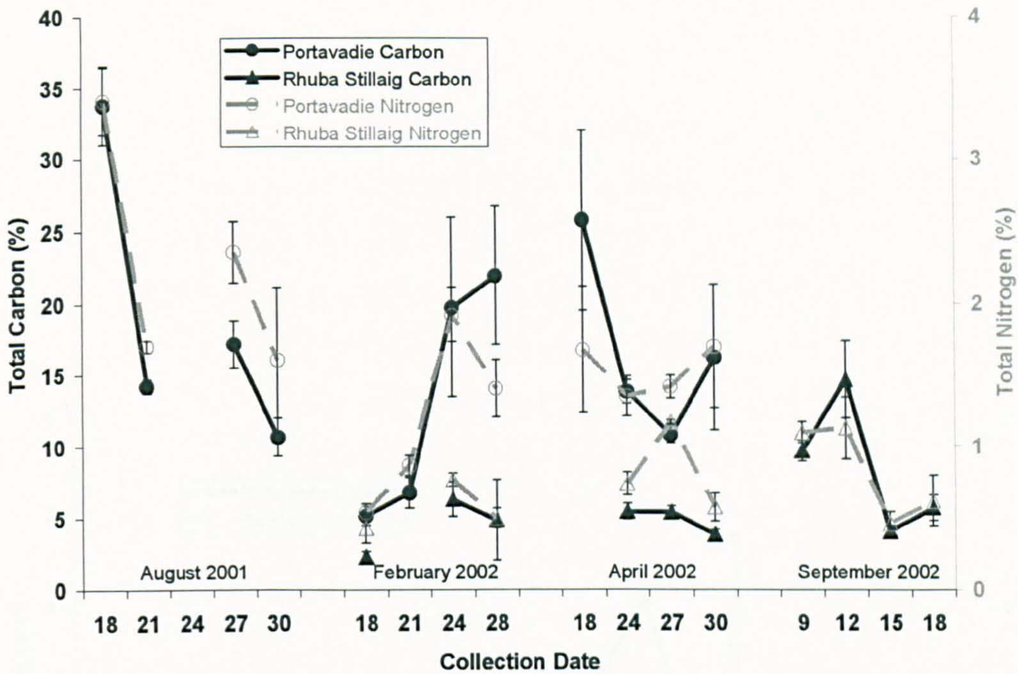


Figure 4.12: Mean percentage total carbon and total nitrogen measured using CHN/O autoanalyser combustion method in samples collected at 15m distance from experimental cages using sediment traps. Values not adjusted with background levels. Error bars = standard error where n = 3 or 4 samples per collection.

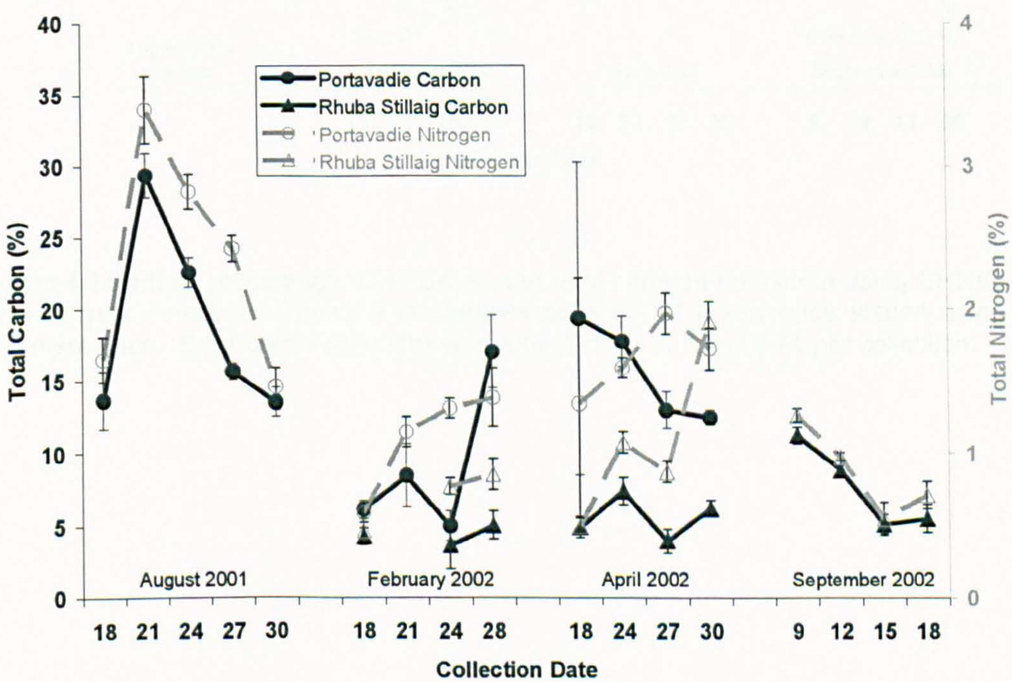


Figure 4.13: Mean percentage total carbon and total nitrogen measured using CHN/O autoanalyser combustion method in samples collected at 25m distance from experimental cages using sediment traps. Values not adjusted with background levels. Error bars = standard error where n = 3 or 4 samples per collection.

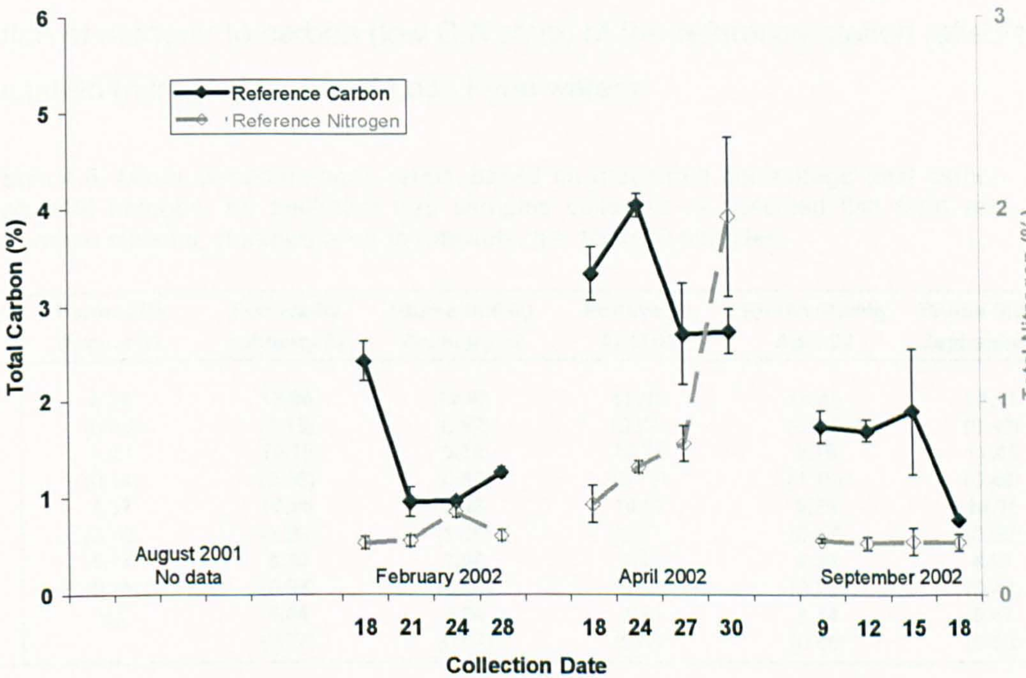


Figure 4.14: Mean percentage total carbon and total nitrogen measured using CHN/O autoanalyser combustion method in samples collected at a reference station using sediment traps. Error bars = standard error where n = 3 or 4 samples per collection.

4.3.3 Carbon nitrogen ratios

Mean carbon/nitrogen (C:N) ratios, based on the %TC and %TN, are shown in Table 4.8. At all collection dates highest C:N ratios were recorded at P_0 and R_0, ranging between 9.29 and 13.19 and all stations reduced with distance from the cage centre. However, at all farm stations C:N ratios were higher than recorded from the background deposition of particulate material at the reference station, and can be attributed to the deposition of material of fish farm origin. The high proportion of nitrogen to carbon (low C:N ratio) at the reference station reflects the hypernutrified (nitrogen) status of Loch Fyne waters.

Table 4.8: Mean carbon/nitrogen ratios, based on measured percentage total carbon and total nitrogen, for sediment trap samples collected at specified fish farm and reference stations, standard error in brackets. n = 16 to 20 samples.

Station	Portavadie August 01	Portavadie February 02	Rhuba Stillaig February 02	Portavadie April 02	Rhuba Stillaig April 02	Rhuba Stillaig September 02
Under	**9.29** (0.35)	**12.86** (1.18)	**12.97** (0.87)	**13.00** (0.78)	**10.85** (0.69)	**13.19** (0.49)
5m	**9.23** (0.14)	**13.10** (0.66)	**9.82** (0.85)	**12.75** (0.71)	**9.18** (1.15)	**11.85** (0.49)
15m	**8.37** (0.40)	**10.96** (1.39)	**7.66** (1.08)	**10.97** (0.87)	**6.26** (0.47)	**10.35** (0.60)
25m	**8.14** (0.25)	**8.50** (0.94)	**7.00** (0.90)	**9.80** (0.98)	**6.33** (0.73)	**8.97** (0.42)
ref	**nd**	**4.84** (0.77)	**4.84** (0.77)	**4.74** (0.68)	**4.74** (0.68)	**5.67** (0.53)

C:N ratios at each station were compared between sites using 2-sample t-tests, on pooled data for February and April only, when samples were collected simultaneously. All data for 0m, 5m and 25m were normally distributed using untransformed data, whilst 15m was 4^{th}-root transformed before analysis. At all stations except 0m the C:N ratio was significantly higher at Portavadie than at the Rubha Stillaig site (2-sample t-test, p = <0.05 – see Appendix 2, Table A2.5) reflecting the higher %TC and %TN at these stations reported in 4.3.2. However, differences in C:N ratio do not appear to reflect the feeding regime used at each site but do, in part, reflect the differences in C:N ratio of the feed pellets being used at each site at the time of collection, with Portavadie using feed pellets with a higher CN ratio. This in turn would affect the C:N ratio of the faeces being deposited from each of the cages.

4.3.4 Carbon and nitrogen sedimentation rates

Sedimentation rate for both carbon and nitrogen, standardized per tonne of growth, at each of the collection dates at Portavadie and Rubha Stillaig are presented in Figures 4.15 to 4.20. All sedimentation rates are based on faeces settlement only (FSR = faecal sedimentation rate), given the limited number of occasions when feed was collected. FSR for both carbon and nitrogen, reduced exponentially with distance from the cage centre (R^2 = >0.62).

Using ln(x) transformed data, carbon FSR with distance provided a good fit to a straight line model (Pearson Correlation Coefficient, p = <0.05) and the carbon component only was analyzed by regression and a General Linear Model Factorial ANOVA. Comparisons were made between (1) all Portavadie collections, (2) all Rubha Stillaig collections and (3) data from February and April 2002 only to test for differences between sites. Curvilinearity was assessed in the respective residual plots and approximate normality in the standardized residuals (Anderson-Darling test p = >0.05). Due to the low number of data points at each station, significant difference in residual variance was assessed using median values (Levene's Test p = >0.05), before analysis.

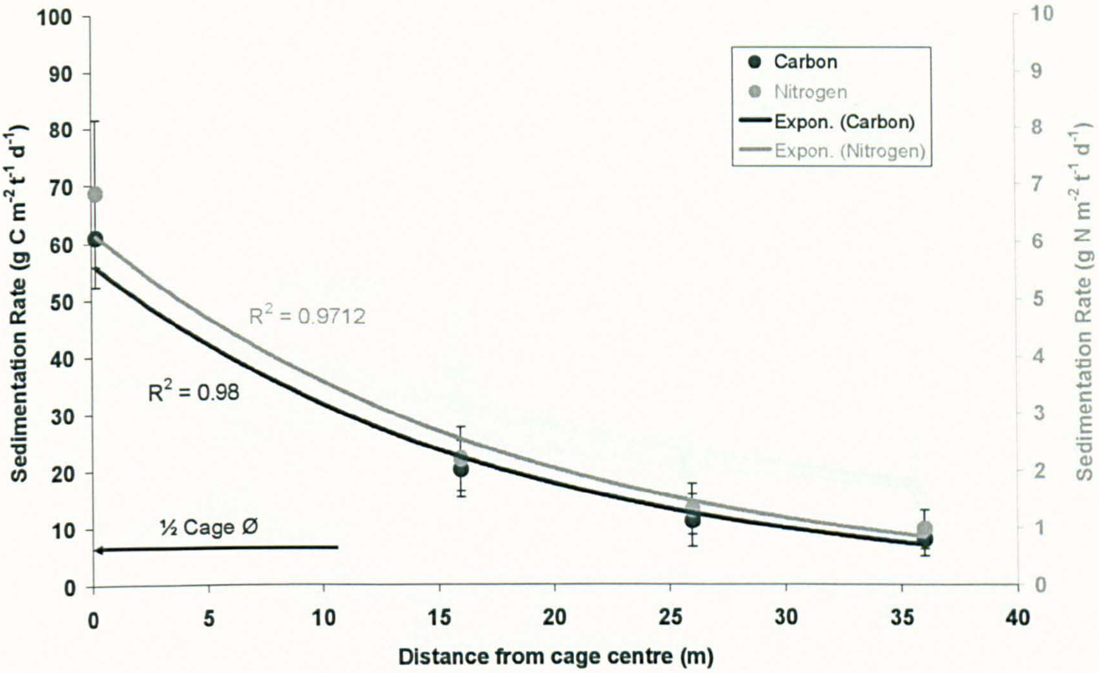


Figure 4.15: Mean sedimentation rate of carbon and nitrogen with distance from cage centre at Portavadie for one sediment trap deployment of 15 days in August 2001. Error bars = standard error where n = 15 to 24 samples collected.

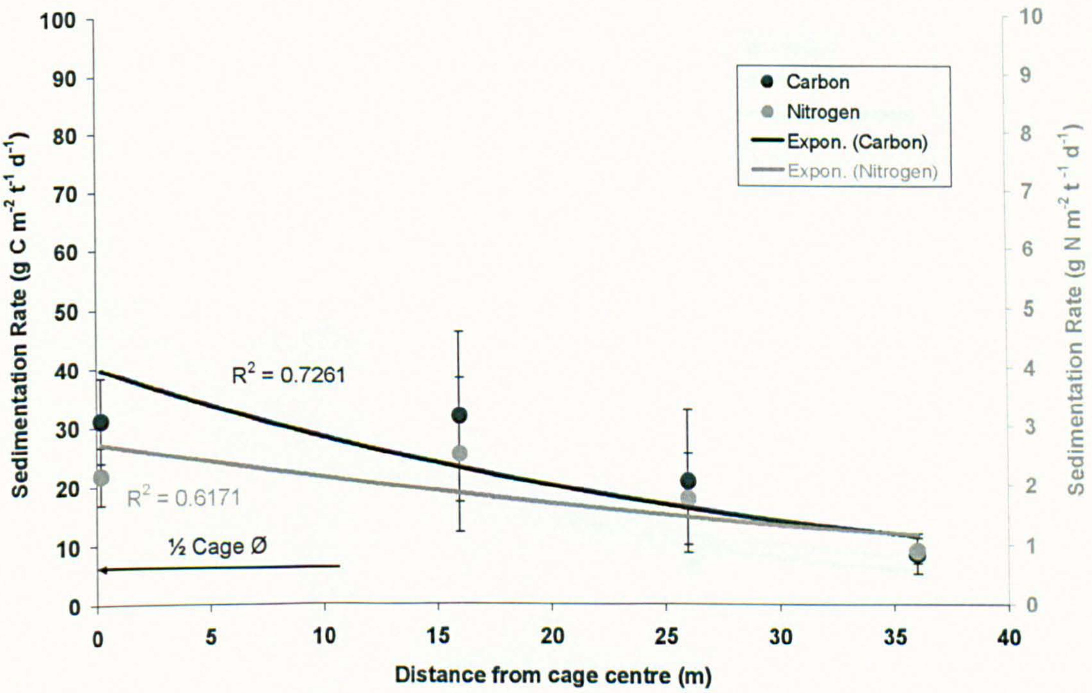


Figure 4.16: Mean sedimentation rate of carbon and nitrogen with distance from cage centre at Portavadie for one sediment trap deployment of 15 days in February 2002. Error bars = standard error where n = 15 to 16 samples collected.

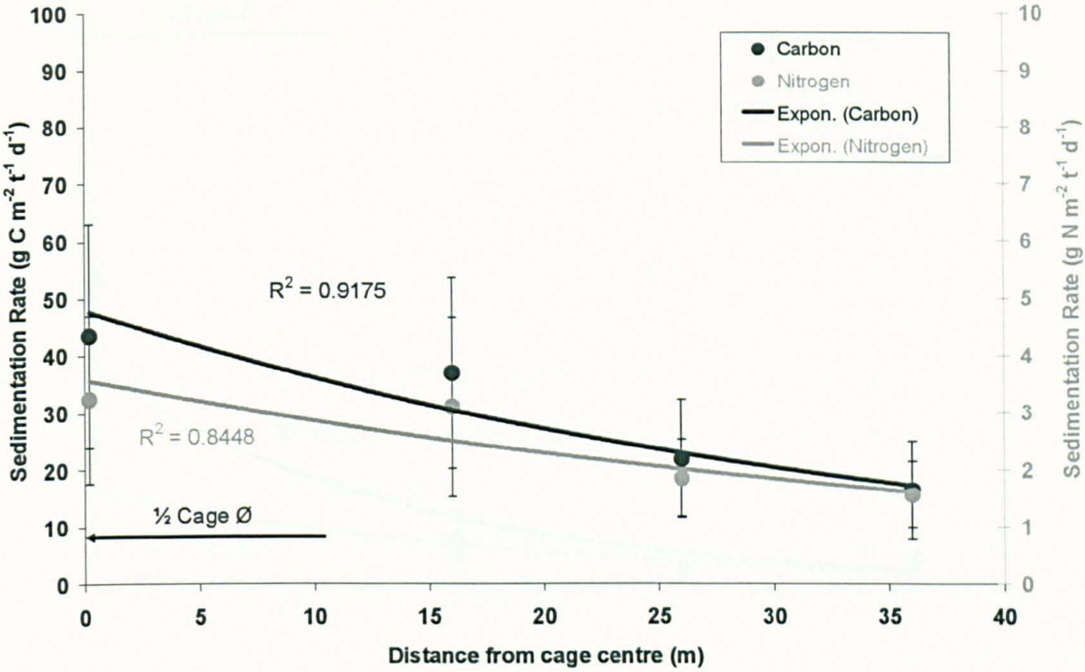


Figure 4.17: Mean sedimentation rate of carbon and nitrogen with distance from cage centre at Portavadie for one sediment trap deployment of 15 days in April 2002. Error bars = standard error where n = 15 to 16 samples collected.

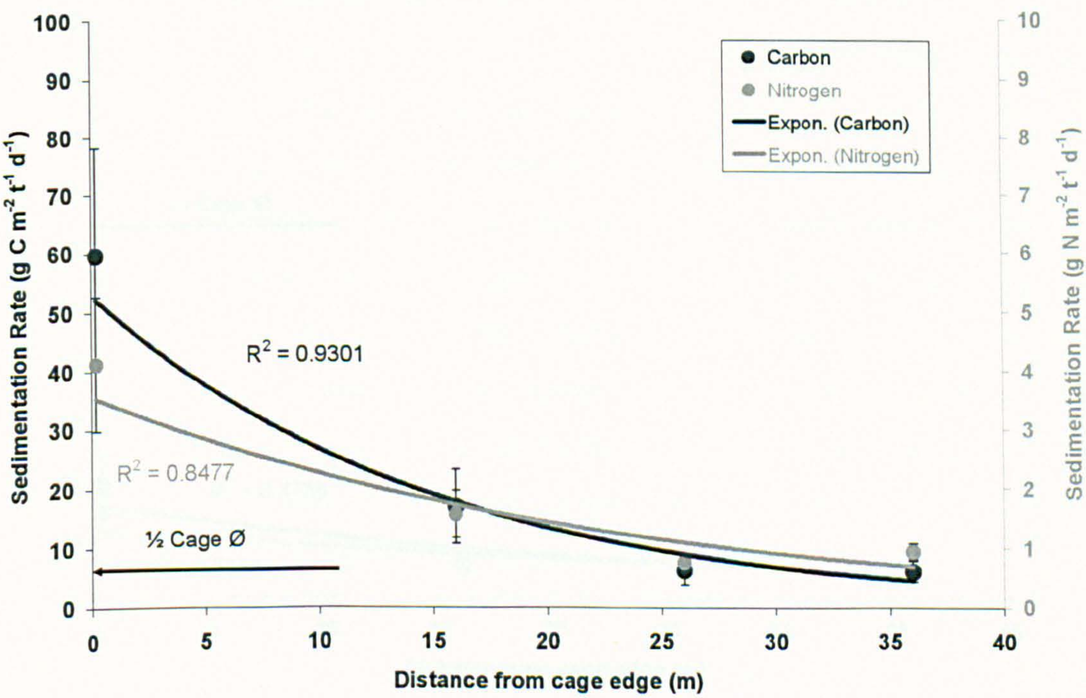


Figure 4.18: Mean sedimentation rate of carbon and nitrogen with distance from cage centre at Rubha Stillaig for one sediment trap deployment of 15 days in February 2002. Error bars = standard error where n = 11 to 16 samples collected.

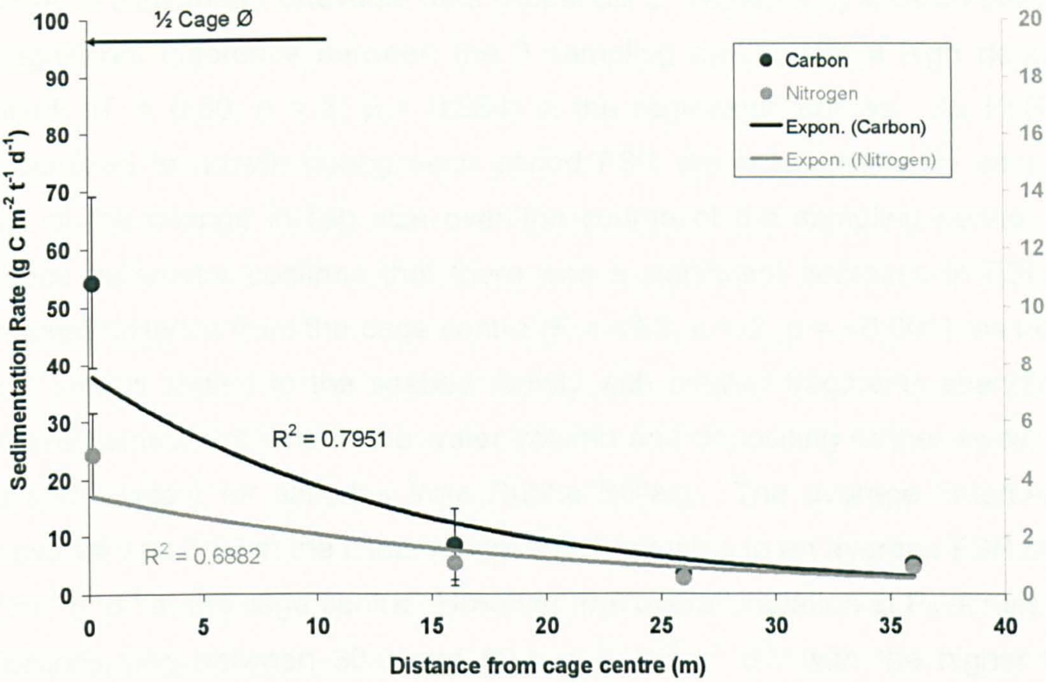


Figure 4.19: Mean sedimentation rate of carbon and nitrogen with distance from cage centre at Rubha Stillaig for one sediment trap deployment of 15 days in April 2002. Error bars = standard error where n = 12 to 16 samples collected.

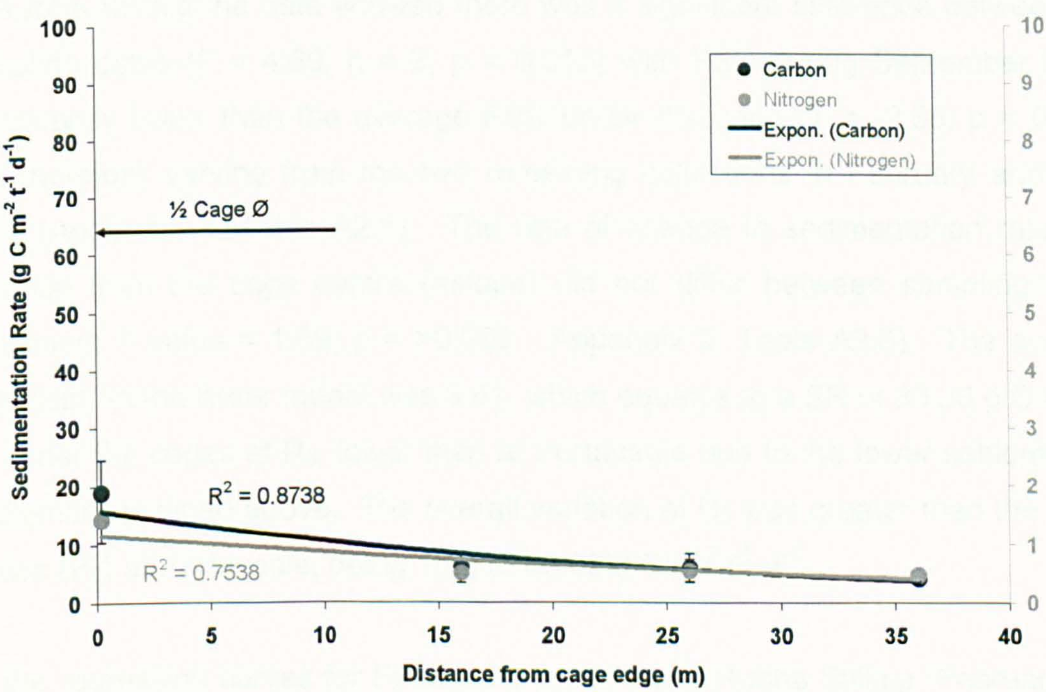


Figure 4.20: Mean sedimentation rate of carbon and nitrogen with distance from cage centre at Rubha Stillaig for one sediment trap deployment of 15 days in September 2002. Error bars = standard error where n = 14 to 25 samples collected.

ANOVA of combined Portavadie data (Appendix 2, Table A2.5) showed there was no significant difference between the 3 sampling dates, with a high degree of similarity ($F = 0.58$, $n = 3$, $p = 0.564$) in the regression curves. As FSR was standardized to growth during each period FSR did not significantly alter as a result of the change in fish size over the course of the sampling period. The distance parameter confirms that there was a significant decrease in FSR with increased distance from the cage centre ($F = 49.3$, $n = 2$, $p = <0.001$), as heavier faecal strings settled to the seabed rapidly with smaller fragments spending an increased amount of time in the water column and depositing further away. This was also evident for samples from Rubha Stillaig. The average "intercept" at Portavadie was 3.82 in the linear model which equates to an average FSR of 45.6 g C m^{-2} t^{-1} d^{-1} at the cage centre. However, the overall variation at P_0 across the 3 collections was between 30.0 and 69.4 g C m^{-2} t^{-1} d^{-1}, with the higher figure resulting from the presence of identifiable faecal strings with higher carbon content in some collected samples and not in others.

At Rubha Stillaig the data showed there was a significant difference between the sampling dates ($F = 4.69$, $n = 3$, $p = 0.015$) with FSR during September being significantly lower than the average FSR under the cage ($T = -2.86$, $p = 0.031$) and therefore varying from the two remaining collections in February and April 2002 (Appendix 2, Table A2.6). The rate of change in sedimentation rate with distance from the cage centre (=slope) did not differ between sampling dates (maximum T value = 1.69, $p = >0.093$ - Appendix 2, Table A2.6). The average "intercept" in the linear model was 3.41, which equates to a SR of 30.26 g C m^{-2} t^{-1} d^{-1} under the cages at R_0, lower than at Portavadie due to the lower settlement in September outlined above. The overall variation at R_0 was greater than the same station (P_0) at Portavadie, being 10.8 to 66.02 g C m^{-2} t^{-1} d^{-1}.

As the regression curves for FSR at Portavadie and Rubha Stillaig (February and April data – see Appendix 2, Table A2.7) did not vary between sampling dates within respective sites, data for the occasions when sampling took place simultaneously at both sites was pooled and comparison made between the two sites.

Analysis of the mean regression curves for each site (Appendix 2, Table A2.7) shows that the amount of faecal carbon settling onto the seabed under the cages at P_0 and R_0 (the intercept) was highly similar (41.3 ± 1.2 g C m^{-2} t^{-1} d^{-1} and 39.6 ± 1.2 g C m^{-2} t^{-1} d^{-1} at P_0 and R_0 respectively; intercept T = 0.08 and -0.08 p = 0.933). The regression curves are shown in Figure 4.21. Therefore, there were no differences in FSR under the cages between the two feeding regimes. However, the distribution of FSR did vary at remaining stations along the transect, as shown by significant differences in the two slopes from the average slope (T = 2.52 and -2.52, p = 0.015 for Portavadie and Rubha Stillaig respectively). The rate of change in sedimentation with distance was lower at Portavadie than at Rubha Stillaig, which means that the sedimentation rate of carbon would return to background levels at a shorter distance from the cage at the Rubha Stillaig site.

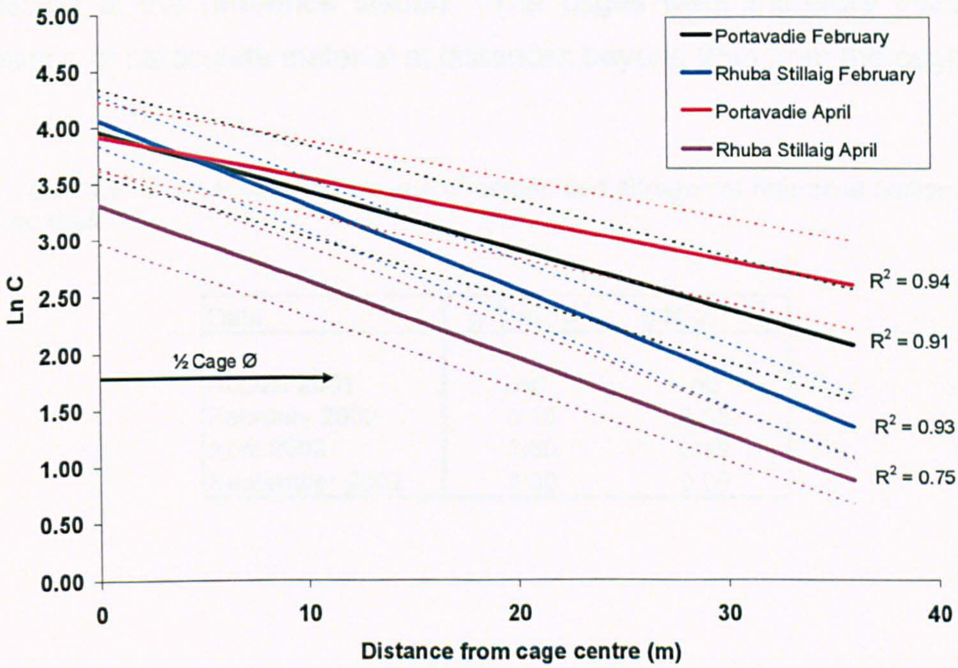


Figure 4.21: Sedimentation rate of faecal carbon at Portavadie and Rubha Stillaig in February and April 2002, based on natural log regression of carbon sedimentation rate (in g C m^{-2} t^{-1} d^{-1}). Dotted lines represent ± standard error (line).

Average sedimentation rate (SR) for the 3 collections made at the reference site is shown in Table 4.9. Maximum background SR was measured in April 2002 at 0.87 g C m^{-2} d^{-1} and 0.43 g N m^{-2} d^{-1} when visual observation of the collected material suggested that phytoplanktonic debris was settling out after a bloom period. Non-standardized carbon SR (in g C m^{-2} d^{-1}) at P_{25} and R_{25} was 3 – 8 times higher and nitrogen SR up to 3.6 times higher on average than the rate of deposition at the reference station. The cages were therefore increasing the deposition of particulate material at distances beyond 25m from the cage edge.

Table 4.9: Mean sedimentation rate of carbon and nitrogen at reference station. nd = no data.

Date	g C m^{-2} d^{-1}	g N m^{-2} d^{-1}
August 2001	nd	nd
February 2002	0.18	0.05
April 2002	0.60	0.19
September 2002	0.30	0.05

4.4 Estimated depositional area and total faecal deposition

The sedimentation rate using non-standardized data from Portavadie and Rubha Stillaig, for February and April 2002, showed no significant differences between the average regression curve and the respective regression curves for each site ($F = 0.70$, $n = 2$, $p = 0.407$ – see Appendix 2, Table A2.8). The respective mean carbon regression curves were used to estimate maximum depositional distance in the cross-current direction at each site and from this to estimate the total depositional area around the respective cages.

Carbon sedimentation rate returned to background levels at an estimated 51m from the cage centre at Portavadie and 40m at Rhuba Stillaig (Figure 4.22). Assuming the magnitude of the current and its direction are equally distributed then the resulting spatial area of deposition for individual cages at Portavadie and Rubha Stillaig was 8,171m^2 and 5,025m^2 respectively. In a simplified model, however, Gillibrand *et al* (2002) have suggested that settlement area forms an elliptical shape, equal to $\pi\ (D_x.D_y)$ where:

$$D_x = U_s.H_s/W_i \qquad (1)$$

$$D_y = V_s.H_s/W_i \qquad (2)$$

Where D_x and D_y are deposition distance from cage centre in the main and cross-current direction respectively, U_s = main current speed (m s^{-1}), V_s = cross-current speed (m s^{-1}), H_s = water depth (m) and W_i = faecal settling velocity (m s^{-1}).

At the sites the estimated distance D_y was defined in part by the position of the sediment traps deployed on the transect in the cross-current direction and by the resulting sedimentation rate curves for carbon and nitrogen. In this study H_s and W_i were the same so the relationship between D_x and D_y is proportional to the relationship between U_s and V_s. Current speed at Portavadie was apportioned between U_s and V_s, with all recordings at ± 45° to main current direction (flow and ebb) apportioned to U_s and the remaining recordings to V_s. The average of the surface and seabed meters for U_s was 0.05m s^{-1} (n = 1404 recordings) and for V_s was 0.025m s^{-1} (n = 741 recordings), a ratio of 2:1. It was therefore possible to apply specific site data to the above model and to compare sites by approximating D_x, the area over which farm derived particulate material settled and to estimate the amount of total carbon and total nitrogen deposited in that area at each site (Appendix 2, Tables A2.9 and A2.10).

The estimated total carbon deposition directly attributable to the experimental cage 8 at Portavadie during February and April 2002 was 32.84 kg C d^{-1}, distributed over a total area of 16,343m^2, giving an average deposition of 2.01 g C m^{-2} d^{-1}. At Rubha Stillaig (cage 11) a depositional area of 10,053m^2 received 9.13 kg C d^{-1} giving an average deposition substantially lower than deposited at Portavadie (0.91 g C m^{-2} d^{-1}) during the same periods.

Cages could not be considered in isolation, however, as the depositional areas of each cage overlap, reducing the farm scale area of deposition relative to the area that would have been covered by the sum of the individual cages. The 2 x 6 cages that form Portavadie (40m between cages in a row and 48m between rows) covered an area of 12,205 m^2 giving an estimated total carbon depositional area of 47,595m^2. If deposition was assumed to be equal from all cages (32.84 kg d^{-1} $cage^{-1}$) then the average rate of deposition increased to 8.3 g C m^{-2} d^{-1} derived from the farm. At Rubha Stillaig the 2 x 10 cages covered an area of 21,002 m^2 giving an estimated depositional area of 52,276 m^2. Again, if deposition was assumed to be equal from all cages (9.13 kg d^{-1} $cage^{-1}$) then the average rate of deposition increased to 3.49 g C m^{-2} d^{-1} derived from the Rubha Stillaig site, being approximately half the rate of deposition at the Portavadie site.

Although the non-standardized carbon regression curves did not significantly differ at the sites, the mean faecal sedimentation rate at Rubha Stillaig was slightly lower at all stations compared to Portavadie. The additional 11m (51m compared to 40m) in the estimated maximum deposition distance (D_y) increased the ratio of cage area to depositional area to 3.9:1 at Portavadie from 2.5:1 at Rubha Stillaig. The combination of these two elements using the Gillibrand *et al* (2002) method suggests that substantially higher quantities of carbon will settle onto the seabed when using the adaptive feeding system (at Portavadie) compared to hand feeding (at Rubha Stillaig).

At an average 32.84 kg C d^{-1}, 492.6 kg of carbon was added to the sediment during each 15-day trial period in February and April 2002 at Portavadie. During each period the estimated average biomass increase was 2,760 kg (Table 4.4) giving an estimated average faecal particulate carbon waste of 178.5 kg C per tonne of production. Using mass balance calculations, faeces represented 31.0 % of the feed carbon input, assuming an average feed input of 220.4 kg d^{-1}, a carbon content of 49.5 % and feed losses of 3 %.

At Rubha Stillaig, applying the same criteria as above, an average 137 kg of carbon (9.13 kg C d^{-1} x 15) was added to the sediment during each trial period in February and April 2002 for an average biomass increase was 2,145 kg (Table 4.4), giving an average faecal carbon waste of 63.9 kg C per tonne of production. As the carbon settlement was derived from fish faeces only then faeces represented 7.4 % of the feed carbon input at Rubha Stillaig, assuming an average feed input of 257 kg d^{-1}, a carbon content of 49.5 % and 3 % direct feed losses.

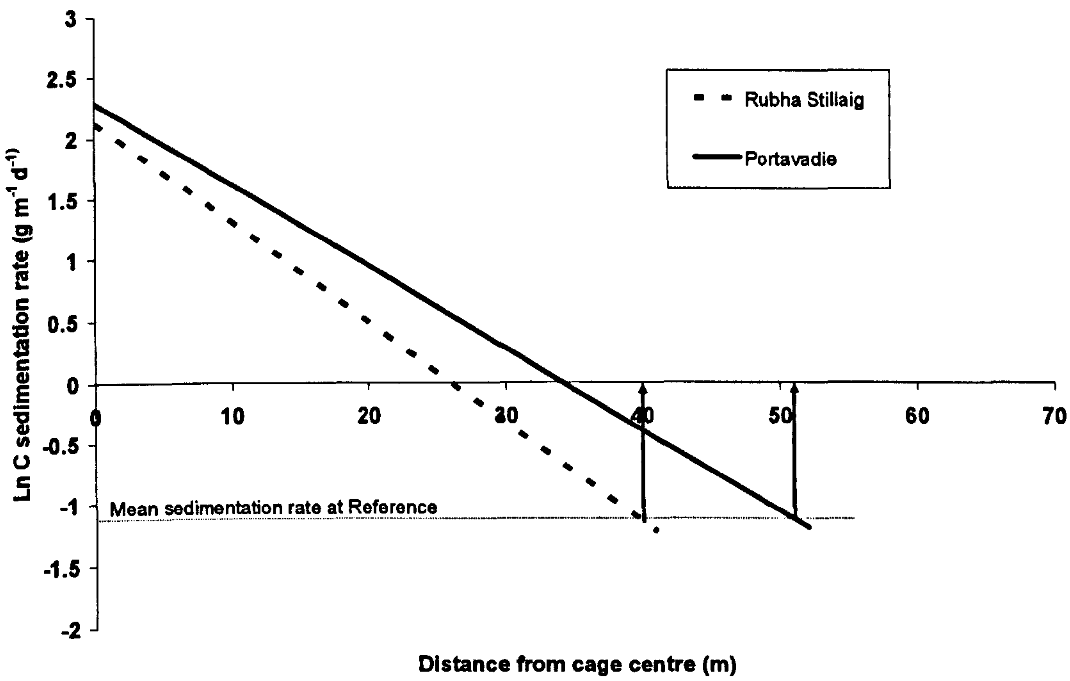


Figure 4.22: Estimated limits of fish farm waste (carbon) deposition in the cross-current direction at Portavadie and Rubha Stillaig based on mean (February 2002 and April 2002) natural log linear reduction of carbon sedimentation rate back to measured mean reference levels.

4.5 Discussion

This study, using sediment traps to assess the output of fish farm waste material to the environment, was substantially larger than other recent similar studies (Cromey *et al*, 2002; Kempf *et al*, 2002). Particulate material was collected at two sites, on 3 separate occasions through the growing season, spanning 6 weeks at each site. Sediment trap studies are conducted for a number of reasons, from estimating levels of deposition (Banse, 1990; Michaels *et al*, 1990) to collection of data for model validation (Cromey *et al*, 2002). This study was the only one of its kind, comparing the waste particulates emanating from Atlantic salmon cages that utilised different feeding methods, with the aim of testing the (null) hypothesis that utilizing adaptive feeding systems result in similar quantities of waste being deposited on the environment to that deposited under hand feeding methods.

A number of researchers have used sediment traps to capture feed and faecal waste material at fish farms (Huntingford, 2001; Kempf *et al*, 2002) and this methodology was the basis of the experimental design. Whilst collection of faecal pellets and faecal debris was successful in this study, this was not so for feed pellets. Portavadie and Rubha Stillaig were both well-run fish farms and collection of feed pellets on only three occasions could be attributable to a high standard of husbandry at each site. It was more likely that feed was under-collected, however, perhaps due to the small surface area of the traps. The high settling velocity of feed pellets (Chen, 1999[a]) in turbulent coastal waters may have biased the traps (Bale, 1998; Buesseler *et al*, 2000) towards the collection of slower faecal pellets, although any bias was acting on both sites. There appears to be a general difficulty in using traps to collect food particles, where an analysis of the data collected by Cromey *et al* (2002) and Kempf *et al* (2002) showed the quantities of feed collected was low and insufficient to assess the quantity of food entering the environment directly. Such difficulty is also shown by the wide variations of feed loss (1 – 15%) reported in the literature (Blyth *et al*, 1993; Findlay and Watling, 1994; Beveridge *et al*, 1997; Cho and Bureau, 1997). Further work in this area is needed, where feed deposition might be assessed using tarpaulins to capture all food falling through cages and using pumps to recover material for analysis.

More generally, deployment and final positioning was outside specific control, being affected at the time of deployment by the weather and tidal currents once the traps were below the surface. Variations in positioning may have contributed to the wide variation in data reported here. Also variations in the spatial and temporal release and distribution of the waste itself would have increased variability in particulate material collected in sediment traps. The use of divers to deploy traps would not necessarily have reduced this variation as cages would also move on the changing tide, altering the position of traps relative to the cage.

Fish farm cages are typically orientated to the main current direction in order to reduce drag (Beveridge, 1996). Sediment traps were deployed at 90° to respective cage orientations and collected particulates in the cross-current direction. Deployment in this direction was unusual (SEPA, 2001; Kempf *et al*, 2002) but it enabled variations in deposition to be measured over relatively short distances. Whilst deposition may have been under-estimated using the cross-current direction, the calculation of total faecal deposition (section 4.4) takes this into account. On a practical level it avoided interference from other cages at the sites.

There were no significant differences in the quantity of faecal solids (FS) deposited between sites, under the different feeding regimes, at three of the four stations measured, although fish fed using the Adaptive Feeding System had a slightly lower faecal output. Cho and Bureau (1997) suggest the quantity of faeces is a function of the Apparent Digestibility Coefficient (ADC) of the individual feed components. They estimate that 85% of the food is digested, but if a fish is deficient in a particular nutrient then it may eat more to compensate and ADC overall may be reduced as those elements the fish was not deficient in are voided as faeces (Booth et al, 2000). Adaptive feeding systems improve the utilization of food, increasing the quantity that is retained by the fish and faecal output has been shown to be lower using these systems (Blythe *et al*, 1993; Huntingford, 2001). This is generally confirmed by the lower FS found at the adaptive feeding (Portavadie) site during February and April 2002, compared to the hand fed site during the same period.

Studies have shown that the quantity of total solids (TS) being deposited on the seabed around fish farm varies widely depending on the size of the farm. In this study differences between the sites, in terms of fish quantity and biomass, were overcome by standardizing per tonne of growth. Dispersion is also dependant on whether the site is depositional or dispersive subject to the hydrographic regime. Similarity between the current speeds of surface and seabed current meters during this study and a recent study at a Scottish fish farm (Cromey *et al*, 2002) suggests both Portavadie and Rubha Stillaig could be described as "depositional sites". The level of particulate deposition (in g m^{-2} d^{-1}) found at each site was 1.5 to 2 times that reported by Cromey *et al* (2002) for a similar site, but to some degree this reflected the differences in the average amount of feed added to the cages per day. The amount of production (= growth) was not specifically reported by Cromey *et al* (2002), so it was not possible to make a direct comparison, taking differences in biomass or growth into account.

Cromey *et al* (2002) deployed sediment traps under a single cage for two one-day trials, as part of the validation process for DEPOMOD, with collections ranging between 8.6 to 12.4 g TS m^{-2} d^{-1} at a dispersive site and 17.1 to 110 g TS m^{-2} d^{-1} at a depositional site. When "feed" samples were included in this study the upper rate was increased to a maximum 320 g TS m^{-2} d^{-1}, suggesting that the DEPOMOD model might have been validated quite restrictively. In weight terms the level of TS can alter dramatically depending on whether large or small feed pellets are being used. This is reflected in the maximum TS recorded at each site. In February, when the highest TS was recorded at Portavadie, there was a failure in the adaptive feeding sensor that meant that all food planned for that period was added irrespective of fish appetite and is thought to be the reason why feed pellets were collected on this occasion only. The reason why Rubha Stillaig encountered pellets at the collection in February and again in April is less clear, but the most likely explanation relates to the increased exposure of the Rubha Stillaig site combined with poor weather in February and an increased wind on specific days in April that may have resulted in fish feeding poorly.

It was difficult to find recent studies that standardized solids deposition to estimated growth, as was done here. Accounting for the removal of feed pellets from the estimate of solids output, the range of material deposited in sediment traps at Portavadie and Rubha Stillaig was similar to other reported sites. Kupka-Hansen *et al* (1991) observed 4 – 40 g TS m^{-1} d^{-1} directly under 5 Atlantic salmon farms in Norway, but ranges between 0.3 – 181 g TS m^{-2} d^{-1} are reported (Hargrave, 1994 and references cited therein). More recently Kempf *et al* (2002) collected 263.5 g TS m^{-2} d^{-1} in a two-day study at an Atlantic salmon farm off the coast of Cherbourg, where feed pellets were detected in the samples. Many of the above studies were conducted over short collection periods (1 or 2 days), whereas this study has shown that longer studies result in occasional substantially higher deposits being detected. Feed weight, length of deployment and season would be important factors to consider in future studies in this field.

Carbon and nitrogen are known to leach from both feed and faecal pellets. During short-term immersion studies, Chen (2000) suggests that both feed and faecal material can lose up to 26% of the carbon and nitrogen in the first 10 minutes after immersion. This contradicts longer term studies (up to 120hrs) which show that reductions in carbon and nitrogen were similar at <10% (Stewart and Grant, 2002). The ratio of carbon to nitrogen measured in deposited material was higher than measured in feed (Table 5.6) and those described for faecal pellets (Chen, 2000), which suggests that nitrogen losses were higher than those from carbon during this study. Bacteria and swimmers are known to convert particulate organic nitrogen (PON) to dissolved form (DON), similar to that experienced in sediments (Blackburn, 1987; Boyd, 1995).

Leaching can therefore affect the estimated percentages of nutrients in settling particulate material. Leaching cannot explain the higher %TC and %TN found at Portavadie stations (except P_0), however, as both sites would be equally subjected to the loss of nutrients. The minimum percentages suggest that particulates consisted of a high proportion of faecal material, with any variation between sites likely to reflect the natural variability in faecal composition from individual fish (Chen, 2000). Faecal composition in any particular period will also vary depending on the composition of the feed used at the time. Thus the reported differences in %TC, %TN and CN ratio between sites at the outer stations would appear to reflect the variability in the composition of faeces, the feed used, plus the general variability in background deposition; rather than reflecting any real differences in these parameters as a result of using either adaptive or hand feeding.

The calculation of sedimentation rate for both carbon and nitrogen is a function of the amount of solids deposited and their composition. The slightly higher solids deposition but lower nutrient composition at Rubha Stillaig resulted in a lower overall sedimentation rate at the outer stations at this site, shown as significant difference in the slopes of the sedimentation regression curves. The movement of waste particulates through the water column is affected strongly by faecal settling velocity (Chen et al, 1999[b]) that may have been lower at Rubha Stillaig by virtue of the smaller fish size, even though comparison was carried out per tonne of growth. The position of Rubha Stillaig was also more exposed with particulates likely to have been subject to enhanced turbulent mixing and increased scatter from wind driven currents. Thus differences in sedimentation rate between sites could have been caused by subtle differences in hydrography, although failure of current meters at the Rubha Stillaig site means this cannot be confirmed.

The lack of a difference with the interaction of time and distance at Portavadie using the General Linear Model (Appendix 2, Table A2.6) suggests there was no difference in the sedimentation rate per tonne of production within this site, although faecal deposition was shown to reduce slightly with increased fish size. Relatively low FCR and high growth during September at the Rubha Stillaig site, combined with good weather and higher sea temperatures, may account for the lower settlement per tonne seen and the resultant difference in sedimentation rate at Rubha Stillaig. Atlantic salmon are known to have a lower relative metabolic rate as they increase in size (Bone *et al*, 1995) and this reduction may account for the fact that significant increases in faecal output per tonne of production were not detected over the growing season. Such data are useful for incorporation into computer models where many models (e.g. Cromey *et al*, 2002; Perez *et al*, 2002) assume that the rate of food input and feed and faecal release per tonne of production to the environment are uniform across the production period. The data presented here suggests it is reasonable to assume that faecal output per tonne of production do not alter over the growing period.

There is a theoretical limit to the distance to which particulate fish farm waste will settle onto the seabed. In simple terms the position that a feed or faecal pellet finally hits the seabed is a function of its settling velocity, water depth and current speed (Gowen, 1988) and that this could also be applied to estimate depositional area (Gillibrand *et al*, 2002). Although applied at site level in the Gillibrand *et al* (2002) report the deposition aspects of their method was used to estimate deposition from single cages at each site. As sedimentation rate differed between sites at the outer stations, the longer deposition distance (D_y) at Portavadie appeared to indicate a real difference under the 2 feeding regimes. This is best illustrated by the ratio of deposition area to cage area, being 2.48:1 under hand feeding at Rubha Stillaig and 3.89:1 at Portavadie using the adaptive feeding system; and the large difference in the estimated total deposition of carbon. The Gillibrand *et al* (2002) model is a simple design, with large assumptions on the use and division of hydrography between D_x and D_y and on the characteristics of settlement. There is no published literature that applies specific deposition data to the Gillibrand *et al* (2002) model, but this study suggests the method is an over-simplification of likely deposition at a fish farm. Specifically the total deposition calculated (in Kg t^{-1}) would appear to be significantly below levels reported by other authors (Hall *et al*, 1990). Taking account of latest husbandry techniques and changes to faeces as a result of feed composition (Storebakken *et al*, 1998, 2000), that are likely to have reduced the outputs per tonne of production since 1990, the 63.9 kg t^{-1} registered at Rubha Stillaig would appear to be far too low to be a realistic estimate.

In this study the limit of deposition along the respective transects was estimated to be an average 51m at Portavadie and 40m at Rubha Stillaig. These figures are broadly similar to other studies (Hall *et al*,1990, and references cited therein). Both Weston (1990) and Johannessen *et al*, (1994) noted effects at increased distances, although their studies were assessing biological and chemical changes to the seabed rather than carbon and nitrogen deposition *per se*. The estimated maximum distance did not take into account any subsequent movement that may take place once the particulates had reached the seabed, as a result of saltation (Chen, 2000) or re-suspension and re-settlement (Stewart and Grant, 2002). Fundamentally, sediment traps represent an artificial "seabed" that cannot physically be subject to such post-depositional movement. It is important to note that post-depositional movement does not increase the estimated deposition per tonne of production, but does act to distribute it more widely and increase the maximal distance affected by farm wastes.

Particulate settlement may also have been affected by the movement of the cages as the ebb and flow of the tide altered the relative position of the cages in relation to the sediment traps. The surface line, used to position the sediment traps relative to the cage edge, was pulled taught prior to each deployment but was observed to be slack on a number of the subsequent collections, the cage having moved nearer to the 5m deployment position. For example, excluding the feed pellets deposited in February, mean FS deposition was higher at the 5m station than under the cage at Portavadie and was thought to be due to the movement of the cage relative to the 2 stations. Cromey *et al* (2002) noted that movement of cages on moorings were unidentified quantities that may affect the deposition of particulate material. Such unknowns are likely to increase the variability in sediment trap data collection, as was seen in this study, and may have contributed to the general similarity in deposition parameters between the two feeding systems.

Overall, the deposition under the cages (at P_0 and R_0) was the most important station within the transect, because the highest deposition occurred at this station. At the 0m stations, %TC, %TN and CN ratio were similar and there was no observed difference in sedimentation rates. At all stations within the respective transects there was no difference in faecal solids deposition per tonne of growth. Combining these results with the overall variability in measured data, brought about by variations in exposure and hydrography, strongly suggests that there was no overall difference in the quantity and composition of waste particulate material emanating from fish cages under the two feeding regimes.

The apparent similarity between the two sites could result in rejection of the (null) hypothesis that there is no significant difference in the quantity and composition of material deposited on to the seabed, under each feeding system. The length of the on-growing phase under the respective feeding regimes also has to be considered in the evaluation, however. Management decisions during the course of this study resulted in neither set of fish studied spending all of their time at a single site under a single type of feeding regime. Taking this into account the information presented in Chapter 2 shows that the on-growing period using the adaptive feeding system was approximately 14 months, compared to 17 months using hand feeding at Rubha Stillaig. Feed input at Rubha Stillaig was completed once or twice per day, feeding for a maximum 1 hour per occasion. Feed input was therefore relatively fast in comparison with Portavadie where the computer system was able to add feed at regular and controlled intervals throughout the day. Bailey *et al* (2003) showed that Atlantic salmon growth was unaffected by feed delivery rate, with the fish being able to adapt quickly to the regularity or irregularity of feed input, but this study suggested that fish seemed to perform a little better when fed using an adaptive feeding system, though not significantly better.

The saving of approximately 3 months in the growth period using the adaptive feeding system has a large implication for the total amount of waste being deposited on the seabed during each growth cycle, which is likely to be significantly higher when using the hand feeding method. The environmental benefit gained from using the adaptive feeding method would only be accrued if the time gained was used constructively by increasing the time that would usually be set aside for fallowing.

Chapter 5

Comparing the effects of nutrient enrichment on the macrofauna under Atlantic salmon farms that use different feeding methods

5.1 Introduction

5.1.1 General Introduction

Short and long-term monitoring of benthic environments is a commonly used method to assess the health of coastal systems (e.g. Manté *et al*, 1995; Burd, 2002) and provides "a useful insight into the functioning of the system" (Thrush *et al*, 1994).

Marine benthic habitats provide a complex inter-relationship between biological, physical and chemical factors (Snelgrove and Butman, 1994) that will vary from location to location. For example, current speed and water depth can have a large effect on grain size, sediment oxygen levels and deposition of particles from the water column that in turn will affect the biological composition of the flora and fauna on and in the sediment. Animals (hereafter referred to as macrofauna) live in functional relationships with these characteristics and in competition for space and food resources. Climax communities in equilibrium are generally species rich, moderate in abundance and high in diversity. Whilst seasonal fluctuations do occur, the equilibrium will not fundamentally change without some disturbance such as enhanced nutrient loading.

An example of a source of nutrient enrichment is the marine cage culture of fish species that deposits particulate material to the seabed in the form of nutrient rich waste feed and faecal material. Recent developments in technology has altered the way in which feed is distributed and computer controlled adaptive feeding systems are being used with increasing regularity. An example of such a system is described in detail in Chapter 2. Whilst the use of this technology is not widespread, a number of studies have established that these systems have the potential to reduce feed waste (Austreng, 1994, cited in Einen *et al*, 1995; Huntingford, 2001). However, thus far no investigations have been carried out that assess the specific implications of using this technology within salmon culture and whether any environmental improvement can be gained by using these systems.

5.1.2 General nutrient enrichment and benthic communities

Increased flux of particulate material to the seabed is a common phenomenon, especially in coastal waters, leading to varying degrees of nutrient enrichment. It is a process that Nixon (1995) termed "benthic eutrophication" because the seabed has the potential to enhance wider eutrophication processes (see Gowen *et al*, 1988; Gowen, 1994) by increasing oxygen demand as bacteria and animals turnover deposited material (see Nilsson and Rosenberg, 1994).

In coastal waters natural spring phytoplankton blooms, and to a lesser extent autumn blooms, cause natural fluctuations in nutrient deposition. In a recent review Grall and Chauvaud (2002) highlighted the effects of this and enrichment from anthropogenic inputs, in terms of benthic processes and consequences for the flora and fauna. Disperse sources of input such as general agricultural and forestry run-off (see Enell and Lof, 1983) increase nutrients over varying spatial and temporal scales. However, sources of general run-off are often difficult to quantify and the effect of point sources of enrichment, such as waste dumping (Pearson and Rosenberg, 1978; Strain *et al*, 1995; Morrisey *et al*, 2003) and aquaculture (Weston, 1990; Perez *et al*, 2002), have been studied in more detail.

Pearson and Rosenburg (1978) provided the first descriptive model of changes in benthic macrofauna along a pollution gradient. They noted a reduction in species diversity and abundance but increases in biomass at high organic loading. This is consistent with large numbers of a few tolerant opportunistic species (or k-strategists) taking advantage of increased space and reduced competition from less tolerant longer lived (r-strategist) species (Thrush *et al*, 1994, Grall and Chauvaud, 2002). At the highest organic loading the descriptive model identifies an afaunal zone where conditions become intolerable for even the hardiest of animal species (Ferraro *et al*, 1991). In addition Weston (1990) noted a general decrease in body size as the level of impact increases, the smaller size increasing the relative surface area to cope with reduced oxygen concentrations, for example. Fauna also occupy shallower positions in the sediment, resulting from lower oxygen penetration in soft sediment (Hall, 1990) and other chemical processes (Weston, 1990). There is also a shift in trophic status towards deposit feeding.

Whilst the relationship between macrofauna and sediments is not only a function of nutrient enrichment (e.g. Snelgrove and Butman, 1994) investigations of species richness, diversity and abundance of macrofauna provide very good data to assess the effects of nutrient enhancement (Hargrave and Thiel, 1983). Such investigations are a typical method used when carrying out an Environmental Impact Assessment (EIA) and for regular monitoring strategies (e.g. Wildish *et al*, 2001). Sampling of macrofauna requires time and skill; for collection, processing and identification; but it highlights impacts that are not detected by chemical and physical characteristics alone (Gowen *et al*, 1991), thus they are an important technique in the on-going assessment of the effects of fish farm wastes on benthic processes and fauna.

5.1.3 Sample Collection Equipment

There is a range of equipment that can be used to quantify benthic communities in soft sediment (see Murdoch and MacKnight, 1994, for a review). Where differentiation with sediment depth is required (Gray, 1982) corers are the preferred tools. These may be deployed in a general area from a surface vessel or at specific points using divers. Where differentiation with sediment depth is not required then grabs may be used.

Examples of grabbing equipment include the Ekman, Van Veen, Smith-McIntyre and Petersen grabs, each available in different sizes from 0.01m^2 to 1m^2, 0.1m^2 being a typical size used. Over and above the experimental design or specific protocol being used size is dependant on the lifting equipment and research vessel available. For example, a 0.025m^2 Van Veen grab is readily lifted to the surface by hand whereas a 0.1m^2 grab can require 400kg of lifting capacity (Murdoch and MacKnight, 1994) and would need a heavy lifting device such as a winch. Depth penetration and thus volume of sediment collected will vary depending on sediment type and compactness, although weights can be added if necessary. However, nearly all benthic sampling is traditionally analysed on the basis of abundance per m^2 surface-area rather than a volume measurement (Grave *et al*, 2001).

In a comparison of 4 sampling equipments to collect sediment, Sommerfield and Clarke (1997) showed that analysis of the macrofauna using univariate measures, such as Shannon-Weiner Index and Evenness did not vary significantly across the methods. However, detailed analysis using multivariate techniques did highlight differences in overall community structure.

In macrofaunal studies there is no uniformity in the number of grab samples taken, although methods for optimising sample size (Bros and Cowell, 1987) and industry specific guidelines are common (CEFAS, 1998; Gillibrand *et al*, 2002). Whilst under-collection may result in under-estimation of the overall species density and community structure, over-collection may result in similar data. Skilleter (1996) showed that repeated measures in the same location may result in an artefact of that measurement, where a gradual reduction in species diversity is not necessarily the result of fundamental changes in habitat but simply resulting from the high grab frequency and the physical disturbance caused by using the equipment. Both cores and grabs remove sediment and the resulting re-distribution of the remaining sediment can influence biodiversity in that area over the short term, so sampling design needs to reflect this. Sampling strategy is also important in all studies requiring a balance between effort required, cost and precision (Bros and Cowell, 1987).

5.1.4 Sample Processing

After collection using grabs or corers the macrofauna is separated from the sediment using sieves. While many off-shore surveys may use 5 - 10mm sieve sizes (e.g. Calloway *et al*, 2002) it is generally accepted that macrofauna can be described as all animals retained on a 1mm sieve mesh size, 500µm if juveniles are included (Holme and MacIntyre, 1984; Wolff *et al*, 1987). Studies have shown that analysis of univariate indices and multivariate methods did not show a significant difference in spatial patterns detected using either a 1mm or 500μm sieve size (James *et al*, 1995; Thompson *et al*, 2003).

Bachelet (1990) highlighted that all studies are a compromise between resolution to identify specific trends and the high costs associated with processing and analysis when a smaller sieve size is used. Resolution may be achieved through an increase in the number of samples taken that is afforded by the reduced effort when using a larger sieve size. However, in a detailed study of *Corophium* species Crewe *et al* (2001) showed that detailed size distributions and densities could only be discerned on a sieve size of 250μm. This highlights the need to make a judgement on the specific outcomes expected from the study.

The many samples collected in a single study may or may not be sieved on site. All samples, however, are normally fixed in a 4% formosaline solution (10% formaldehyde) for a minimum 3-4 days before processing and preserved in 70% ethanol back at the laboratory. Stains, such as Rose Bengal, are used at the preference of the researcher but are thought to enhance taxonomic features for identification purposes at concentrations of 4g l^{-1} of 40% formaldehyde (Hartley *et al*, 1987).

Identification of the macrofauna collected is tailored to the level required by the researcher. In the main this is done to species level (Weston, 1990; Krönke and Rachor, 1992; Johannessen *et al*, 1994; Thrush *et al*, 1994; Karakassiss *et al*, 1999). However, several workers have suggested that spatial and temporal patterns may be discerned at higher taxonomic levels (Warwick, 1988; 1993; Krassulya, 2001; Jong-Geel *et al*, 2003), which also has the potential benefit of reducing the costs associated with the analysis.

Faunal studies are not done in isolation and collections for carbon, nitrogen and phosphorous analysis, redox potential, particle size analysis, oxygen depletion and sulphide are also carried out to provide insight into the functional relationships between macrofauna and their habitat (Nilsson and Rosenberg, 1994; Findlay and

Watling, 1997; Hall *et al*, 1990; 1992; MacDougall and Black, 1999; Domínguez *et al*, 2001).

5.1.5 Macrobenthic studies at fish farms.

The majority of the reported macrofaunal investigations around fish farms have taken place in temperate waters at Atlantic salmon farms (Gowen and Bradbury, 1987; Brown *et al*, 1987; Weston, 1990; Kraufvelin *et al*, 2001; Kempf *et al*, 2002). Environmental concerns in developing countries are less of a societal priority (Boyd, 2003) but other species and environments are increasingly being investigated (Tsutsumi *et al*, 1991; Karakassis *et al*, 1997). When an improved localised environment can be shown to have a financial gain for the fish farmer, through reduced mortality from poor water quality, there is an increasing willingness to incorporate changes in husbandry and management practice (Barton, 1997; Carroll *et al*, 2003). This will also contribute to the environmental sustainability of the aquaculture industry (Wu, 1995; Newkirk, 1996).

Whilst the consequences of fish farm wastes are generally understood (e.g. Weston, 1990), there has been a limited number of comprehensive assessments of faunal changes before, during and after fish farm operations, in the form of a Before-After-Control-Impact (BACI) survey (Green, 1979; Underwood, 1991). Hargrave and Thiel (1983) noted that true pollution-induced changes can only be highlighted with knowledge of successional changes at undisturbed sites. However, our knowledge of the state of localised environments prior to the commencement of fish farming is limited.

In Scotland full Environmental Impact Assessments are now a normal requirement of licence applications for marine fish farms greater than 100 tonnes (See Thompson *et al*, 1995; Henderson and Davis, 2000). The Scottish Environment Protection Agency (SEPA) has monitored fish farm sites since production began in earnest during the late 1970's and presently require a biennial monitoring programme, at peak biomass, for every fish farm site in the country. There is therefore a large database of information but much of it is confidential between the fish farmer and the regulating authority and is not available publicly.

Nutrient enrichment of the sediment at fish farms is caused by the deposition of waste feed and faecal material (Beveridge *et al*, 1991) and levels vary over temporal and spatial scales. As fish grow the relative amount of faeces produced and feed added reduces with increased fish size as metabolic rate reduces (Bergheim *et al*, 1984). Smaller fish eat smaller feed pellets and produce smaller faecal pellets that can be deposited further from the cage, depending on the hydrographic regime.

Deposition around cages (Perez *et al*, 2002) reduces with distance and results in zonation of macrobenthic species similar to the Pearson and Rosenberg (1978) model. Under cages, where nutrient deposition is at its highest, the transition between anoxic and oxic sediments may occur directly at the sediment surface and *Beggiotoa* spp. may form white bacterial mats in what is otherwise an area devoid of fauna. Brown *et al* (1987) called this area the azoic zone. At increased distances from the cage block further zones are apparent, which Henderson and Ross (1995) describe as gross impact, heavy impact, moderate impact and non-impacted. They analysed data from a number of farm sites in Scotland and while all data did not necessarily agree, with distinct zones at some sites but not at others, some general patterns are proposed as shown in table 5.1.

Table 5.1: Levels of impact, based on the deposition of particulate waste from eight fish farms in different loch systems on the west coast of Scotland, and the effect on univariate measures of macrobenthic populations (data adapted from Henderson and Ross, 1995).

Level of Impact	Species Richness	Diversity	Biomass	Shannon-Weiner Index
Gross	<5	Low	High	0 – 1.7
Heavy	Improved	Improved	High	1.7 – 3.0
Moderate	< background levels	Moderate	Moderate	>3.1
None	High	High	Moderate	>4

Henderson and Ross (1995) concentrate on univariate measures of species richness, biodiversity, biomass and the Shannon-Weiner index (H') to differentiate zonation along a pollution gradient around fish farms. As a measure of biodiversity a Shannon-Weiner Index of 4.5 is indicative of unpolluted, unstressed benthic habitats (Frontier and Pichod-Viale, 1991, cited in Kempf *et al*, 2002), except perhaps in transitional waters. Such indicators have proved successful in identifying zones of impact (Weston, 1990).

It is difficult to ascribe actual distances to these zones because the area affected by sedimentation and nutrient enrichment will vary with water depth and hydrography. However, studies have shown that the effects can be regarded as localised, with no measured effects 250m from a farm in very deep water (75 - 110m) (Johannessen *et al*, 1994) and up to 40m being more typical (Brown *et al*, 1987; Lumb, 1989; Henderson *et al*, 1997).

Johannessen *et al* (1994) studied a farm in Norway and showed a general decline in species numbers close to the farm, from 65 (before) to 11 (during) and a subsequent slight increase (29 species) after closure of the farm. Species specific changes in abundance and diversity will vary from location to location. However, Tsutsumi *et al* (1991) showed that an increased abundance of Polychaeta species is a characteristic phenomenon around fish farms and a number of workers have noted that polychaetes dominate sediments at short distances from the cage block (Brown *et al*, 1987; Henderson *et al*, 1997; Karikassis *et al*, 1999). Of these *Capitella* spp. is a typical example of an opportunistic species taking advantage of increased nutrient enrichment and because of the often high density of these organisms they are very important in mineralization processes (Tsutsumi *et al*, 1991; Heilskov and Holmer, 2001) and has a worldwide distribution (eg Weston, 1990; Tsutsumi *et al*, 1990).

5.1.6 Aims of this Study

To date no published literature has assessed the environmental consequences of using adaptive feeding technology in salmon culture, in terms of the potential to reduce the impact on benthic species populations and physio-chemical parameters. The aim of this study is to assess whether the species composition and diversity, and nutrient composition (carbon and nitrogen) of sediment beneath cages at a farm that uses adaptive feeding is different to that found on the seabed at a hand-fed site.

The objectives of this study are as follows:

1) To analyze physio-chemical parameters at two commercial fish farms, one (Portavadie) that uses an adaptive feeding system to feed fish and the other (Rubha Stillaig) that uses hand feeding, and

2) To track changes in macrofauna abundance and diversity over the course of a complete 24 month production cycle, and

3) To assess qualitative differences between the two sites using videographic survey, and

4) To clarify differences in macrofauna composition between two fish farms (Portavadie and Rubha Stillaig), to test the null hypothesis that there are no significant differences in sediment characteristics between sites.

5.2 Materials and Methods

The approach used during this study of benthic fauna consisted of analyzing sediment carbon and nitrogen content and particle size, with fauna evaluated through a qualitative assessment of video images and repeated collections of sediment samples for analysis of macrofauna.

5.2.1 CHN analysis

One 0.025m^2 grab sample was collected from each of 8 farm stations plus a reference station for analysis of total carbon and total nitrogen (CN). Samples were collected on a single transect from respective cage edges (see Chapter 3.2) on a bearing of 80° at Portavadie and 30° at Rubha Stillaig as indicated by the arrows in Figure 2.1. Samples were collected at 5m, 15m, 25m and 50m from the cage edge, the distance set by tying rope to the cage and boat and backing off. After being brought to the surface the samples were double bagged with appropriate identification for transportation to the laboratory. No formalin was added. Samples were stored deep frozen until analysis. Analysis of samples was as described in Chapter 3.3. No CN samples were collected in August 2001. No samples were collected at Portavadie in April 2003 due to the completion of fish farming activities, the removal of cages and an inability to maintain an accurate position due to wind and tidal effects.

5.2.2 Particle size analysis

One 0.025m^2 grab sample at each location was collected for sediment particle size analysis (PSA). After being brought to the surface the samples were double bagged with appropriate identification for transportation to the laboratory. No formalin was added. Samples were stored deep frozen until analysis. Analysis of samples is as described in Chapter 3.4. No PSA samples were collected in August 2001. Also, no samples were collected at Portavadie in April 2003 due to the completion of fish farming activities, the removal of cages and an inability to maintain an accurate position due to wind and tidal effects.

5.2.3 Videographic survey

A single videographic survey, by diver held camera, was conducted in October 2002 to provide qualitative data on sediment characteristics and benthic species composition. The dive team consisted of 5 personnel; one diver in the water, one on standby, one boat handler, one video operator and an overall dive coordinator.

The video was taken on transects from cage 8 at Portavadie and from cage 11 at Rubha Stillaig and at a Reference Site (Figure 2.1). At the two cages a concrete block with attached weighted ground-line was positioned at the centre of the cage by diver. The transect extended out to 55m beyond the cage edge in the same direction as the benthic sample collections (this Chapter) and the Sediment Trap Study (Chapter 4), that is 80° at Portavadie and 30° at Rubha Stillaig. The ground-line was marked at the cage edge and then at 5m intervals along its length. The reference site for the video survey was limited in distance from the cages by the need to have a solid mooring for the boat. The reference site was thus approximately 400m east of Rubha Stillaig. A 50m weighted and marked (every 5m) ground-line with concrete weights and riser buoy at each end was positioned parallel to the shore and in line with the cage system orientation.

Video was recorded using a Submatec Seaspy Camera Control Unit aboard the boat and a head mounted Osprey camera with the diver. Light was provided by 2 head mounted 100 watt 240V torches. Air and communication was also provided from the surface.

Each dive consisted of 2 passes along the transect line, starting from the cage centre heading away from the cages and a return to the cage centre. The diver passed along the transect line approximately 1m off the seabed, giving a video-view of approximately 1m across. Speed was controlled by the author from the surface and the length of time available governed by dive computer. The length of each video was therefore 19 minutes at Portavadie, 11.5 minutes at Rubha Stillaig and 14 minutes at the reference site.

5.2.4 Benthic fauna - preliminary sample collection (August 2001)

5.2.4.1 Preliminary study collection procedure (August 2001)

Preliminary samples for the analysis of benthic fauna and establishment of a manageable sample size for the main study were collected from Portavadie and Rubha Stillaig sites in August 2001. Positions were as described in Chapter 3.2., and samples were collected on a single transect commensurate with PSA samples (section 4.2.1). No samples were collected from under the cages due to the cost of employing divers. Water depth at each site was 27m and 30m at Portavadie and Rubha Stillaig respectively.

Samples for benthic analysis were collected using a 0.025m^2 Van Veen grab with top opening panels, 10 replicates at each site (80 samples in total), with all samples collected by hand from a small boat provided by the fish farm. Grab samples that had stones or shells in the jaws of the grab and resulted in it not closing completely, with subsequent loss of sediment during the lifting process, were rejected. Grabs were continued at the same location until the required 10 replicates were achieved.

Once on board the boat each sample was placed into a polythene bag and 40% buffered formalin was added, diluted to 4% in seawater. For transportation purposes all samples were double bagged with appropriate identification inside and outside the bag. Samples were then transferred back to the laboratory to await analysis. Only samples collected at Portavadie were analyzed to assess the number of grab samples required for the main study.

5.2.4.2 Preliminary study post collection identification (August 2001)

Benthic grabs remained in buffered formalin for up to 12 weeks after collection. Samples were sieved using a 1mm-mesh size, rinsing through with fresh water. After washing into white plastic sorting trays samples were processed by hand, under a desktop magnifier and illuminator if required, and stored in 70% alcohol.

No stains were used. The identification number ascribed to each of the replicate grabs occurred randomly in the order they were processed and did not represent the order of collection. Samples were subject to quality assurance procedures that entailed re-picking previously processed samples to determine whether all fauna was selected. No additional specimens were removed from those samples tested.

Samples were identified under a stereo dissecting microscope with a magnification of 10 to 40x and an appropriate light source to clearly identify taxonomic features for both incident and transmitted light. If further magnification was required a compound stereomicroscope with magnification of 100 to 1000x was used. All biota was identified to species level where possible and at least to family level. Where the species had a discernable head and rear end, such as with the Polychaeta, abundance was determined by counting heads only.

The initial identification sources were The Marine Fauna of the British Isles and North-west Europe Volume 1: Introduction and Protozoans to Arthropods and Volume 2: Molluscs to Chordates (Hayward and Ryland, 1990); British Marine Amphipoda: Gammaridae (Lincoln, 1979); Polychaeta: Terebellomorpha (Holthe, 1986); Polychaetes from Scottish Waters: Part 1 Family Polynoidae (Tebble and Chambers, 1982); Echinoderms (Southward and Tyler, 1982) and British Bivalve Seashells (2nd Edition) (Tebble, 1976). Specialist keys and papers were subsequently used to complete the identification. The procedures used in the laboratory during this study were subject to the National Marine Biology AQC scheme (NMBAQC, 2004).

5.2.5 Benthic fauna - main study sample collection (August 2001, April 2002, April 2003)

Samples for analysis of benthic fauna were collected from Portavadie and Rubha Stillaig sites in August 2001, from both sites and a reference site in April 2002 and from Rubha Stillaig and reference sites in April 2003. Only 2 benthic samples were collected from the reference site in August 2001 but these have been included in the analysis. Given the short distances between sample stations and the need to be relatively accurate, no samples were collected from Portavadie in April 2003 due to the completion of fish production, the removal of the fish cages and subsequent difficulty in maintaining an accurate position due to wind and tidal effects. The collection procedure was as described in section 5.2.4.1 except 5 replicates were collected per site (see results 5.3.4), representing a total area of $0.125m^2$. Samples were processed and analyzed as described in section 5.2.4.2.

5.2.6 Statistical analysis

Once identified, the species and their abundance were recorded on data sheets. Species and abundance per grab and per station were incorporated into a benthic analysis package called "Worms", based on DBase 3+ (Borland Software International, California) software, developed by Colin Moore at Heriot-Watt University in Edinburgh. The program ensures easy and consistent data entry and allows species data to be sorted into phylum groups. Data output is in a format that can be entered directly into software to calculate a range of univariate measures including total abundance, Shannon-Weiner Index, Species Richness and Evenness measures (Moore, 1983). Worms follows the nomenclature of Howson (1988), a standard checklist of British fauna.

Data was further analysed using the statistical package Minitab v13 and multivariate statistical package MVSP v3.1 (Kovach Computing Services Ltd, Anglesey, UK). Variation in abundance and taxonomic richness data was analyzed using one-way ANOVA on data that conformed to normality and equality of variance tests (Bartlett's Test) and differences assessed using Tukey's Pairwise Comparison; or the non-parametric Kruskal-Wallis Test; to test for significant differences between the measured stations. Shannon-Weiner Index (Hs) scores were compared within sites using the non-parametric Kruskal-Wallis test on untransformed data. Between-site variations were conducted on Hs values at each location as a proportion of references levels within the appropriate year and were therefore arcsine ($\sqrt{x}$) transformed after standardization.

Two multivariate techniques were used to analyze the data, namely Cluster Analysis and Ordination. Methods of cluster analysis use abundance data to assess similarities or dissimilarities between samples in the form of a matrix with a dendogram output that joins similar data together. Specifically the techniques group together points representing individuals with similar characteristics in mathematical space, in this case grouping data for each collection at each station based on quantitative measures of species number and abundance. It is an iterative process in which passes are made through the data looking for pairs that most resemble each other and fusing this pair with a similarity or dissimilarity distance before repeating the process, until all data is assessed (Kent and Coker, 1992). The sorting method used here was Percent Similarity on Unweight Pair Groups using arithmetic Averages (UPGMA) as recommended in Krebs (1999) for animal community level data. UPGMA (Sokal and Michener, 1958, cited in Kent and Coker, 1992) is the simplest of the sorting methods, weighting the data to provide an equal distance relationship in the dendogram output (that is then regarded as unweighted). Percent Similarity was used as it is relatively unaffected by sample size and species diversity (Krebs, 1999), useful when comparing sites that may have a wide variability in species diversity, such as at fish farms.

Detrended Correspondence Analysis (DECORANA) (Hill, 1979) is an ordination technique that uses trends in species similarity to produce a matrix and scatter-plot, where distances between samples represent similarity defined by the ordination axis. Data for multivariate analysis was Log_{10} (x + 1) transformed to reduce the significance of dominant species (Krebs, 1999). All other transformations are identified in the text.

5.3 Results

5.3.1 Sediment carbon and nitrogen analysis

There were no identifiable feed or faecal pellets within sediment samples collected in April 2002 and April 2003 at either of the fish farm sites. This was reflected in the low percentage of total carbon and total nitrogen found in the sediment at all distances from the two cage sites, in both 2002 and 2003.

At Portavadie in 2002, percentage total carbon and total nitrogen values across the site were higher, closer to the farm cage. Figures 5.1 and 5.2 show there is a high degree of similarity between the P_5 and P_{15} stations and between the P_{25} and P_{50} stations. All stations were higher than total carbon and total nitrogen levels measured at the reference site in 2002, suggesting the fish farm was increasing the deposition of nutrients above what would occur naturally.

At Rubha Stillaig in 2002, stations R_{15}, R_{25} and R_{50} had measured carbon lower than at the reference site that suggests the measurements were within natural sediment variation. It also suggests that any impact the cage may have been having on the seabed, in terms of nutrient deposition, was both low in intensity and restricted to stations closest to the cage edge. This reflected the fact that Rubha Stillaig had been fallowed for 12 months prior to production commencing in December 2001. This was also the reason that it was not realistic to compare Portavadie and Rubha Stillaig data for 2002. However, by 2003 the length of the production period at Rubha Stillaig was similar to that experienced by Portavadie up until 2002 and is reflected in the increased level of nutrients in sediment at the Rubha Stillaig site. Carbon values at Rubha Stillaig ranged between >0.4% and 2.1% with a reducing profile similar to that experienced by Portavadie in 2002 (Figure 5.1).

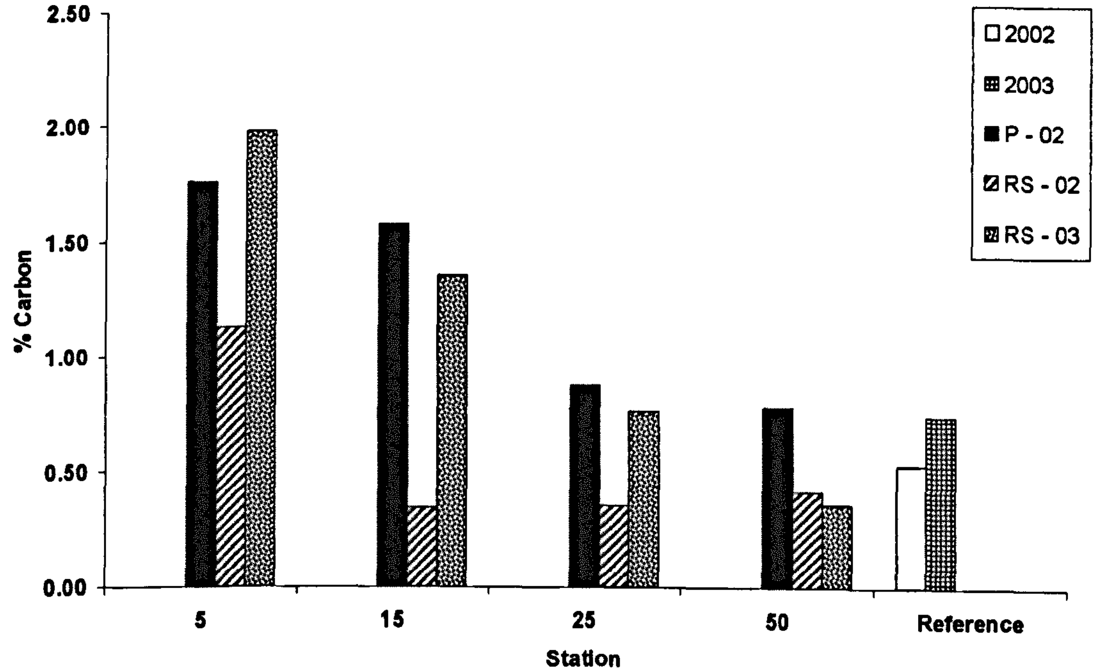


Figure 5.1: Percentage carbon by dry weight in sediment at Portavadie (P), Rubha Stillaig (RS) and reference sites. Samples collected April 2002 and April 2003.

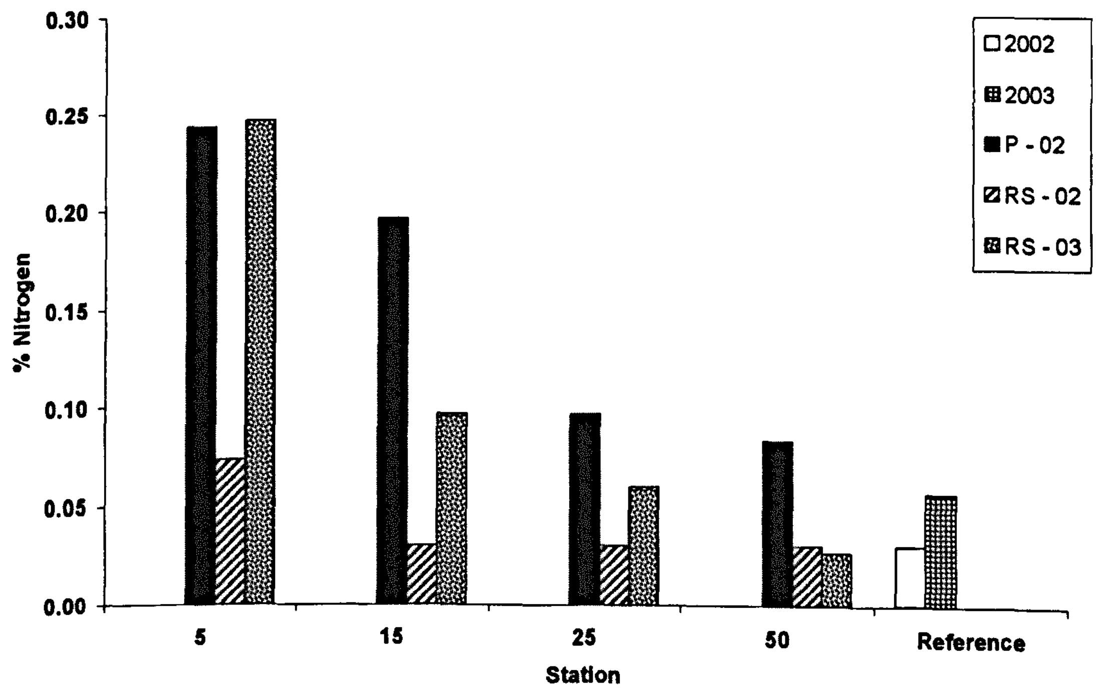


Figure 5.2: Percentage carbon by dry weight in sediment at Portavadie (P), Rubha Stillaig (RS) and reference sites. Samples collected April 2002 and April 2003.

Although carbon levels had increased at R_5 in 2003 the most notable increases occurred at stations R_{15} and R_{25} as the nutrient deposition from the farm increased its range and impact on the seabed. This is also shown in Figure 5.2 where deposition of faeces and possibly feed resulted in increased levels of nitrogen being found in the sediment.

The difference that the year between April 2002 and April 2003 had made to the Rubha Stillaig site is also shown in the CN ratio (Figure 5.3), where the proportion of total carbon to total nitrogen was reduced at R_5 but remained similar at the remaining stations. It was not possible to test statistically any difference in carbon, nitrogen and CN ratio of sediments between sites because a single sample was collected at each site.

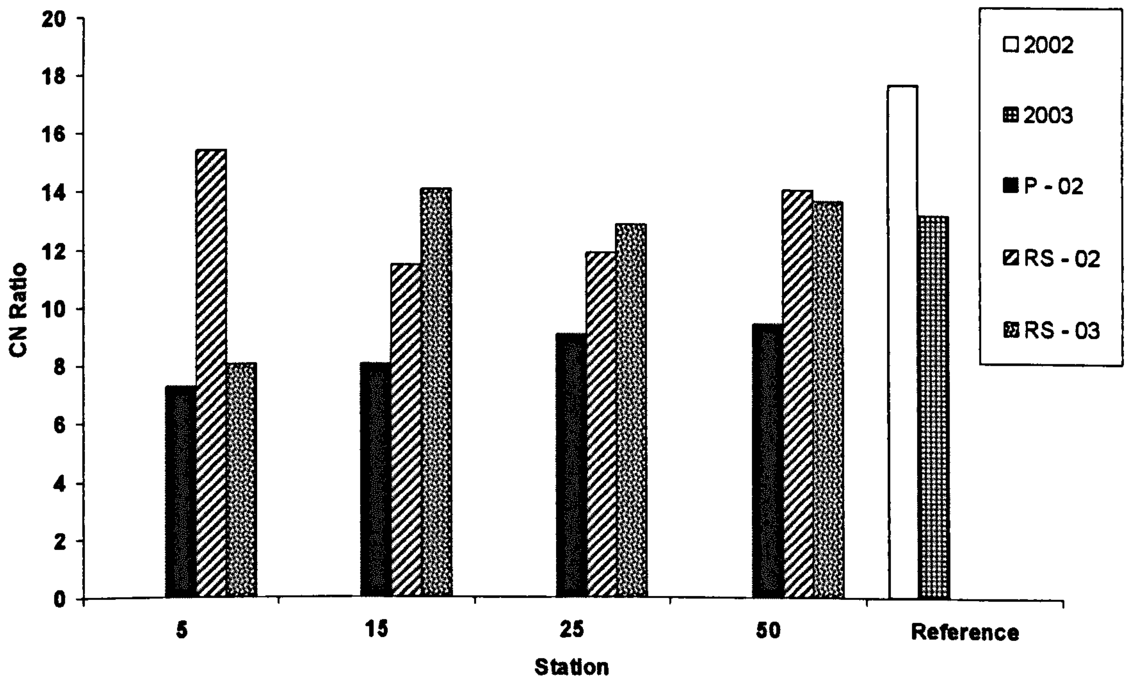


Figure 5.3: Carbon/Nitrogen ratio in sediment at Portavadie (P), Rubha Stillaig (RS) and reference sites. Samples collected April 2002 and April 2003.

5.3.2 Particle size analysis

All stations analyzed, except P_{50} in 2002, were classified, according to the Wentworth Classification of Sediments, as "fine sand" with median grain size between 125µm and 250µm (Phi units between 2 and 3). The slightly lower median grain size at P_{50} was due to the higher proportion of particles in the silt/clay fraction (<63µm, phi >4) at that station (Figure 5.4). At Portavadie the median grain size reduced with increased distance from the cage (Table 5.2), suggesting that the cages at the site were in some way influencing the settlement of particulate material to the seabed, with slightly larger particles settling closer to the cages. This trend was also noted at Rubha Stillaig in 2003, though not in 2002. However, there was no way of determining whether the feeding method at the two sites influenced the sediment grain size, as the slightly increased grain size at P_5 and R_5 were equally likely to result from the siting of nets and cages at the site than to the feeding method.

The negative skewness (Table 5.2) suggests the sediment has more coarse material than fine material. However, the low skewness values and similarity between quartile deviations suggests all stations were well sorted. Rubha Stillaig and Portavadie differed slightly in 2002, primarily the result of a higher proportion of sediment between 1 and 2mm in diameter at the Rubha Stillaig site (Figures 5.4 and 5.5) and identified as shell fragments. Otherwise the distribution of particle size within all sediments in the bay was broadly similar (Figures 5.4, 5.5 and 5.6) and as such was unlikely to influence the macrobenthic community structure at the cage sites over what was occurring naturally at the reference site.

Table 5.2: Particle size parameters for sediments at Portavadie (P), Rubha Stillaig (R) and Reference site in 2002 and Rubha Stillaig and Reference site in 2003, based on the Wentworth Classification Scheme. Subscripts represent distance from cage edge in metres.

2002	Median Phi Score	Quartile Deviation	Skewness	Median Grain Size (μm)	Classification
P_5	2.28	0.690	-0.200	205.8	Fine sand
P_{15}	2.39	0.475	-0.005	190.7	Fine sand
P_{25}	2.56	0.450	0.000	169.5	Fine sand
P_{50}	3.01	0.550	0.000	124.1	Very fine sand
R_5	2.39	0.660	-0.180	190.7	Fine sand
R_{15}	2.52	0.490	-0.020	174.3	Fine sand
R_{25}	2.49	0.450	-0.030	178.0	Fine sand
R_{50}	2.38	0.965	-0.435	192.1	Fine sand
Reference	2.99	0.450	-0.060	125.8	Fine sand
2003					
R_5	2.39	0.460	-0.040	190.7	Fine sand
R_{15}	2.46	0.545	-0.095	181.7	Fine sand
R_{25}	2.49	0.435	-0.015	177.9	Fine sand
R_{50}	2.71	0.415	-0.005	152.8	Fine sand
Reference	2.41	0.530	0.000	188.1	Fine sand

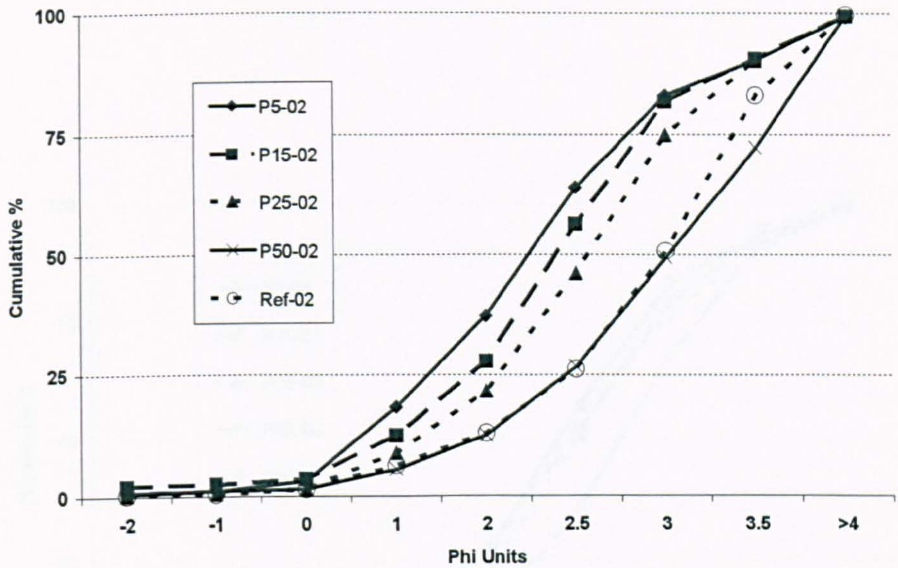


Figure 5.4: Cumulative percentage plot of sediment particle size for samples collected at Portavadie fish farm and reference site in April 2002 by Van Veen grab and analyzed by wet and dry sieving.

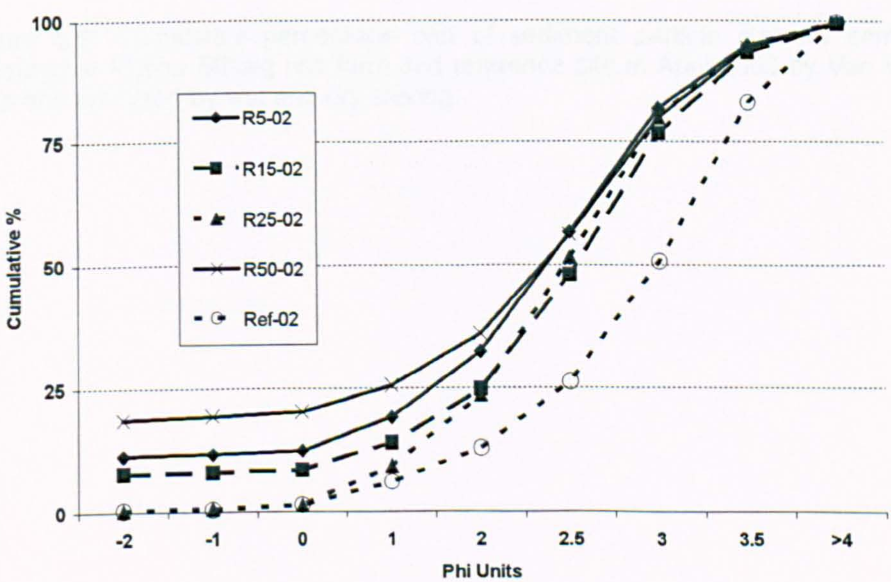


Figure 5.5: Cumulative percentage plot of sediment particle size for samples collected at Rubha Stillaig fish farm and reference site in April 2002 by Van Veen grab and analyzed by wet and dry sieving.

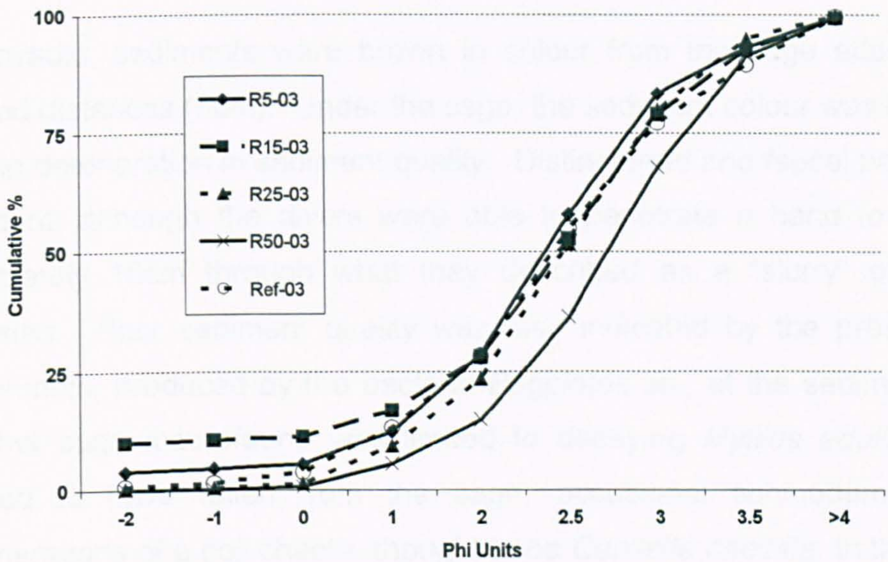


Figure 5.6: Cumulative percentage plot of sediment particle size for samples collected at Rubha Stillaig fish farm and reference site in April 2003 by Van Veen grab and analyzed by wet and dry sieving.

5.3.3 Videographic survey

The video produced in October 2002 (Appendix 3) was analyzed qualitatively to assess the state of the seabed in terms of sediment colour, visible deposition of particulate feed and faecal material, for evidence of macrofauna and general state.

At Portavadie, sediments were brown in colour from the cage edge out to all measured distances (50m). Under the cage, the sediment colour was brown/black indicating deterioration in sediment quality. Distinct feed and faecal particles were not evident, although the divers were able to penetrate a hand to a depth of approximately 10cm through what they described as a "slurry" of degrading particulates. Poor sediment quality was also indicated by the presence white bacterial mats, produced by the bacteria *Beggiotoa* sp., at the sediment surface. Under the cage macrofauna was limited to decaying *Mytilus edulis* that were presumed to have fallen from the cage, occasional echinoderms and red conglomerations of a polychaete, thought to be *Capitella capitata*, in the upper 1 – 2mm of sediment (Plate 5.1). The divers did not observe indigenous fish eating particulate material that had fallen from the cage. At further distances various macrofauna were visible, including portunid and pagurid crabs, benthic fish species (Family: Gobiidae), echinoderms and polychaete worm casts (Plate 5.2). At all distances shell halves and fragments were visible on the sediment surface, although it was not possible to determine whether they emanated from the cages or from within the sediment.

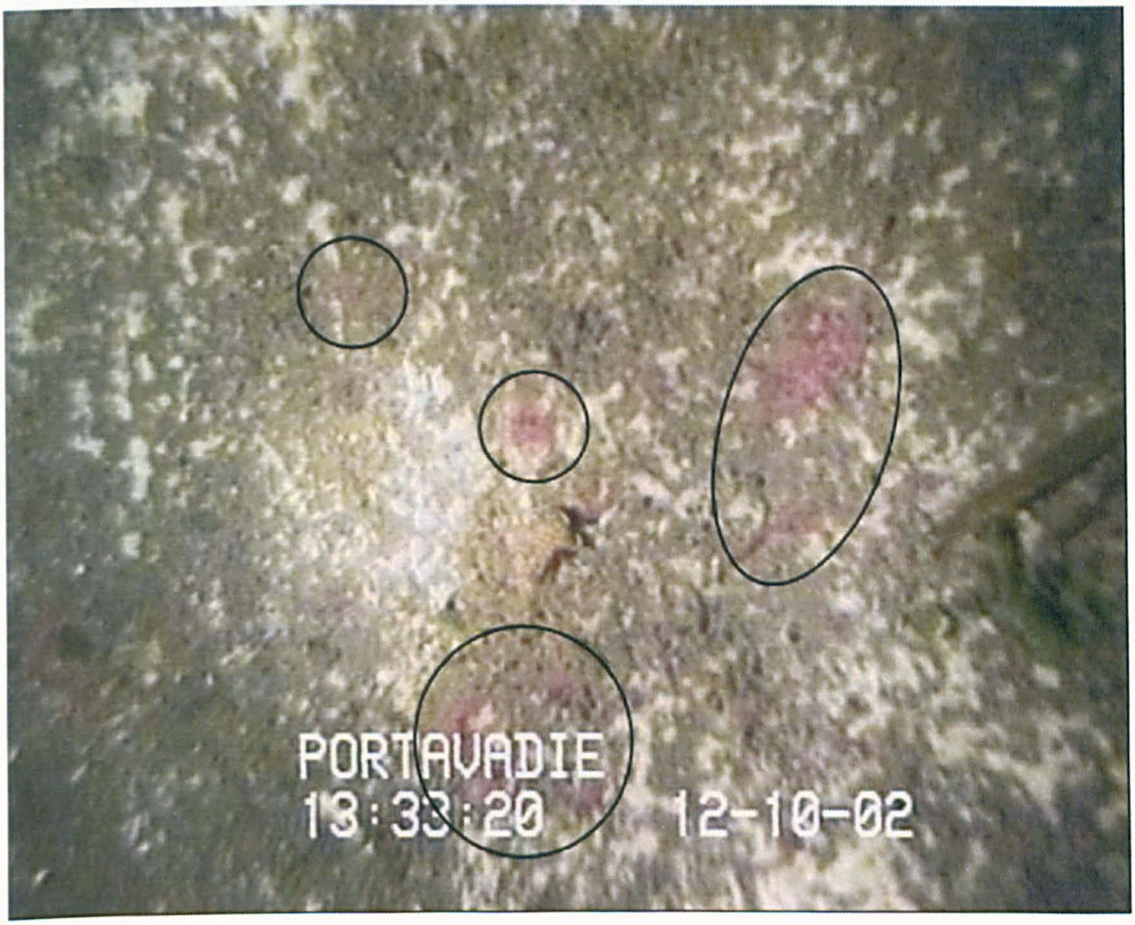


Plate 5.1: Videographic still of seabed below a cage at Portavadie fish farm showing patchiness in the distribution of a polychaete species, thought to be *Capitella capitata* (circled). White patches are bacterial mats of the bacteria genus *Beggiotoa* sp.. Approx. scale: 0.5m across.

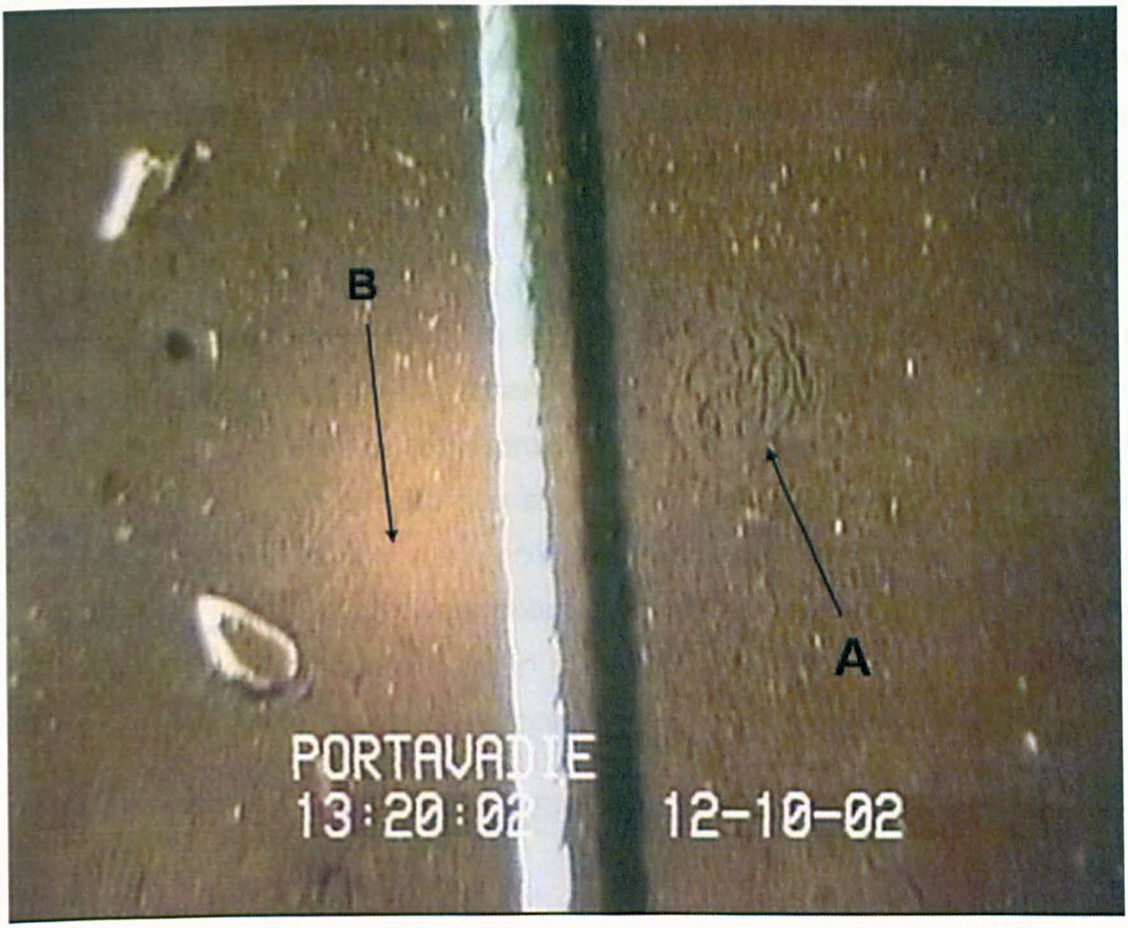


Plate 5.2: Videographic still of seabed, approximately 8m distance from the cage edge at Portavadie fish farm identifying the brown colour of the sediment, the existence of burrowing species (A) and tracks created by an unknown species (B). Approx. scale: 0.5m across.

At Rubha Stillaig the sediment at all distances, including under the cage, were brown in colour until approximately 42m from the cage edge. Here bacterial mats, of the bacteria species *Beggiotoa* sp., were visible (Plate 5.3). This suggested that the present cage position had been moved within the bounds of the leased seabed area but away from the specific location of previous production at the site. Macrofauna numbers were increased at Rubha Stillaig, especially echinoderms, urchins and crabs. The sediment surface sporadically provided a harder substrate, in the form of small rocks and stones that provided a footing for plumose anemones (*Metridium* sp.). There was also a significant quantity of shell halves and fragments.

The reference site provided a typical seabed, containing soft sediments, found in this part of Loch Fyne. The sediment was fine grained and brown along the transect length. Occasional rocks provided shelter for galatheid lobster and a hard substrate for anemone and sea squirt settlement (Plate 5.4). Burrows and worm casts suggested the presence of polychaete species and, possibly, burrowing Amphipoda. The divers noted an abundance of large scallops in the vicinity. Shell fragments were lower in number than at both farm sites, although they were visible on the sediment surface.

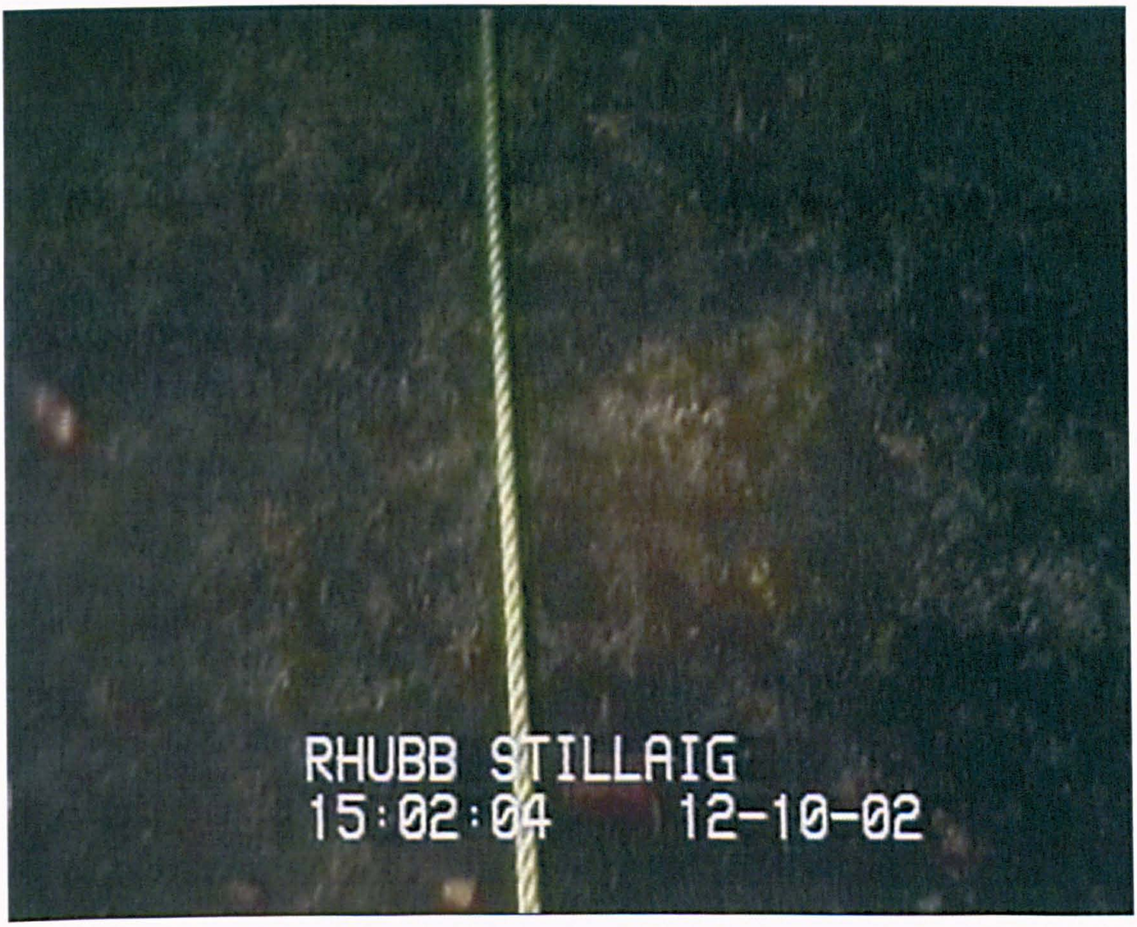


Plate 5.3: Videographic still of seabed, approximately 42m distance from the cage edge at Rubha Stillaig fish farm identifying a large patch of seabed thought to be an area of previous fish farming activity, due to the presence of bacterial mats of the bacteria genus *Beggiatoa* sp.. Approx. scale: 0.5m across.

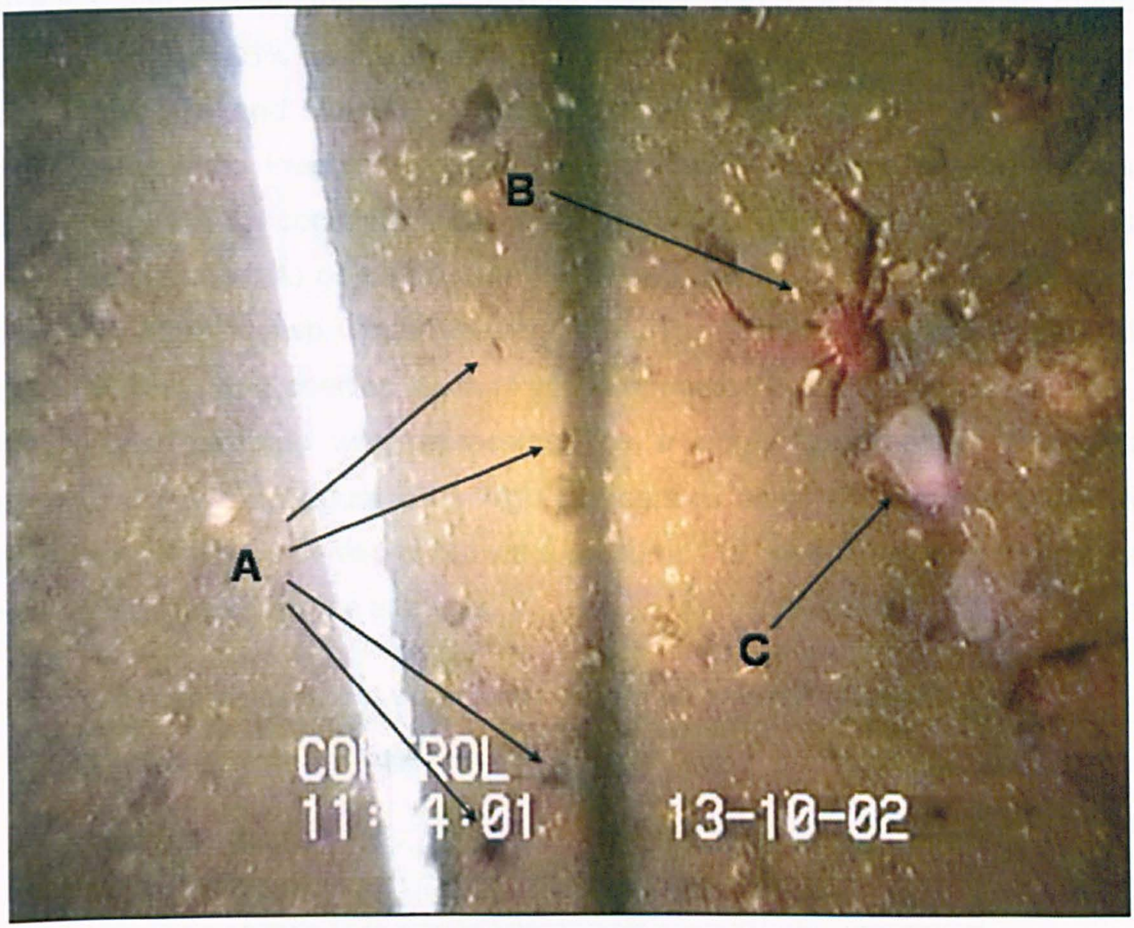


Plate 5.4: Videographic still of seabed at reference station showing brown shelly sediment. (A) burrowing holes of unknown species (B) squat lobster, genus unknown (C) seasquirt, species unknown. Approx. scale: 0.5m across.

5.3.4 Benthic Fauna Preliminary Survey (August 2001).

The macrofauna present in the 40 grabs made at Portavadie in August 2001 were represented by 8 phyla. Four phyla occurred in all four locations, as identified in Table 5.3, and between them accounted for greater than 91% of the species found and greater than 96% of the abundance at each of the stations. Of the lesser phyla, Sipuncula and Nemertea occurred at all stations except P_5, although Sipuncula occurred irregularly across the grab samples and in small numbers (See Appendix 4 for complete list). Whilst an unidentified species of Nemertea (called Nemertea sp.A) occurred evenly between grabs at each station a further species, *Cerebratulus* sp., occurred irregularly. The Cnidaria were represented at location P_5 only, by 2 species, each of which appeared in a separate single grab. The phylum Priapulida was represented by a single specimen of *Priapulus caudatus* at P_{50}. In August 2001 high numbers of small (2-3mm max) *Mytilus edulis* were found at all stations but were presumed to have fallen from the cages and were removed from the analysis.

Table 5.3: Phyla represented at each of the 4 stations sampled at Portavadie fish farm in August 2001, using a $0.025m^2$ Van Veen grab, 10 replicates per station. Subscripts represent distance from cage edge in metres.

P_5	P_{15}	P_{25}	P_{50}
Annelida	Annelida	Annelida	Annelida
Arthropoda	Arthropoda	Arthropoda	Arthropoda
Mollusca	Mollusca	Mollusca	Mollusca
Echinodermata	Echinodermata	Echinodermata	Echinodermata
Cnideria	Nemertea	Nemertea	Nemertea
	Sipuncula	Sipuncula	Sipuncula
			Priapulida

The Annelida (Class: Polychaeta) are an important group within the macrofauna contributing to the bioturbation of fish farm sediments species. The Annelida were represented at all stations and dominated in terms of both number of species and abundance as shown in Table 5.4. At P_{50} Polychaeta accounted for 48% of all the species present but 84.7% of the total abundance. At closer distances to the cage the number of polychaetes remained broadly similar but dominance had increased due to the decline in the total number of species.

Table 5.4: Number of species and Abundance of Annelida collected at 4 stations sampled at Portavadie fish farm in August 2001, using a $0.025m^2$ Van Veen grab, 10 replicates per station. Subscripts represent distance from cage edge in metres.

	P_5	P_{15}	P_{25}	P_{50}
Total Species	23	37	47	48
Annelida species	17	23	24	23
Total Abundance	2228	2584	3382	1487
Annelida Abundance	2176	2324	3140	1260
% Annelida Abundance	97.7	89.9	92.8	84.7

At Portavadie the total number of species in 10 grab samples (representing an area of $0.25m^2$) increased with distance away from the cage as shown in Table 5.4, with the maximum 48 species occurring at a distance of 50m. This is not unexpected as the settlement of waste feed and faecal material from the cage reduces with distance and the sediment quality would therefore be improved here. Total abundance was at its maximum at a distance of 25m from the cage edge, with the number of species not dissimilar from P_{50} and this may be indicative of the macrofauna taking advantage of slightly increased enrichment. The 23 species found at P_5 was indicative of a relatively healthy habitat close to the cage and was above the minimum quantity required in Scotland (<2 species) under regulatory quality standards. However, across the site, the mean number of species per grab was highly variable and this variability between the total number of species and the mean number of species per grab indicates a lack of consistency that may have been due to the presence of the cages.

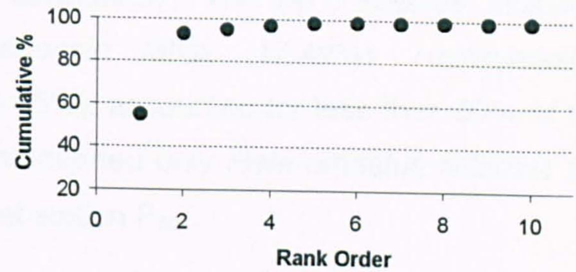
However, the top 10 ranked species for each station ranked according to total abundance (Figure 5.7) shows there was a relative degree of uniformity within station. At P_5 the top 5 species occur regularly, appearing in >8 grab samples, with the remaining 5 species being found in less than half of the grabs. This was not unexpected as sediments close to cages are known to be dominated by a few tolerant opportunistic species. Also the 5 species ranked 6-10 account for less than 1% of the abundance, the sediment being dominated by *Ophryotrocha puerilis* and *Capitella capitata*.

At P_{15} the dominance of *Ophryotrocha puerilis* was increased as the quantity of *Capitella capitata* was reduced. This reduction in *Capitella capitata* may reflect a lowering of the sediment nutrient levels 15m from the cage edge with the increase in *Ophryotrocha puerilis* enabled by reduced competition for space and resources. However, the nutrient-loving but less tolerant species such as *Heteromastus filiformis* and *Abra alba* were found in higher numbers than at P_5. Seven of the top 10 ranked species at P_{15} appear consistently in all of the grab samples taken at that station.

Moderate enrichment can result in increased abundance and diversity of species. At P_{25} that is suggested, as the highest abundance of all stations was found here, although this was primarily because of the high abundance of *Ophryotrocha puerilis* found in all grabs. Also there was high uniformity across the top 10 species with 7 species occurring in all 10 grabs made and the remaining 3 species found in 9 of the 10 grabs.

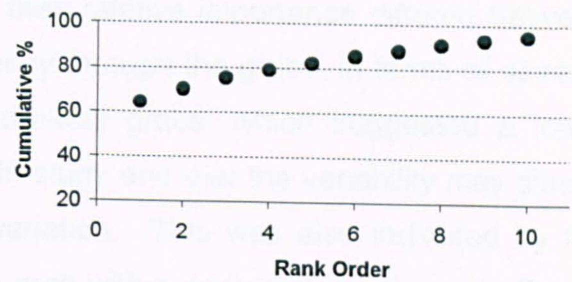
STATION P_5

RANK	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	1223	54.94
2	*Capitella capitata*	850	38.19
3	*Malacoceros fuliginosa*	50	2.25
4	*Abra alba*	41	1.84
5	*Eteone longa*	27	1.21
6	*Platynereis dumerilii*	5	0.22
7	*Thyasira flexuosa*	4	0.18
8	*Anaitides maculata*	3	0.13
9	*Chaetozone sp.*	3	0.13
10	*Heteromastus filiformis*	3	0.13



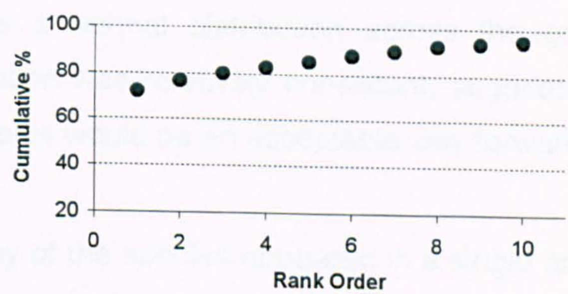
STATION P_{15}

RANK	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	1663	64.38
2	*Heteromastus filiformis*	156	6.04
3	*Abra alba*	126	4.88
4	*Capitella capitata*	97	3.76
5	*Thyasira flexuosa*	82	3.17
6	*Scalibregma inflatum*	81	3.14
7	*Eteone longa*	71	2.75
8	*Anaitides maculata*	71	2.75
9	*Prionospio fallax*	56	2.17
10	*Scoloplos armiger*	50	1.94



STATION P_{25}

RANK	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	2446	72.32
2	*Scalibregma inflatum*	155	4.58
3	*Prionospio fallax*	115	3.4
4	*Heteromastus filiformis*	91	2.69
5	*Thyasira flexuosa*	89	2.63
6	*Eteone longa*	84	2.48
7	*Anaitides maculata*	76	2.25
8	*Scoloplos armiger*	69	2.04
9	*CIRRATULIDAE spp. indet*	42	1.24
10	*NEMERTINI sp. A*	41	1.21



STATION P_{50}

RANK	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	372	25.02
2	*Scalibregma inflatum*	264	17.75
3	*Prionospio fallax*	259	17.42
4	*Heteromastus filiformis*	153	10.29
5	*Eteone longa*	87	5.85
6	*Thyasira flexuosa*	59	3.97
7	*NEMERTINI sp. A*	40	2.69
8	*Owenia fusiformis*	36	2.42
9	*Scoloplos armiger*	31	2.08
10	*Thracia sp. indet.*	21	1.41

Figure 5.7: Rank order and cumulative percentage of top 10 macrofauna species from 4 stations at Portavadie (P) fish farm collected in august 2001. N = abundance in 10 x 0.025m^2 Van Veen grabs, % = percentage of total abundance. Subscripts represent distance from cage edge in metres.

At P_{50} the dominance of *Ophryotrocha puerilis* was considerably reduced, where it accounted for only 25.02% of the total abundance. The top 5 species, including *Scalibregma inflatum* (17.75%), *Prionospio fallax* (17.42%), *Heteromastus filiformis* (10.29%) and *Eteone longa* (5.85%), accounted for less than 80% of the total abundance. Of these 5 species mentioned only *Heteromastus filiformis* did not appear in every grab sample taken at station P_{50}.

Although the species composition and their relative importance differed between stations there was a degree of consistency through the grabs, in terms of species composition and abundance within individual grabs, which suggested a lower number grabs could be used in the main study and that the variability may simply be due to naturally occurring spatial variation. This was also indicated by the Shannon-Weiner Index values for each grab within each station, shown in Figure 5.8. This univariate measure follows a normal distribution across the grab samples at each station and within station was relatively consistant, suggesting that selection of a lower number of samples would be an acceptable way forward.

In the majority of the grab samples many of the species appeared in a single grab or two and always in small numbers (See Appendix 4). The proposed research was to assess the composition of numerically dominant species that are important for bioturbation of sediments. Individual or a few specimens would, therefore, not play a significant role in this process, especially where the species at low numbers are mainly predatory species, as here.

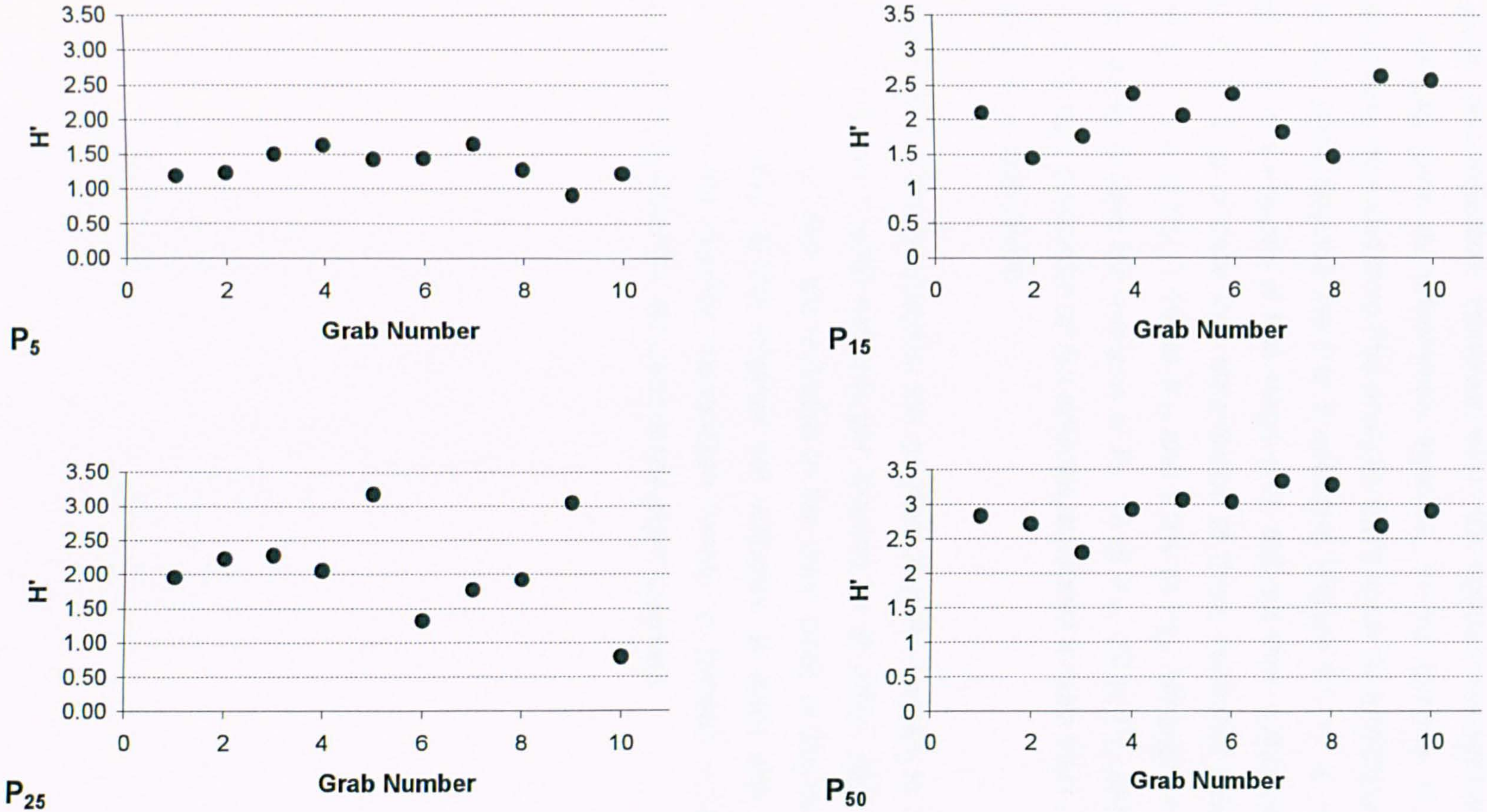


Figure 5.8: Variation in Shannon-Weiner Index (H') for 10x x0.025m^2 Van Veen grabs sampled from 4 stations at Portavadie (P) fish farm in August 2001. Subscripts are distances from the cage edge in metres.

An analysis of the data showing the progressive number of additional species in each subsequent grab sample, for the whole species list, is shown in Figure 5.10. The number of "new" species was reduced to 1 by the tenth grab sample at all stations. Within the first 5 grabs taken approximately 75% of the species diversity, in 10 grabs, was collected. However, when the species that appear in less than 2 grabs (defined here as uncommon species, in the context of the 10 grabs collected) were removed from this analysis there were no additional species being seen in the grabs beyond the first 5 collected (Figure 5.11). In removing these species from the analysis at this stage only, but not from subsequent analysis in section 5.3.5, it is of note that abundance of those removed species constitute 1.5% at P_5, 1.5% at P_{15}, 1.1% at P_{25} and 2.2% at P_{50}, although in species terms 17 were removed from the analysis at P_5, 19 at P_{15}, 24 at P_{25} and 23 species at P_{50}. Thus a high proportion of the species appeared in less than 2 grab samples and all with low abundance.

As a result of the above analysis, the number of grab samples to be collected for the main study was 5 grab samples per location for all sites. Half of the samples collected in August 2001 are included in the main body of the study, detailed in section 5.3.5. Five of the original ten samples at each site were selected randomly, using the number generation facility in Minitab v.13; one number generation, of five from ten, for each of the eight locations.

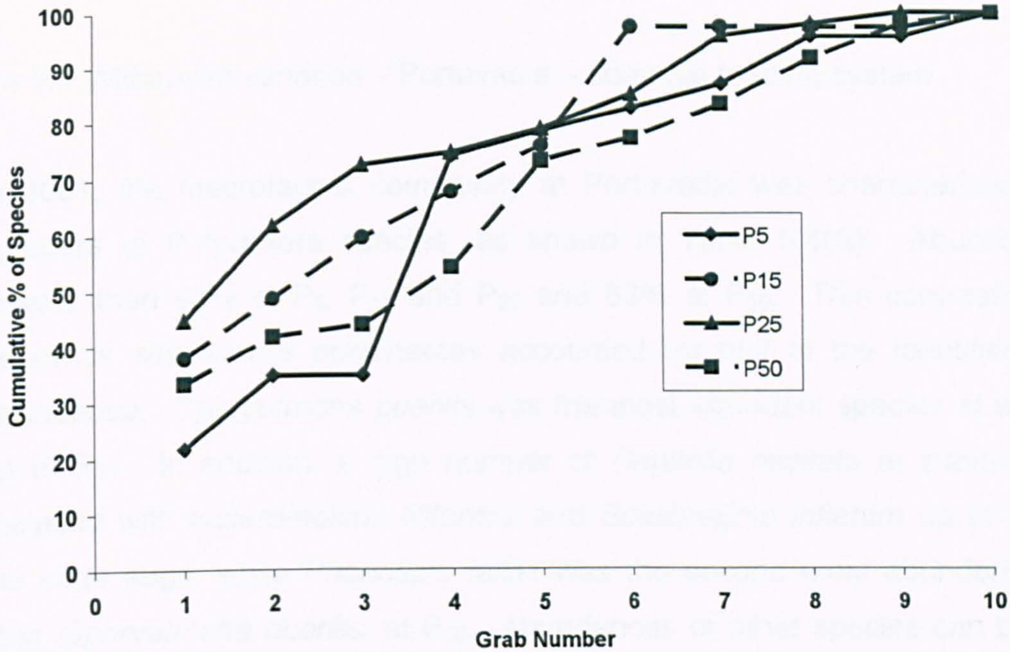


Figure 5.9: Cumulative number of new species that appear in progressive grab samples collected from Portavadie fish farm in August 2001. Species number in grab 1 is the actual number identified. P followed by a number in the legend represents distance from the cage edge in metres.

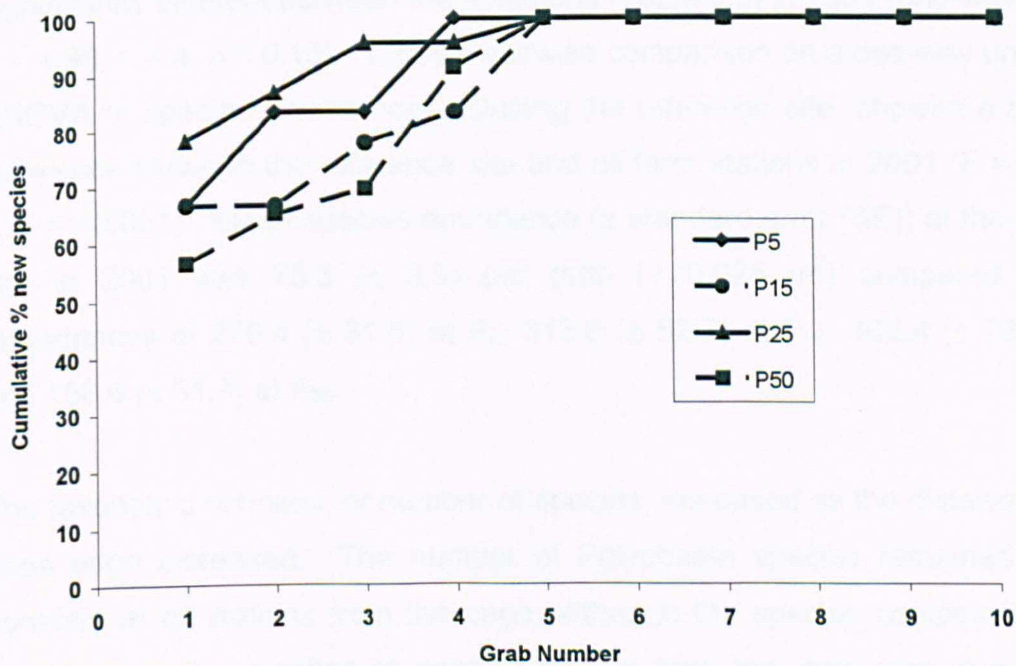


Figure 5.10: Cumulative number of new species that appear in progressive grab samples collected from Portavadie fish farm in August 2001, with species that occur in fewer than 2 grabs (defined here as uncommon species) removed from the analysis. P followed by a number in the legend represents distance from the cage edge in metres.

5.3.5 Benthic Fauna Analysis – Main Study (August 2001, April 2002, April 2003)

5.3.5.1 Within-site variation – Portavadie – adaptive feeding system

In 2001, the macrofaunal community at Portavadie was characterized by high numbers of Polychaeta species, as shown in Table 5.4(A). Abundance was greater than 90% at P_5, P_{15} and P_{25} and 83% at P_{50}. This contrasts with the reference site, where polychaetes accounted for half of the identified species abundance. *Ophryotrocha puerilis* was the most abundant species at all stations up to P_{50}. In addition, a high number of *Capitella capitata* at station P_5 were replaced with *Heteromastus filiformis* and *Scalibregma inflatum* up to 25m from the cage edge, while *Prionospio fallax* was the second most abundant species, after *Ophryotrocha puerilis*, at P_{50}. Abundances of other species can be seen in Table 5.7 and Appendix 4.

Total abundance at the farm site, excluding the reference site, was not significantly different between the 4 stations (Table 5.5) in 2001 (one-way ANOVA; $F = 1.96$, $n = 4$, $p = 0.16$). Tukey's pairwise comparison on a one-way unbalanced ANOVA of species abundance, including the reference site, showed a significant difference between the reference site and all farm stations in 2001 ($F = 9.34$, $n = 5$, $p = <0.001$). Mean species abundance (± standard error (SE)) at the reference site in 2001 was 75.5 (± 3.5) per grab (= 0.025 m^2) compared to mean abundances of 270.4 (± 31.9) at P_5, 313.8 (± 52.7) at P_{15}, 302.4 (± 73.7) at P_{25} and 158.6 (± 31.7) at P_{50}.

The taxonomic richness, or number of species, increased as the distance from the cage edge increased. The number of Polychaeta species remained relatively constant at all stations from the cage, although the species composition varied. Increases in the number of species further from the farm was due to higher numbers of Arthropoda, Mollusca and "other" species found at those stations (Table 5.5).

Table 5.5: Abundance and taxonomic richness in 5 replicate 0.025m^2 Van Veen grab samples of identified groups. Samples taken at Portavadie fish farm and reference site in (A) August 2001 and (B) April 2002. Subscripts in stations represent distance from cage edge in metres. C = reference site.

(A)

2001 Group	Stations P_5	P_{15}	P_{25}	P_{50}	C
Abundance					
Polychaeta	1328	1416	1384	656	76
Arthropoda	2	9	9	6	12
Mollusca	19	129	85	93	26
Others	3	15	34	38	37
Total	**1352**	**1569**	**1512**	**793**	**151**
Number of Species					
Polychaeta	15	20	21	17	29
Arthropoda	1	2	4	5	7
Mollusca	2	6	8	11	6
Others	2	5	5	5	5
Total	**20**	**33**	**38**	**38**	**47**

(B)

2002 Group	Stations P_5	P_{15}	P_{25}	P_{50}	C
Abundance					
Polychaeta	5443	2561	3118	1102	183
Arthropoda	21	12	79	33	21
Mollusca	1	47	71	246	49
Others	0	0	4	13	84
Total	**5465**	**2620**	**3272**	**1394**	**337**
Number of Species					
Polychaeta	5	10	14	14	29
Arthropoda	2	2	5	2	7
Mollusca	1	5	5	7	9
Others	0	0	1	4	7
Total	**8**	**17**	**25**	**27**	**52**

In 2002, the macrofaunal community altered at the fish farm stations, with taxonomic richness reduced within all groups at all stations (Table 5.5(B)). Most notably this occurred at station P_5, with species that were present in 2001 at low abundance, such as *Harmothoe sp.*, *Eteone longa* and *Prionospio fallax*, having disappeared at this station by 2002 (Table 5.7). This provides evidence that environmental conditions had deteriorated at the farm site during the intervening year. The distribution of less tolerant species, such as those mentioned, shifted further away from the farm cage edge, with *Prionospio fallax* found at station P_{15} onwards, *Harmothoe sp.* from P_{25} onwards and *Eteone longa* at P_{50} only. In all cases the abundance of these species was reduced from 2001 data. Overall, Polychaeta abundance was increased at stations P_5, P_{15} and P_{25} and accounted for 99.6%, 97.7% and 95.3% of total abundance, respectively, although the increase at P_{15} was not significant (one-way ANOVA; $F = 2.72$, $n = 2$, $p = 0.138$). The increase in polychaete abundance was brought about by higher numbers of the opportunistic species *Capitella capitata* being present.

Table 5.6: Rank order of top 10 macrofauna species from 4 stations at Portavadie (P) fish farm collected in August 2001 and April 2002. N = abundance in 5 x $0.025m^2$ Van Veen grabs, % = percentage of total abundance. Subscripts represent distance from cage edge in metres.

Station P_5

	August 2001			April 2002		
RANK	SPECIES	N	%	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	793	58.74	*Capitella capitata*	4874	89.19
2	*Capitella capitata*	477	35.33	*Ophyrotrocha puerilis siberti*	393	7.19
3	*Malacoceros fuliginosa*	24	1.78	*Malacoceros fuliginosa*	173	3.17
4	*Eteone longa*	15	1.11	*Corophium* sp. indet.	15	0.27
5	*Abra alba*	15	1.11	*Nebalia bipes*	6	0.11
6	*Platynereis dumerilii*	5	0.37	*Anaitides maculata*	2	0.04
7	*Thyasira flexuosa*	4	0.30	*Protodorvillea kefersteini*	1	0.02
8	*Anaitides maculata*	3	0.22	*Mysella bidentata*	1	0.02
9	*Chaetozone sp.*	2	0.15			
10	*Pectinaria belgica*	2	0.15			

Station P_{15}

RANK	SPECIES	N	%	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	977	62.27	*Capitella capitata*	1694	64.66
2	*Heteromastus filiformis*	110	7.01	*Ophyrotrocha puerilis siberti*	732	27.94
3	*Abra alba*	75	4.78	*Malacoceros fuliginosa*	56	2.14
4	*Anaitides maculata*	57	3.63	*Heteromastus filiformis*	43	1.64
5	*Eteone longa*	54	3.44	*Anaitides maculata*	29	1.11
6	*Scalibregma inflatum*	49	3.12	*Mysella bidentata*	21	0.8
7	*Thyasira flexuosa*	43	2.74	*Thracia* sp. indet.	13	0.5
8	*Scoloplos armiger*	38	2.42	*Corophium* sp. indet.	11	0.42
9	*Prionospio fallax*	38	2.42	*Thyasira flexuosa*	10	0.38
10	*Capitella capitata*	38	2.42	*Scalibregma inflatum*	3	0.11

Station P_{25}

RANK	SPECIES	N	%	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	1020	67.46	*Capitella capitata*	1815	55.54
2	*Scalibregma inflatum*	90	5.95	*Ophyrotrocha puerilis siberti*	1179	36.08
3	*Heteromastus filiformis*	56	3.70	*Heteromastus filiformis*	79	2.42
4	*Prionospio fallax*	50	3.31	*Corophium* sp. indet.	75	2.29
5	*Thyasira flexuosa*	45	2.98	*Mysella bidentata*	34	1.04
6	*Eteone longa*	39	2.58	*Anaitides maculata*	26	0.8
7	*Anaitides maculata*	35	2.31	*Thyasira flexuosa*	19	0.58
8	*Scoloplos armiger*	35	2.31	*Abra alba*	11	0.34
9	NEMERTINI sp. A	23	1.52	*Prionospio fallax*	6	0.18
10	CIRRATULIDAE spp. indet	17	1.12	*Scoloplos armiger*	4	0.12

Station P_{50}

RANK	SPECIES	N	%	SPECIES	N	%
1	*Ophyrotrocha puerilis siberti*	207	26.10	*Ophyrotrocha puerilis siberti*	777	56.1
2	*Scalibregma inflatum*	136	17.15	*Capitella capitata*	124	8.95
3	*Prionospio fallax*	127	16.02	*Abra alba*	87	6.28
4	*Heteromastus filiformis*	73	9.21	*Prionospio fallax*	81	5.85
5	*Eteone longa*	41	5.17	*Mysella bidentata*	75	5.42
6	*Thyasira flexuosa*	37	4.67	*Heteromastus filiformis*	71	5.13
7	*Owenia fusiformis*	21	2.65	*Thyasira flexuosa*	65	4.69
8	*Scoloplos armiger*	19	2.40	*Corophium* sp. indet.	32	2.31
9	NEMERTINI sp. A	17	2.14	*Eteone longa*	16	1.16
10	*Thracia sp. indet.*	15	1.89	*Thracia* sp. indet.	11	0.79

One-way ANOVA comparing changes between the two sampling periods showed significant increases in abundance at Stations P_5 and P_{25} only. At station P_5 in 2002, *Capitella capitata* accounted for 90% of the total abundance with 4874 animals found in the 5 replicates (total area $0.125m^2$), an order of magnitude increase over 2001, with overall abundance increased from 1352 to 5465 macrofauna. *Capitella capitata* succeeded *Ophryotrocha puerilis* as the dominant species at all stations up to P_{25}, as shown in Table 5.7, with the abundance of the latter species not increasing significantly over the two sampling periods (one-way ANOVA; $F = 0.07$, $n = 2$, $p = 0.935$). At P_{25} total abundance was also increased, from 1512 in 2001 to 3272 macrofauna in 2002. Many of the species that appeared at this station in 2001 continued to appear in 2002 but in lower abundance. The increase in overall abundance at P_{25} was entirely due to greater numbers of *Capitella capitata* being present. Similar to 2001, a one-way ANOVA Tukey's pairwise comparison of species abundance showed the reference site mean abundance of 67.2 (± 9.19 = SE) differed significantly from all farm stations ($F = 31.27$, $n = 5$, $p = >0.001$).

The change in community structure is also shown by univariate indices. In both years Shannon-Weiner Index (Hs) values above 4 and Pielou Evenness above 0.8 suggest the undisturbed community at the reference site was both highly diverse and evenly distributed. The diversity indices for P_{50}, based on total grabs taken, was lower than at the reference site, but mean diversity indices of individual grabs did not differ significantly in 2001 (Kruskal-Wallis Test; $H = 3.75$, $df = 1$, $p = 0.053$), showing the outer station measured at the farm site was similar to the reference site. However, the reduction in species diversity at all farm stations in 2002 resulted in the diversity at P_{50} dropping below background levels (Kruskal-Wallis Test; $H = 6.86$, $df = 1$, $p = 0.009$).

Table 5.7: Univariate measures for benthic samples taken at Portavadie and reference site in (A) August 2001 and (B) April 2002, 5 replicates per station using $0.025m^2$ Van veen grab, representing an area of $0.125m^2$. N = species abundance, S = taxonomic richness, D = Simpsons Index, Hb = Brillouin Index, Hs = Shannon-Weiner Index, P = Pielou Evenness and Eh = Heip Evenness. Subscripts represent distance from cage edge in metres. C = Reference Site.

(A) August 2001

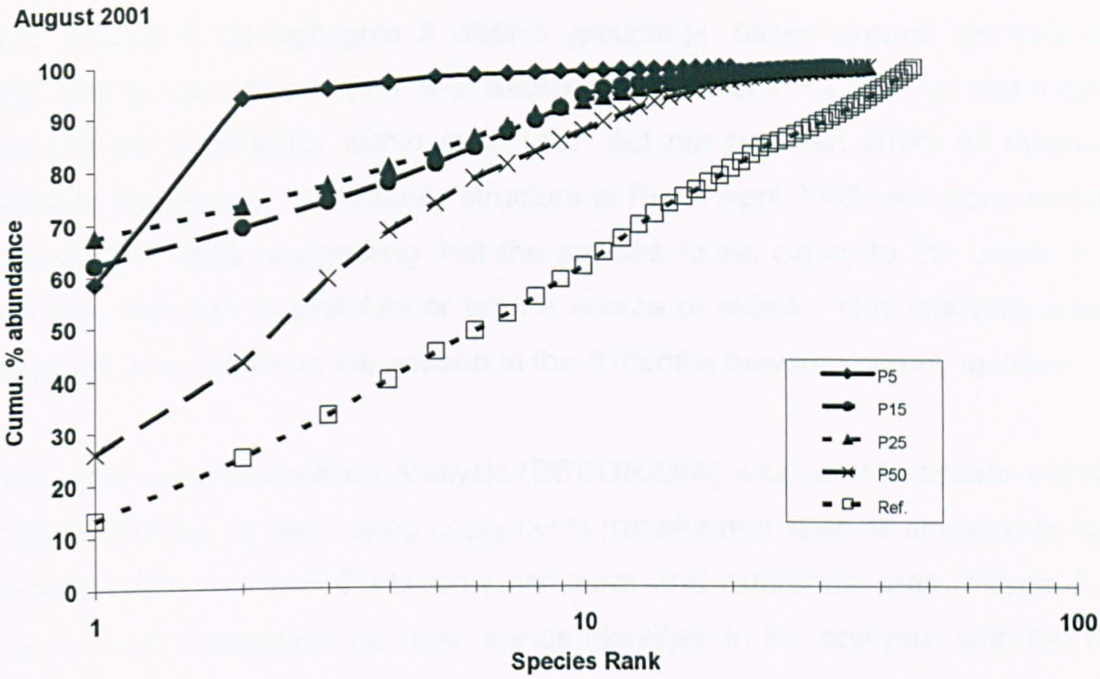
STATION	N	S	D	Hb	Hs	P	Eh
P_5	1352	20	0.53	0.95	1.42	0.33	0.09
P_{15}	1569	33	0.60	1.62	2.39	0.47	0.13
P_{25}	1512	38	0.54	1.49	2.22	0.42	0.10
P_{50}	793	38	0.86	2.35	3.51	0.67	0.28
C	151	47	0.95	2.89	4.79	0.86	0.58

(B) April 2002

P_5	5465	8	0.20	0.43	0.62	0.21	0.08
P_{15}	2620	17	0.50	0.97	1.41	0.35	0.10
P_{25}	3272	25	0.56	1.06	1.56	0.34	0.08
P_{50}	1394	27	0.66	1.67	2.50	0.52	0.18
C	336	52	0.92	2.87	4.27	0.80	0.45

In each year, the Hs and evenness measures of diversity increased with distance from the cage edge at all stations from P_5 to P_{50}. Species abundance and taxonomic richness did not highlight significant differences for all stations between the two sampling dates. However, species diversity was significantly reduced at all stations between 2001 and 2002, as shown by a statistically significant reduction in Hs (Appendix 4).

Further evidence of the reduction in the quality of the sediment is provided by k-dominance curves shown in Figure 5.11. In both years the reference sites were not dominated by any one species, whereas a single species, *Capitella capitata* or *Ophryotrocha puerilis* depending on the year (see above), accounted for a significant proportion of the total abundance at each farm station. In all cases the curves for the farm stations are to the upper left of the reference and in 2002 the curves are elevated above 2001, showing a reduction in diversity as the contribution of one or two species to the total abundance increased.



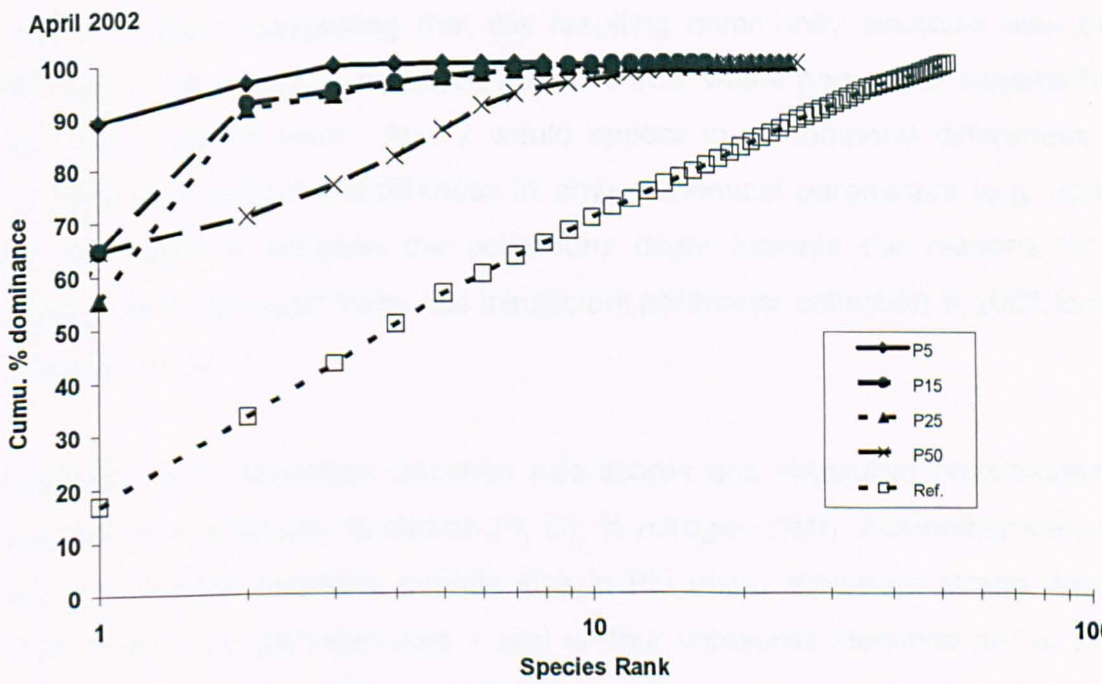


Figure 5.11: k-dominance curves for replicate samples taken at Portavadie fish farm and reference site in August 2001 and April 2002.

Cluster analysis using percentage similarity of species abundance for Portavadie data (Figure 5.12) highlights 3 distinct groupings, based around the reference sites and groups of year data, with exceptions. Stations P_{15} and P_{25} had a similar macrofaunal community within each year, but not between years as taxonomic richness decreased. Community structure at P_{50} in April 2002 was more similar to August 2001 data, suggesting that the species found closer to the cages in the first year had had to shift further from a source of stress. This indicates a wider dispersal of nutrients on the seabed in the 8 months between sampling dates.

Detrended Correspondence Analysis (DECORANA) was used to assess trends in species community data using Log_{10} (x+1) transformed species abundance for all samples collected from Portavadie fish farm and reference sites (Figure 5.13). The two axes represent the main trends identified in the analysis, with the main distribution being along Axis 1 and year group data along Axis 2. Data from station P_5 in both years is clearly identifiable at the start of Axis 1 with all stations in sequence and the reference sites at the end. Thus Axis 1 is showing a clear nutrient gradient suggesting that the resulting community structure was being affected by the impact of increased nutrients from waste particulate material being deposited from the farm. Axis 2 would appear to be temporal differences and correlation between the differences in physio-chemical parameters (e.g. carbon) and the distance between the collections might indicate the reasons for the temporal shift, although there was insufficient parameter collection in 2001 to carry out this analysis.

Spearman rank correlation between axis scores and measured physio-chemical parameters (specifically % carbon (% C), % nitrogen (%N), carbon/nitrogen (CN) ratio and median sediment particle size in Phi units) showed a strong negative linear relationship between Axis 1 and all four measures identified above (Table 5.8). However, it should be noted that the analysis was limited to the availability of the above physio-chemical measures in 2002 only, with no samples collected in 2001, representing a limited number of data points, which highly influenced the rank order used in the correlation. There were no significant correlations with Axis 2.

Table 5.8: Spearman Rank Correlations (coefficient) and probability of significance (p-value) between Axis 1 and Axis 2 variable scores from Detrended Correspondence Analysis of Log_{10} (x+1) transformed macrofaunal species abundance at Portavadie, collected in August 2001, and measured physio-chemical parameters. PS = particle size. * = Significant.

Physio-Chemical Measure	Axis 1		Axis 2	
	Coefficent	p-value	Coefficent	p-value
% Carbon	-1.000	>0.001*	-0.100	0.873
% Nitrogen	-1.000	>0.001*	-0.100	0.873
CN ratio	-1.000	>0.001*	-0.100	0.873
Median PS	0.900	0.037*	0.300	0.624

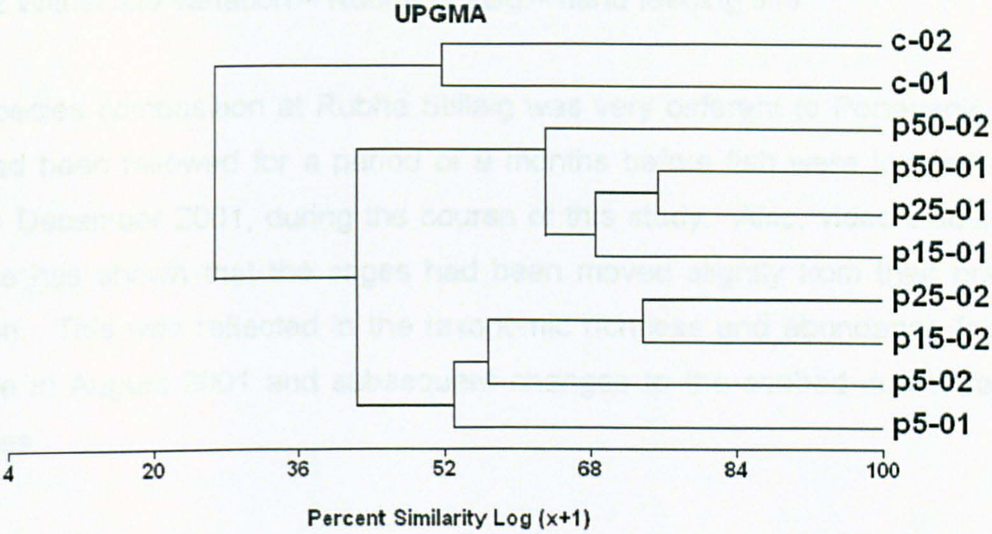


Figure 5.12: Dendogram of multivariate cluster analysis using percentage similarity with UPGMA sorting on Log_{10} (x+1) transformed species abundance for macrofaunal samples collected in August 2001 and April 2002 at Portavadie fish farm and reference sites.

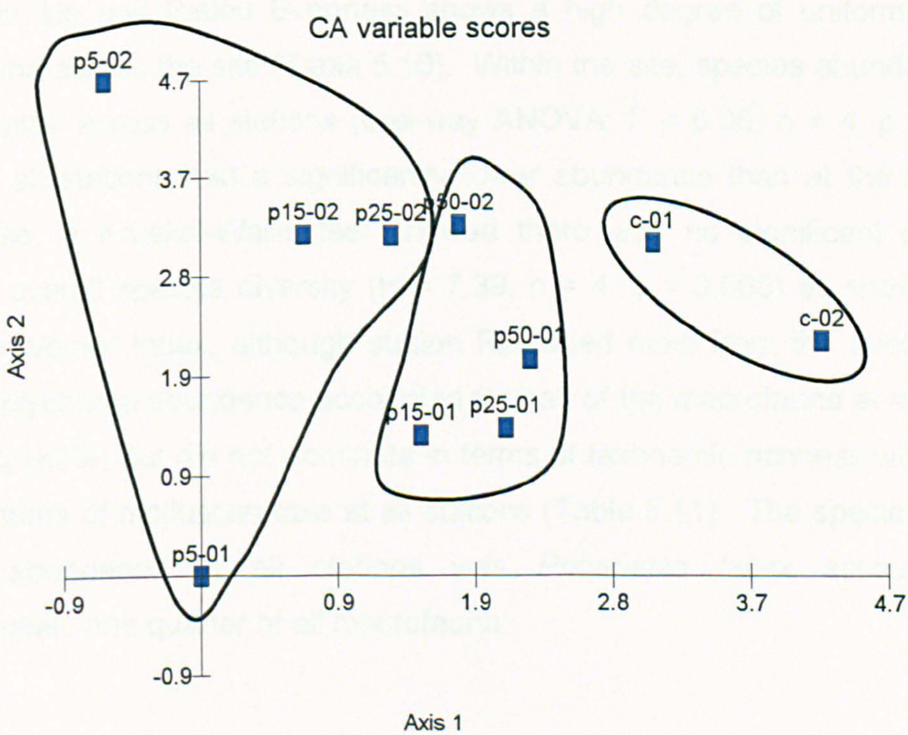


Figure 5.13: Scatter-plot of ordination analysis Detrended Correspondence Analysis (DECORANA) for log_{10} (x+1) transformed abundance of macrofauna collected from Portavadie and reference sites in August 2001 and April 2002.

5.3.5.2 Within site variation – Rubha Stillaig – hand feeding site

The species composition at Rubha Stillaig was very different to Portavadie. The site had been fallowed for a period of 9 months before fish were located at the site, in December 2001, during the course of this study. Also, video evidence of the site has shown that the cages had been moved slightly from their previous position. This was reflected in the taxonomic richness and abundance found at the site in August 2001 and subsequent changes to the seabed due to farming activities.

In 2001, total abundance was low at all stations. Abundance in 5 replicate grabs totalled less than 200 macrofauna at all stations (Table 5.9), with mean abundances per grab (±se) of 30.0 (±6.28) at R_5, 33.2 (±5.06) at R_{15}, 33.0 (±7.48) at R_{25} and 31.8 (±5.83) at R_{50}. However, taxonomic richness was relatively high resulting in reasonable overall diversity and evenness between grab samples. Values for Hs and Pielou Evenness shows a high degree of uniformity in the macrofauna across the site (Table 5.10). Within the site, species abundance was highly similar across all stations (one-way ANOVA; $F = 0.06$, $n = 4$, $p = 0.982$), although all stations had a significantly lower abundance than at the reference site. Also, a Kruskal-Wallis test showed there was no significant difference between overall species diversity ($H = 7.39$, $n = 4$, $p = 0.060$) as shown by the Shannon-Weiner Index, although station R_5 varied most from the average rank order. Polychaeta abundance accounted for half of the macrofauna at all stations except R_5 (82%) but did not dominate in terms of taxonomic richness with equally high numbers of molluscan taxa at all stations (Table 5.11). The species with the highest abundance at all stations was *Prionospio fallax* accounting for approximately one quarter of all macrofauna.

Table 5.9: Abundance and taxonomic richness in 5 replicate 0.025m^2 Van Veen grab samples of identified groups,. Samples taken at Rubha Stillaig fish farm and reference site in (a) August 2001, (b) April 2002 and (c) April 2003. Subscripts in stations represent distance from cage edge in metres. C = reference site.

(a)

2001	Stations				
Group	R_5	R_{15}	R_{25}	R_{50}	C
Abundance					
Polychaeta	123	93	74	80	76
Arthropoda	3	14	19	10	12
Mollusca	16	39	56	42	26
Others	8	20	16	27	37
Total	**150**	**166**	**165**	**159**	**151**
Number of Species					
Polychaeta	8	17	19	14	29
Arthropoda	2	5	8	5	7
Mollusca	5	12	9	10	6
Others	2	6	6	5	5
Total	**17**	**40**	**42**	**34**	**47**

(b)

2002	Stations				
Group	R_5	R_{15}	R_{25}	R_{50}	C
Abundance					
Polychaeta	46	29	118	169	183
Arthropoda	13	8	16	6	21
Mollusca	33	36	62	70	49
Others	7	3	32	35	84
Total	**99**	**76**	**228**	**280**	**337**
Number of Species					
Polychaeta	17	14	13	16	29
Arthropoda	5	4	3	3	7
Mollusca	6	9	8	11	9
Others	2	1	3	3	7
Total	**30**	**28**	**27**	**33**	**52**

(c)

2003	Stations				
Group	R_5	R_{15}	R_{25}	R_{50}	C
Abundance					
Polychaeta	3017	1718	349	217	327
Arthropoda	125	10	11	12	37
Mollusca	28	9	2	38	60
Others	31	0	4	15	55
Total	**3201**	**1737**	**366**	**282**	**479**
Number of Species					
Polychaeta	13	10	8	15	33
Arthropoda	4	2	3	5	8
Mollusca	4	4	1	5	13
Others	2	0	4	7	5
Total	**23**	**16**	**16**	**32**	**59**

Table 5.10: Univariate measures for benthic samples taken at Rubha Stillaig and reference sites in (a) August 2001, (b) April 2002 and (c) April 2003, 5 replicates per station using $0.025m^2$ Van veen grab, representing an area of $0.125m^2$. N = species abundance, S = taxonomic richness, D = Simpsons Index, Hb = Brillions Index, Hs = Shannon-Weiner Index, P = Pielou Evenness and Eh = Heip Evenness. Subscripts represent distance from cage edge in metres. C = Reference Site.

(a) August 2001

STATION	N	S	D	Hb	Hs	P	Eh
R_5	150	17	0.85	1.98	3.21	0.77	0.50
R_{15}	166	40	0.92	2.68	4.33	0.81	0.49
R_{25}	165	42	0.94	2.80	4.55	0.84	0.55
R_{50}	159	34	0.90	2.52	4.11	0.80	0.48
C	151	47	0.95	2.89	4.79	0.86	0.58

(b) April 2002

R_5	99	30	0.94	2.54	4.29	0.87	0.63
R_{15}	76	28	0.90	2.38	4.05	0.84	0.58
R_{25}	228	27	0.90	2.39	3.99	0.83	0.56
R_{50}	268	32	0.90	2.55	4.03	0.79	0.46
C	336	52	0.92	2.87	4.27	0.80	0.45

(c) April 2003

R_5	3201	23	0.36	0.92	1.34	0.30	0.07
R_{15}	1737	16	0.19	0.48	0.71	0.18	0.04
R_{25}	366	16	0.15	0.41	0.68	0.17	0.04
R_{50}	282	32	0.82	2.14	3.34	0.67	0.30
C	479	59	0.94	3.12	4.81	0.82	0.46

By 2002, species abundance had fallen at stations R_5 and R_{15} but increased at the remaining stations (Figure 5.10). The number of species at R_5 had increased, that suggested macrofauna were taking advantage of an increase in available nutrients. Specifically there was an increase in the abundance of enrichment tolerant species, such as *Abra alba* and *Corophium* sp. (Figure 5.11). However, a one-way ANOVA on Log_{10} transformed species abundance showed a significant difference between stations at Rubha Stillaig ($F = 5.41$, $n = 4$, $p = 0.009$) and a Tukey's Pairwise Comparison showed that station R_{15} differed from R_{50}, with all other stations being similar.

Table 5.11: Rank order of top 10 macrofauna species from 4 stations at Rubha Stillaig (R) fish farm collected in August 2001, April 2002 and April 2003. N = abundance in 5 x $0.025m^2$ Van Veen grabs, % = percentage of total abundance. Subscripts represent distance from cage edge in metres.

Station R_5

RANK	August 2001 SPECIES	N	%	April 2002 SPECIES	N	%	April 2003 SPECIES	N	%
1	*Prionospio fallax*	40	27.97	*Thyasira flexuosa*	14	14.74	*Capitella capitata*	2556	79.85
2	*Capitella capitata*	22	15.38	*Heteromastus filiformis*	12	12.63	*Malacoceros fuliginosa*	204	6.37
3	*Heteromastus filiformis*	20	13.99	*Prionospio fallax*	9	9.47	*Ophyrotrocha puerilis sib*	97	3.03
4	*Ophyrotrocha puerilis siberti*	16	11.19	*Corophium* sp. indet	8	8.42	*Anaitides maculata*	92	2.87
5	*Scalibregma inflatum*	14	9.79	*Abra alba*	8	8.42	*Nebalia bipes*	81	2.53
6	*Thyasira flexuosa*	10	6.99	*Mysella bidentata*	5	5.26	*Phoxichilidium femoratum*	40	1.25
7	*Owenia fusiformis*	9	6.29	*Ophyrotrocha puerilis siberti*	4	4.21	*Eteone longa*	30	0.94
8	*Nucula tenuis*	3	2.10	*Capitella capitata*	4	4.21	NEMERTINI sp. A	28	0.87
9	*Westwoodilla caecula*	2	1.40	*Thracia* sp. indet	4	4.21	*Mysella bidentata*	22	0.69
10	*Eteone longa*	1	0.70	*Glycera alba*	3	3.16	*Heteromastus filiformis*	21	0.66

Station R_{15}

RANK	August 2001 SPECIES	N	%	April 2002 SPECIES	N	%	April 2003 SPECIES	N	%
1	*Prionospio fallax*	34	20.61	*Thyasira flexuosa*	21	27.63	*Capitella capitata*	1558	89.69
2	*Thracia* sp. indet	21	12.73	*Prionospio fallax*	8	10.53	*Ophyrotrocha puerilis siberti*	96	5.53
3	*Ophyrotrocha puerilis siberti*	12	7.27	*Abra alba*	5	6.58	*Malacoceros fuliginosa*	39	2.25
4	*Ophiura* sp. juv.	12	7.27	*Owenia fusiformis*	4	5.26	*Heteromastus filiformis*	11	0.63
5	*Scalibregma inflatum*	10	6.06	*Ophyrotrocha puerilis siberti*	3	3.95	*Mysella bidentata*	6	0.35
6	*Heteromastus filiformis*	9	5.45	*Westwoodilla caecula*	3	3.95	*Eteone longa*	5	0.29
7	*Corophium* sp. indet	7	4.24	*Ampelisca typica*	3	3.95	*Anaitides maculata*	5	0.29
8	*Capitella capitata*	6	3.64	*Ophiura* sp. juv.	3	3.95	*Nebalia bipes*	5	0.29
9	*Abra alba*	5	3.03	*Eteone longa*	2	2.63	*Corophium* sp. indet	5	0.29
10	*Scoloplos armiger*	4	2.42	*Diplocirrus glaucus*	2	2.63	*Protodorvillea kefersteini*	1	0.06

Station R_{25}

RANK	August 2001 SPECIES	N	%	April 2002 SPECIES	N	%	April 2003 SPECIES	N	%
1	*Prionospio fallax*	27	16.56	*Prionospio fallax*	27	25.00	*Capitella capitata*	337	92.33
2	*Mysella bidentata*	15	9.20	*Thyasira flexuosa*	16	14.81	*Corophium* sp. indet	7	1.92
3	*Thracia* sp. indet	15	9.20	*Ophiura* sp. juv.	9	8.33	*Heteromastus filiformis*	4	1.10
4	*Abra alba*	12	7.36	*Owenia fusiformis*	8	7.41	*Eteone longa*	3	0.82
5	*Scalibregma inflatum*	10	6.13	*Corophium* sp. indet	5	4.63	*Pariambus typicus*	3	0.82
6	*Jasmineira elegans*	9	5.52	*Eteone longa*	4	3.70	*Mysella bidentata*	2	0.55
7	*Cheirocratus sundevalli*	8	4.91	*Ophyrotrocha puerilis siberti*	4	3.70	NEMERTINI sp. A	1	0.27
8	*Amphiura* sp. juv.	7	4.29	*Diplocirrus glaucus*	4	3.70	*Cerebratulus* sp.	1	0.27
9	*Heteromastus filiformis*	6	3.68	*Capitella capitata*	3	2.78	*Phascolion strombi*	1	0.27
10	*Cirromphalus casina*	5	3.07	*Abra alba*	3	2.78	*Glycera alba*	1	0.27

Station R_{50}

RANK	August 2001 SPECIES	N	%	April 2002 SPECIES	N	%	April 2003 SPECIES	N	%
1	*Prionospio fallax*	40	25.81	*Prionospio fallax*	69	25.75	*Ophyrotrocha puerilis siberti*	86	30.60
2	*Amphiura* sp. juv.	19	12.26	*Rhodine loveni*	31	11.57	*Capitella capitata*	74	26.33
3	*Owenia fusiformis*	13	8.39	*Ophiura* sp. juv.	21	7.84	*Prionospio fallax*	20	7.12
4	*Nucula tenuis*	10	6.45	*Turritella communis*	18	6.72	*Thyasira flexuosa*	18	6.41
5	*Thyasira flexuosa*	8	5.16	*Owenia fusiformis*	17	6.34	*Cirratulus caudatus*	12	4.27
6	*Abra alba*	7	4.52	*Nucula tenuis*	16	5.97	*Mysella bidentata*	11	3.91
7	*Cheirocratus sundevalli*	6	3.87	*Gouldia minima*	12	4.48	*Phryganella marinus*	8	2.85
8	*Heteromastus filiformis*	5	3.23	*Diplocirrus glaucus*	11	4.10	*Eteone longa*	6	2.14
9	*Cirromphalus casina*	5	3.23	*Jasmineira elegans*	9	3.36	*Abra alba*	6	2.14
10	*Eteone longa*	4	2.58	*Thyasira flexuosa*	9	3.36	*Ampelisca brevicornis*	5	1.78

Despite this, overall species diversity and evenness, as measured by Hs and Pielou Evenness, continued to remain high at all stations and had improved at R_5 in 2002. Also, there was no significant difference in Hs between stations, including the reference site (Kruskal-Wallis Test; H = 7.60, n = 2, p = 0.107). In addition to the increase in nutrient tolerant species, identified above, Table 5.11 also shows how the species composition altered between 2001 and 2002. The number of *Prionospio fallax* was reduced at all stations except R_{50} and the number of molluscan taxa had increased, particularly the abundance of *Thyasira flexuosa*. However, species such as *Scalibregma inflatum* and *Jasmineira elegans* had disappeared from the sediment at all but the outlying farm station (R_{50}).

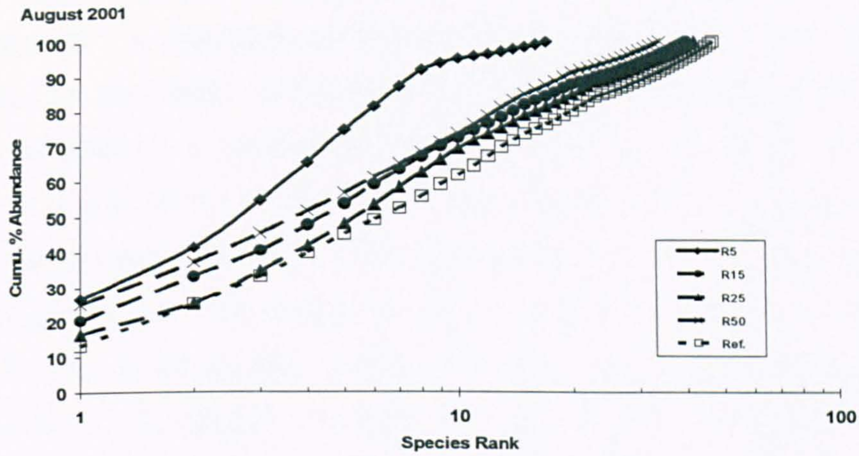
By 2003 there had been fish in the cages at Rubha Stillaig for 16 months. Since the previous April there was an increase in abundance but a reduction in taxonomic richness at all stations (Table 5.9). Stations R_5, R_{15} and R_{25} all showed an increase in the organic-nutrient tolerant opportunistic species *Capitella capitata*, *Malacoceros filiginosa* and *Ophryotrocha puerilis* (Table 5.11) indicating nutrients in sediment at the site had increased. These stations also showed marked reductions in diversity and evenness (Table 5.10) resulting in at least one site differing from another (Kruskal-Wallis Test; H = 13.86, df = 3, p = 0.003). The largest reductions occurred at stations R_{15} and R_{25} having reduced from 4.05 and 3.99 in 2002 to 0.71 and 0.68 in 2003, respectively.

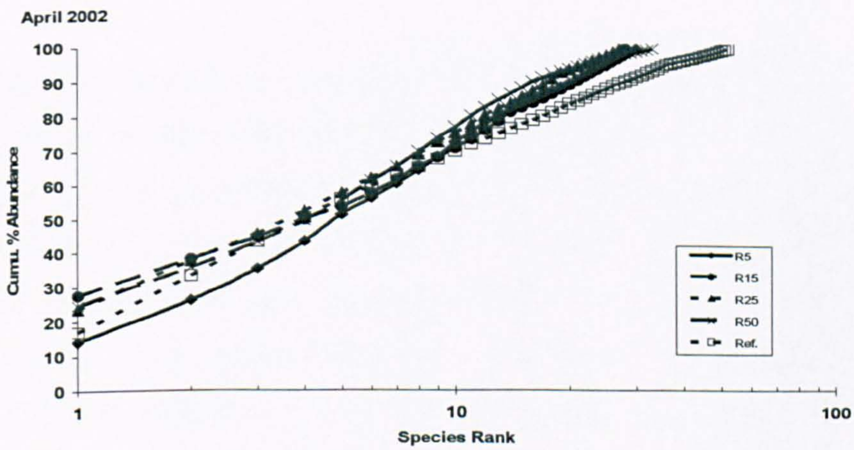
A homogeneity of variance test on 2003 abundance at the farm site (R_5 – R_{50}) showed that variances were not equal (Bartlett's test p = 0.028) even after transformation, and a Kruskal-Wallis test on Log_{10} transformed data showed there was a significant difference between sites (H = 16.42, n = 4, p = 0.001), with at least one station varying from another. R_5 differed most from the average rank order as result of the higher abundance, although all stations differed. Mean abundance per grab rose at all stations, including the reference site, to 640.2 (±79.1) at R_5, 347.4 (±42.3) at R_{15}, 73.2 (±17.9) at R_{25}, 56.4 (±30.0) at R_{50} and 95.8(±5.82) at the reference site.

Analysis of the 3 years' data showed there was significant change to the community structure of macrobenthic species between the sampling periods. At R_5 the increase in abundance in 2003 contributed most to the difference between samples over time (Kruskal Wallis Test; $H = 9.62$, $df = 2$, $p = 0.008$) and the median Hs of 0.99 also significantly reduced the diversity at this station from the start of farming activities. At R_{15} mean abundance was significantly different across all dates (one-way ANOVA; Log_{10} transformed data, $F = 50.25$, $n = 3$, $p = >0.001$). In 2002, the reduction in abundance may have been due to a reduced number of nutrient intolerant species but insufficient time for opportunistic species to have proliferated. However, by 2003 nutrient tolerant polychaete species, such as *Capitella capitata*, had increased in number at R_{15}.

One-way ANOVA on Log_{10} transformed data showed that stations R_{25} and R_{50} did not vary in abundance across the 3 years of sampling ($F = 2.48$, $n = 2$, $p = 0.125$ and $F = 0.90$, $n = 2$, $p = 0.432$ respectively), suggesting that farming activities were not affecting the seabed beyond 25m from the cages edge. However, the lower Hs scores in 2003, particularly at R_{25}, showed that the reduced taxonomic richness was affecting overall diversity at that station.

K-dominance curves for all 3 sampling dates are shown in Figure 5.14 and provide evidence of a reduction in the quality of the sediment. In 2001 and 2002 the curves for all stations at the farm site were similar to the reference, in that they were not dominated by any one particular species. The elevated curve for station R_5 in 2001 resulted from lower taxonomic richness at this station compared to the remaining stations. However, in 2003, a high proportion of the total abundance at stations R_5, R_{15} and R_{25} was accounted for by a single species, in this case *Capitella capitata*, as indicated by the elevated curves, showing a reduction in diversity at these stations.





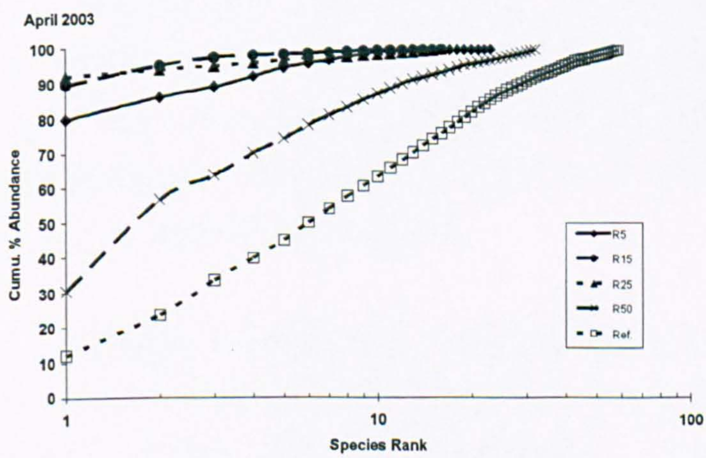


Figure 5.14: k-dominance curves for replicate samples taken at Rubha Stillaig fish farm and reference site in August 2001, April 2002 and April 2003.

Cluster analysis using percentage similarity for Rubha Stillaig data (Figure 5.15) species abundance highlights distinct groupings, based around the reference sites and groups of year data. Three distinct groups are in evidence with cluster one including all stations in years 2001 and 2002 with the exception of R_{50} in 2002. Within this cluster there are 2 sub-clusters with R_{15}, R_{25} and R_{50} in 2001 being more similar to each other than to the 2002 data. R_5 in 2001 is included within the group being 42% to 55% similar to other years' group data. A second cluster includes the 3 reference sites in years 2001-2003 and station R_{50} in 2002. Cluster 3 consists of all farm stations in 2003, with the minimum similarity between these samples and all remaining samples, being as low as 14%.

Detrended Correspondence Analysis (DECORANA) for all samples collected from Rubha Stillaig fish farm and reference sites is presented in Figure 5.16. The two axes represent the main trends identified in the analysis, with the main distribution being along Axis 1. Data from 2003 is clearly identifiable at the start of Axis 1 with the reference sites at the end. Although the 2001 and 2002 groups intermingled in the centre of the axis, overall there was a clear temporal trend applying to the data in axis 1, slightly different from the data reported for Portavadie. However, Axis 1 still represents a nutrient gradient, where the data for 2001 and 2002 appear at the cleaner end of the axis and confirms the lack of impact highlighted by the Shannon-Weiner and k-dominance curves presented above. Axis 1 is therefore linked to the impact from the on-going farming activities, probably as a result of particulate waste settlement. This is also indicated by the structure within each grouping, where $R_5 - R_{50}$ appear in sequence.

Spearman rank correlations between axis scores (re-run for 2002 and 2003 data only) and measured physio-chemical parameters showed one significant relationship (Table 5.14) with percentage total nitrogen. However, it was also useful to correlate the differences in measured physio-chemical parameters with the distances between corresponding stations from the DECORANA analysis to assess whether the temporal shift between years might be explained. Distances between stations in each year where measured directly off Figure 5.16 and compared against differences in sediment carbon, nitrogen and median grain size between the years at each station. Parameter data for 2001 was not collected but conditions were assumed to be similar to 2002 as described above and thus distance between the mid-point (2001/2002) and 2003 stations were compared (table 5.13). However, there was no significant correlation with the measured parameters that might explain the shift in axis 1.

Table 5.12: Spearman Rank Correlations (coefficient) and probability of significance (p-value) between Axis 1 and Axis 2 variable scores from Detrended Correspondence Analysis of Log_{10} (x+1) transformed macrofaunal species abundance at Rubha Stillaig, collected in April 2002 and April 2003, and measured physio-chemical parameters. PS = particle size. * = Significant.

Physio-Chemical Measure	Axis 1 Coefficent	Axis 1 p-value	Axis 2 Coefficent	Axis 2 p-value
% Carbon	-0.578	0.080	-0.480	0.160
% Nitrogen	-0.714	0.020*	-0.441	0.202
CN ratio	0.370	0.293	-0.018	0.960
Median PS	0.171	0.637	0.006	0.987

Table 5.13: Spearman Rank Correlations (coefficient) and probability (p-value) from distance between the mid-point (2001/2002) and 2003 stations and measured physio-chemical parameters. PS = particle size. (See text also).

Physio-Chemical Measure	Distance Coefficent	Distance p-value
% Carbon	-0.279	0.721
% Nitrogen	0.440	0.560
Median PS	-0.496	0.504

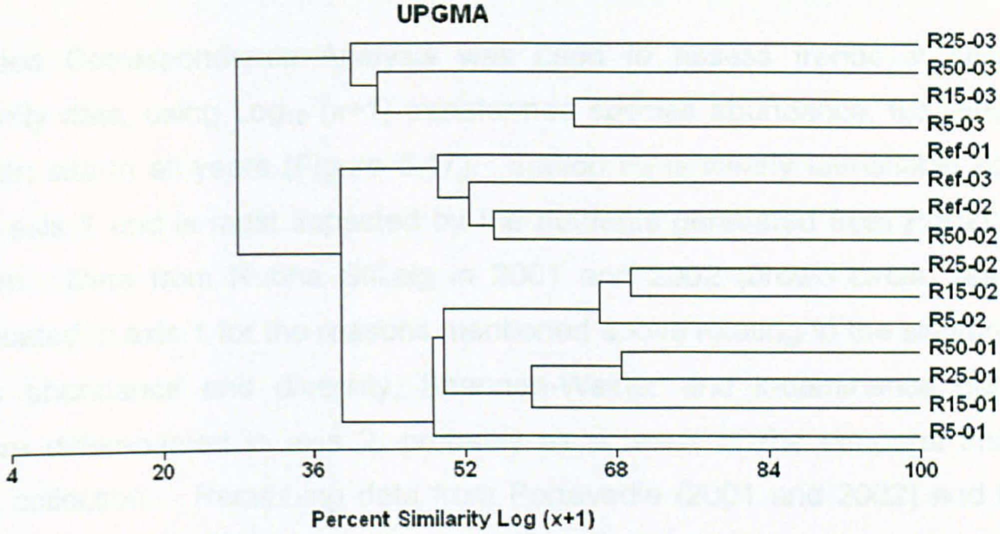


Figure 5.15: Dendogram of multivariate cluster analysis using percentage similarity with UPGMA sorting on $Log_{10} + 1$ transformed species abundance for macrofaunal samples collected in August 2001, April 2002 and April 2003 at Rubha Stillaig fish farm and reference sites.

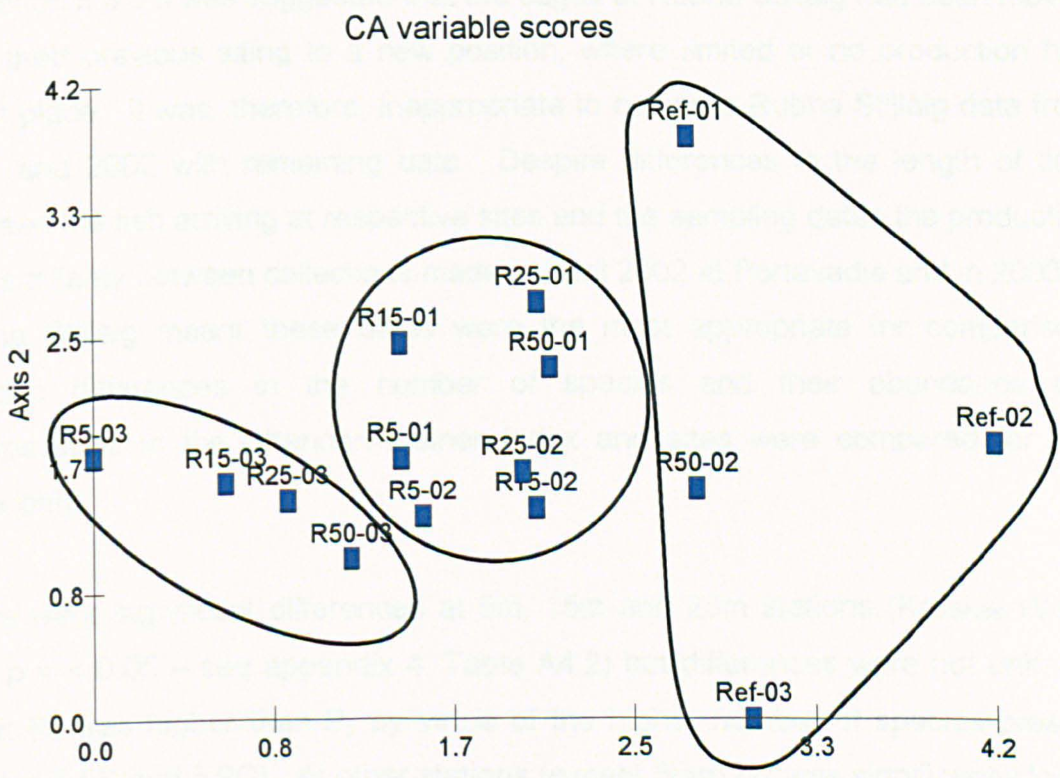


Figure 5.16: Scatter-plot of ordination analysis Detrended Correspondence Analysis for $\log_{10}$ (x+1) transformed abundance of macrofauna collected from Rubha Stillaig and reference sites in August 2001, April 2002 and April 2003.

5.3.5.3 Between-site variation

Detrended Correspondence Analysis was used to assess trends in species community data, using Log_{10} (x+1) transformed species abundance, for samples from both site in all years (Figure 5.17). Station P_5 is clearly identifiable at the start of axis 1 and is most impacted by the nutrients generated from Portavadie fish farm. Data from Rubha Stillaig in 2001 and 2002 (brown circle) are not differentiated in axis 1 for the reasons mentioned above relating to the similarity in species abundance and diversity, Shannon-Weiner and k-dominance curves. They are differentiated in axis 2, probably as a result of the temporal shift in sample collection. Remaining data from Portavadie (2001 and 2002) and from Rubha Stillaig (2003) show similar responses to the level of nutrient input (axis 1 – black circle).

In section 5.3.3 it was suggested that the cages at Rubha Stillaig had been moved from their previous siting to a new position, where limited or no production had taken place. It was, therefore, inappropriate to compare Rubha Stillaig data from 2001 and 2002 with remaining data. Despite differences in the length of time between the fish arriving at respective sites and the sampling dates the production time similarity between collections made in April 2002 at Portavadie and in 2003 at Rubha Stillaig meant these dates were the most appropriate for comparison. Specific differences in the number of species and their abundance are encapsulated in the Shannon-Weiner Index and sites were compared for this index only.

There were significant differences at 5m, 15m and 25m stations (Kruskal Wallis test, $p = < 0.05$ – see appendix 4, Table A4.2) but differences were not uniform. Hs at R_5 was higher than P_5 by virtue of the higher number of species present (Tables 5.5B and 5.9C). At other stations (except 50m) Hs was significantly lower at Rubha Stillaig than at Portavadie which suggests the sediments beneath the cages at Rubha Stillaig had been more impacted using the hand feeding regime. However, the lower Hs values at Rubha Stillaig in 2003 may have been due to difficulties in gaining sufficient penetration of the sediment using the Van Veen grab. Combining data from all stations for a comparison between sites showed there was no significant difference in Shannon-Weiner Index between Portavadie and Rubha Stillaig sites (Kruskal Wallis test, $H = 0.40$, $df = 1$, $p = 0.525$).

Similarity in the species composition between sites was described in section 5.3.5.2 above. Specifically, the most abundant species at each site consisted of *Capitella capitata*, *Malacoceros filiginosa* and *Ophryotrocha puerilis*, species that are known to tolerate high levels of organic nutrient deposition under fish farm cages. Distinction between the abundance of these species may have resulted from the immediate history of previous production (or lack of production e.g. Rubha Stillaig) and natural variability. It is therefore suggested that the stress applied to species in the vicinity of cages, by the deposition of particulate material rich in organic nutrients, resulted in no fundamental differences in benthic populations using the different feeding systems at each site.

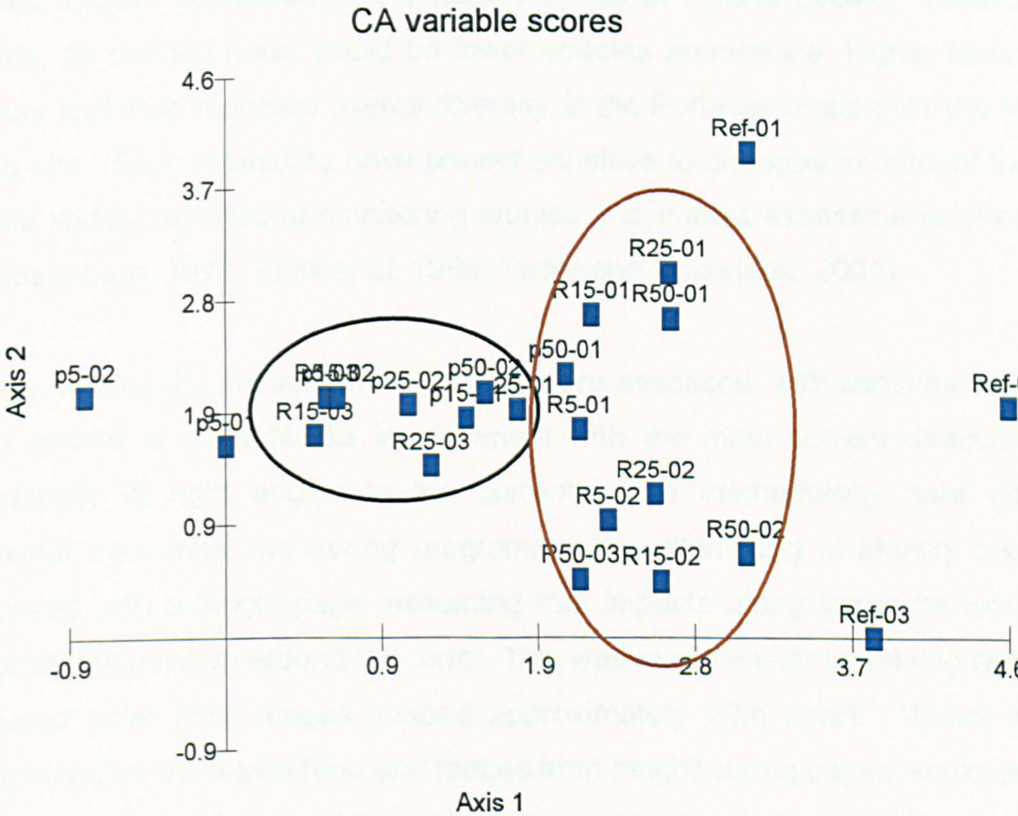


Figure 5.17: Scatter-plot of ordination analysis Detrended Correspondence Analysis for log_{10} (x+1) transformed abundance for all macrofauna collected from Portavadie (p), Rubha Stillaig (R) and reference sites (Ref) in August 2001, April 2002 and April 2003.

5.4 Discussion

The detailed macrofaunal and physio-chemical analysis presented here reflect the need to understand in detail the progressive changes in community structure at each of the two sites. This was needed to assess whether the use of an adaptive feeding system at Portavadie provided intermediate-term benefits, in the form of a reduced impact, compared to the hand fed site at Rubha Stillaig. Measurable benefits, as defined here, would be lower species abundance, higher taxonomic richness and thus improved overall diversity at the Portavadie site over the Rubha Stillaig site. Such measures have proved sensitive to changes in nutrient loading and are widely reported in monitoring studies and impact assessments (Pearson and Rosenberg, 1978; Ervik *et al*, 1998; Telfer and Beveridge, 2001).

In many studies the impacts of whole farms are assessed, with samples collected in the middle of cage blocks in alignment with the main current direction and occasionally at right angles to the current. The methodology here differed somewhat from most monitoring programmes by attempting to identify changes associated with a single cage, assuming that impacts along transects would be similar at any position around the farm. This was made easier by having two sites that used polar circle cages spaced approximately 20m apart. Whilst it was recognised that the waste feed and faeces from neighbouring cages would deposit within the transect line used for collection, equally the experimental cage would "lose" particulates to other cages and the overall effect on species abundance and diversity would be the same. By adopting this strategy at both sites it allowed direct comparison of the data, without reference to and making adjustments for variations in stocking density, fish biomass and number of cages across each of the sites.

5.4.1 Preliminary benthic analysis

The selection of 5 grabs for the main study was justified, where the removal of uncommon species suggested that the majority of the abundance would appear in the first 5 grabs only. The removal of "uncommon" species, defined here as species whose abundance was low in each sample, is often done in benthic studies (Brazner and Beals, 1997), with uncommon species thought to contribute little to community analysis. In a recent analysis of benthic indicator species for use in the implementation of the Water Framework Directive (Hiscock et al, 2004) it was noted that rare species are unsuitable to reliably identify affects. Thus, in the context of fish farms, where the stressor (nutrient waste) is significant, rare species are unlikely to contribute to the analysis.

Importantly, the reduction in the number of Van Veen grabs from 10 to 5 made little difference to the number of the less common species found in the main study with the total number of species remaining similar between the preliminary and main studies. Also, the 5 replicates used in the main study at Portavadie and Rubha Stillaig were in line with published literature (Yi *et al*, 1988).

5.4.2 Main Study

Henderson and Ross (1995) provide the most comprehensive assessment of the state of the seabed surrounding fish farms on the west coast of Scotland, in their comprehensive study of data from up to 50 farms. The 13-57 species they observed in Lower Loch Fyne are similar to the 8-52 species found in this study, although the >13,000 ind. $0.1m^{-2}$ they observed was much larger than the 5,400 ind. $0.125m^{-2}$ found here, despite similar methods of data collection and equipment. The 3.4% organic carbon they measured in sediments is higher than recorded total carbon found at either Portavadie or Rubha Stillaig (< 2%, Figure 5.1) between 2001 and 2003 and may be one of the reasons lower abundances were found here. Henderson and Ross (1995) requested data from "moderate to large farms...>250t" chosen to represent "high tonnage farms (synonymous with high feed usage, wastage and organic input)", which by today's standards is considered small. Also they do not detail site specific data, such as feed input and number of cages, so it is difficult to ascribe differences to any particular reason.

There was a high degree of similarity between the species found in this study, other studies in Scottish sea lochs (Weston, 1990; Henderson and Ross, 1995) and at sites worldwide (Tsutsumi *et al*, 1990). In this study, Polychaeta and Mollusca dominated all stations in both species number and abundance. Polychaeta especially are important in mineralization processes and within this group, *Capitella capitata* is known to increase rates of mineralization by 87% (Heilskov and Holmer, 2001). Although the presence of *Capitella capitata* is regarded as part of the negative impact experienced at fish farms, some researchers have proposed spiking sediments with these polychaetes, at densities of 59,000 ind. m^{-2}, because they had been shown to counter the onset of reduced conditions by rapidly decomposing organic matter at rates of 1-2g C m^{-2} d^{-1} (Chareonpanich *et al*, 1993). Clearly, the 5,400 individuals per $0.125m^{-2}$ found at Portavadie in this study was insufficient to counter the effects of sedimentation from the cages and it resulted in reduced conditions, indicated by the presence of the sulphur-oxidizing bacteria *Beggiotoa* sp. at the sediment surface (Plate 5.1). Henderson and Ross (1995) have shown that much higher densities of polychaetes are found at some locations and it is reasonable to assume that the increased abundance found at both sites over time could benefit the sediment, through bioturbation and mineralization processes.

Henderson and Ross (1995) describe *Capitella capitata*, *Ophryotrocha puerilis* and *Malacoceros fuliginosa* as some of the few species able to exploit grossly impacted sediments. All 3 species were found at both sites during this study and were the top 3 most abundant species 5m from the cage edge at Portavadie (both years) and at Rubha Stillaig in 2003. Importantly neither site experienced an azoic zone during the course of this study; the presence of macrofauna illustrated with the video survey. Brown *et al* (1987) noted that in extreme cases no species are able to survive where conditions had deteriorated the most, particularly under cages. Although grab samples were not taken directly below cages and the use of divers was too expensive to collect cores, the video produced in October 2002, half way between the last two sampling dates, showed at least one species of polychaete, thought to be *Capitella capitata*, living under the cages at Portavadie.

Other studies have also shown that deteriorating conditions do not necessarily result in complete defaunation (Findlay *et al*, 1995).

A number of studies have identified that cage culture impacts the seabed (Brown *et al*, 1987; Weston, 1990; Gowen *et al*, 1994; Kempf *et al*, 2002) but few have reported any direct correlations between levels of carbon and nitrogen in sediments and impact in terms of benthic fauna. This was a relatively small study with a few measures limited to two sites only, but both Portavadie (in 2002) and Rubha Stillaig (in 2003) provided a distinct negative correlation between axis scores from detrended correspondence analysis, reflecting the overall community structure, with carbon and nitrogen in sediment, particle size and CN ratio.

Although these were all statistically significant relationships it was important to identify factors that cause some effect rather than pure trends in the data. Particle size is one of the characteristics of sediments known to influence species composition (Etter and Grassle, 1992; Snelgrove and Butman, 1994). Weston (1990) noted a correlation with particle size at fish farm sites but in this context the settlement of larger particles to the seabed may be an artefact of a reduction in water flow (Inoue, 1972; Black, unpublished data) caused by the presence of cages. Of itself, it cannot explain increases in opportunistic species, reductions in diversity and the increased abundance seen at the sites. During this study particle size was similar at all stations and at both sites and was not thought to influence any difference in community structure between sites.

Similarly, a lower CN ratio than was present at the reference site is a by-product of mineralization processes in the sediment after they have been impacted and abundances increased. Macrofauna preferentially use nitrogen in mineralization processes (Boyd, 1975) and lower CN ratio cannot be treated as a significant cause of macrofaunal changes except through secondary effects. It was therefore more likely that deposition of carbon and nitrogen in waste food and faeces were the primary and driving factors that impact sediment and changes to macrofaunal composition. Other studies have also shown that water currents and water depth influence the degree of impact found at sites with deep water and high speed currents spreading waste over a wider area (Carroll *et al*, 2003) but current speed and water depth at each site studied were similar.

Conditions were visually less extreme at Rubha Stillaig, in that no *Beggiotoa* sp. could be seen on the sediment surface nor were they present in sediment grabs. However, conditions had deteriorated here in 2003, indicated by the reduced diversity and increased abundance at all stations. Henderson and Ross (1995) suggest that percentage dominance of a single species at <30% is indicative of undisturbed sediments. The k-dominance curves for 2001 and 2002 (Figure 5.14) showed the most dominant species accounted for less than 30% of the overall abundance. The slight elevation of R_5 in 2001 resulted from lower taxonomic richness at that station, but this had improved by 2002, such that all k-dominance curves were similar in that year. However, the overall condition of Rubha Stillaig had deteriorated markedly by 2003, shown by the shift of the k-dominance curve to the upper left, by increased dominance of *Capitella capitata* (Table 5.9) and by the lower diversity (Table 5.10). At both of these sites k-dominance curves provided a good method for assessing the impact of fish farming on sediment quality, without the need to quantitatively weigh individual species as would be required to assess changes using abundance biomass curves (ABC) (Costello *et al*, 2001). However, within the 2 – 3 year time-span available there was insufficient data to know whether conditions would continue to worsen at either site, whether some equilibrium, albeit impacted, would be reached by either sites, or whether this equilibrium community would be different between the sites.

A number of studies that identified the influence of cage culture with impacts on the seabed have ascribed distances associated with degree of impact (Brown *et al*, 1987; Weston, 1990; Johannessen *et al*, 1994), with variations in fish biomass, hydrography, and water-depth and sediment type accounting for differences as identified above. Henderson and Ross (1995) are more cautious in their approach and do not set distances to their proposed zones of impact. They define 4 zones; Gross, Heavy, Moderate and non-impacted; based primarily on univariate measures of abundance, number of species and diversity measures, such as Shannon-Weiner Diversity Index. Using these same measures the impact at

Portavadie became higher in 2002 with all stations to P_{25} being grossly impacted and P_{50} being heavily impacted compared to the previous year. At Rubha Stillaig the change is more obvious having shifted from non-impacted at the first two sampling periods to grossly impacted at stations R_5 – R_{25} in 2003. In applying these zones there seems to be little difference between the two sites once production had begun in earnest, as indicated by the similarity between Portavadie in 2002 and Rubha Stillaig in 2003 (Table 5.14).

Table 5.14: Classification of impact at study sites base on zones described in Henderson and Ross (1995) and based on univariate measures on taxonomic richness, species abundance and Shannon-Weiner Diversity Index.

	Portavadie		Rhuba Stillaig		
Station	2001	2002	2001	2002	2003
5	Gross	Gross	Moderate	non-	Gross
15	Heavy	Gross	non-	non-	Gross
25	Heavy	Gross	non-	non-	Gross
50	Moderate	Heavy	non-	non-	Moderate
Reference	non-	non-	non-	non-	non-

The species composition at the reference site bore some similarities to the farm stations, with the presence of *Prionospio fallax*, *Scalibregma inflatum*, *Thyasira flexuosa* and *Mysella bidentata*, all nutrient tolerant species. The occurrence of these species at the reference site suggests a degree of disturbance may exist (Henderson and Ross, 1995), despite high Shannon-Weiner Index values (Table 5.10) indicating a high background level of community diversity throughout the area. This may have been the result of historical work in the bay where an oil-rig production facility was built some 27 years before. It may also simply reflect the more general biodiversity in Loch Fyne, but there is insufficient data from other reference stations to confirm this. It was not feasible to select an alternative reference site, as the central channel in Loch Fyne was much deeper (80-90m) and experiences faster currents that would have made this choice of site inappropriate.

Particular interest was generated in the Rubha Stillaig site when it became apparent that the present location was different from previous production at the site. Video evidence identified the start of *Beggiotoa* sp. bacterial mats some 42m distance from the cage edge and although the full extent could not be assessed, it was assumed that the bacterial mat identified it as the previous location. If a similar cage layout was assumed then the overall shift in position was approximately 100m, within the bounds of the lease.

The Shannon-Weiner measure of diversity at all Rubha Stillaig stations in 2001 and 2002 was not statistically different from the reference site and were regarded as undisturbed at the start of the study. However, there were noticeable differences in species composition between the two sampling dates that suggested an increase in nutrient deposition had occurred even if univariate measures did not identify it. Specifically, nutrient tolerant species such as *Abra alba* and *Corophium* sp. that were not present at R_5 in 2001 were collected in 2002 and less tolerant species such as *Jasminiera elegans* and *Scalibregma inflatum* had disappeared from that station. Such subtle changes were more difficult to determine at greater distances, with reduced abundance of some species possibly being the result of natural variability rather than impact from the cage site. This was recognised by Henderson and Ross (1995) who noted that distinction between moderate and mild impacts were difficult to determine.

Grall and Chauvaud (2002) note that whereas meiofauna and bacteria react quickly to changes in nutrient levels (days), macrofauna lag some way behind (weeks). Whilst this general assertion conflicts with the work of Mazzola *et al* (1999), who showed that nematode abundance increased and other meiofauna decreased within weeks of the start of fish production, it is generally accepted that macrofaunal changes take longer and may be measured in weeks or months.

In this study, there was no fundamental alteration in macrofaunal community structure within the first 12 – 16 weeks of production at Rubha Stillaig, the fish having arrived in late December 2001 and the second sampling having taken place in April 2002. Thus the change occurred sometime between 5 and 16 months (between 2002 and 2003). This initial 5+ months may represent a transitional period, with the lower abundances indicating some degree of impact as species were coming to terms with an increase in nutrients, but that insufficient time had passed to allow the proliferation (through recruitment) of opportunistic species, such as *Capitella capitata*, more commonly associated with impacted sediment.

The fallow period at Rubha Stillaig prior to this study, the continued production at Portavadie and the apparent relocation of cages at Rubha Stillaig resulted in different starting points against which to make a direct comparison between sites, which resulted in 2001 and 2002 data from Rubha Stillaig being discounted. In reality the only valid comparison from all the data was between 2002 at Portavadie and data from 2003 at Rubha Stillaig (Section 5.3.5.3). There were some differences between comparable stations at each site on these dates using univariate measures. Also Ordination techniques highlighted station P_5 as the most impacted of all the stations (Figure 5.17) but overall, specific differences between data for Portavaide (2002) and Rubha Stillaig (2003) were not highlighted. It was concluded that the use of the adaptive feeding systems did not inherently benefit the benthic community at the Portavadie site and the available data suggests the hypothesis that no difference in benthic species composition would exist between sites using different feeding systems was proven.

In an idealised situation, comparison of this type would require two previously unused sites. Despite the fact that cage layout, fish number and biomass; feed type and ration can all be controlled, two identical sites do not exist in the marine environment because of differences in bathymetry, hydrography and exposure. The sites would also have to be assessed prior to the commencement of production (Ervik *et al*, 1998) to provide a detailed picture of the state of the seabed and natural variability in macrofaunal composition. Continued assessment during and after production would complete the BACI design (Green, 1979; Underwood, 1991) and only then might the short and intermediate term effects on the benthic structure between the different feeding methods be compared. Such sites were not available during this study.

That both sites in this study experienced degradation of the sediment under and around fish farm cages was not unexpected and compares with other studies at fish farms (Weston, 1990; Henderson and Ross, 1995). Even if the use of current adaptive feeding technology was able to eliminate high nutrient feed waste, which it does not proclaim, the use of open cage systems mean that faecal material would continue to be deposited on the seabed, with subsequent changes in physio-chemical parameters and biological data.

Chapter 6

Comparing waste dispersal at two farms that employ different feeding methods using a GIS-based modelling approach.

6.1 Introduction

6.1.1 Environmental sustainability

Although the Scottish Executive (2002) determined that levels of fish farm feed and faecal waste would not be a factor that limits future marine cage production of finfish in Scotland, there continues to be a concerted effort to reduce waste outputs and to maintain the environmental sustainability of the fish farming industry. Increased environmental awareness, a better understanding of feeding behaviour (e.g. Blythe *et al*, 1999), better husbandry, feed composition (e.g. Cho and Bureau, 1997) and the fish farmers need to reduce feed costs have all contributed to lowering nutrient loads around fish farms. In the wider context of aquaculture nutrient sustainability, estimation of a lochs' carrying capacity is receiving much attention in the aquaculture community (Scottish Executive, 2002; 2003). The Scottish Executive (2003) define carrying capacity as the "ability of a fjordic loch system to assimilate nutrients........without detrimental effect.......". An assessment of carrying capacity aims to integrate much of what is understood about the effects of marine culture operations with an understanding of natural nutrient flux (productivity, tidal water exchange); the nutrients from agricultural run-off, rivers and other uses to which lochs are subjected; the consequences of these on nutrient flux and potential for algal blooms, for example, and then to set specific sustainable production limits for each water body.

It would be unrealistic to suggest that loch-wide water bodies undergo direct assessment through an Environmental Impact Assessment (EIA) process due to time and cost constraints, although this is done at the farm level under specific legislative (Environmental Impact Assessment (Fish Farming in Marine Waters) Regulations 1999) and EU guidelines (EU Council Directive 97/11/EC, which amended directive 85/337/EEC), with the costs incurred by the fish farmer. Also carrying capacity, using the Scottish Executive definition, considers only the ecological capacity of the water system to cope with a certain level of production and does not explicitly consider an evaluation of social and economic impacts, factors that are implicit in an EIA (GESAMP, 1991). The ecological perspectives of both assessments are similar, however, by predicting then assessing ecological consequences of varying levels of production against acceptable levels of impact and/or pre-determined standards.

6.1.2 Modelling perspectives

Although direct monitoring continues to be the primary source of data, computer models are increasingly being used as a cost-effective alternative to assess likely impacts. Models also provide a "what-if" capability to evaluate different outcomes, to set quality standards and to aid the decision making process (Cuenco, 1989). Characteristics that define modelling types and model development processes, and differences between empirical and theoretical models, are described in detail in Chen (2000). Such models attempt to represent a simplified realism that simulates variables, relationships and processes occurring in the environment via equations that represent the fundamental relationships.

There are currently no comprehensive carrying capacity models in existence, although attempts have been made at integrating many of the processes involved with specific water uses (Duarte *et al*, 2003; Lee *et al*, 2003; Nunes *et al*, 2003), provision of simplified nutrient flux models using tidal flushing data (Gillibrand, 2001; Lee *et al*, 2003) and eutrophication effects (Humborg *et al*, 2000), without defining holistic carrying capacity *per se*. Many of these are so-called "black-box" empirical models in which inputs and outputs are time or site specific and cannot be used to evaluate conditions outside of those seen at the time or at other sites. Fish farmers are particularly interested in the how many fish might be grown within a specific water body, but present models over-simplify the processes involved or simply do not work (Telfer, pers comm.).

6.1.3 Particulate waste dispersion models

There are a number of models that have made progress with certain aspects of fish farming activities that would ultimately feed into the development of wider carrying capacity models. Henderson *et al* (2001) provide a review of the current models available across Europe. Of these the most developed relate to dispersion of dissolved (e.g. Pancheng *et al*, 1997; Doglioli *et al*, 2004) and high nutrient particulate wastes (e.g. Dudley *et al*, 2000; Cromey *et al*, 2002; Perez et al, 2002; Doglioli *et al*, 2004).

Understanding the distribution of particulate waste material is an important function as levels of waste are known to affect various aspects of sediment chemistry and biology, which in turn have potential effects on the wider loch system. In particular, identification of changes in sediment chemical cycling (Gowen and Bradbury, 1987; Weston, 1990; Silvert, 1992; Black *et al*, 1996 Davies *et al*, 1996; Findlay and Watling, 1997; Kempf *et al*, 2002), oxygen availability (Enell and Löf, 1983; Hall *et al*, 1990) and localized alterations in the number and diversity of benthic species (Brown *et al*, 1987; Gowen and Bradbury, 1987; Weston, 1990; Henderson and Ross, 1995; Kempf *et al*, 2002) have been identified. The extent to which the seabed is affected is dependent upon the type and quantity of particulate material being deposited around the fish farm with specific emphasis on nutrient composition, the prevailing currents and subsequent turnover of that sediment by benthic organisms and bacteria. The extent to which these changes are incorporated into modelling packages also varies.

Across Europe the extent to which models developed for aquaculture are used and applied varies widely (Henderson *et al*, 2001). In Scotland, for example, the particle dispersion model DEPOMOD (Cromey *et al*, 2002) is used as part of the EIA process and to estimate the likely seabed deposition of in-feed sea-lice treatments (as AutoDEPOMOD), as part of a licence application process (SEPA, 2001). Many of the particulate dispersion models in use are based on an original concept presented by Gowen *et al* (1989), using simple mass balance calculations to estimate waste levels, a single particle settling velocity and hydrographic data to assess the downward movement and settlement of particles. Subsequent developments include fish growth sub-models to more accurately predict waste quantities (Silvert, 1992; 1994; McDonald *et al*, 1996), assessments of food digestibility data to predict waste quantities (Pereira, 1997), bathymetry variation (Hevia *et al*, 1996) and variable settling velocities for feed and faecal components (Chen *et al*, 1999a; 1999b). DEPOMOD (Cromey *et al*, 2002) incorporates the above activities within particle tracking (dispersion), re-suspension and benthic modules and is used by SEPA, the regulatory authority in Scotland, as part of its statutory regulatory process. More generally the DEPOMOD model is the industry standard in Scotland, against which other models must be compared.

The quality of the modelling is dependant on taking account of as many variables as possible whilst maintaining its functionality and having as close a representation to the actual processes involved as possible. One aspect of fish farming that has a direct impact on the distribution of waste but has not yet been incorporated into any deposition model is the effect of cage movement.

All models currently assume that cages are static, assigned fixed positions within the modelling grid, but this is an unrealistic assumption. As part of their reported fieldwork for model validation Cromey *et al* (2002) suggest that cage movement may account for some of the variation in sediment trap collections, although the amount of movement and its subsequent effect on sedimentation were not known. Cage movement is a phenomenon that in general goes un-noticed at fish farms. This is because there is a lack of solid reference positions or structures close to cages against which to compare. Also cages or cage blocks at a single farm may cover an area of many thousand square meters and movement of a few metres in any one direction does not register. However, notice was drawn to the extent of potential movement during sediment trap studies at Portavadie and Rubha Stillaig sites (Chapter 5). Sediment traps that were set 5m from a cage edge on deployment were under the cage when collected days later, suggesting that movement might not be insubstantial and that particulate waste deposition may also be affected.

Cage blocks and arrays of circular cages are generally moored from multiple anchor points in a grid system that aims to restrict cage movement. The number and tension of the mooring ropes, current speed and direction, wind and wave action, changes in tidal height and gravity (see Beveridge, 1996) will all affect the amount of movement. Goudey *et al* (2001) estimate a 2 – 70 fold decrease in deposition (per m^2) and environmental improvement under cages by using large single point moorings (SPM) that allow cages to move with the wind, tide and prevailing currents and for particulate material to be spread over a greatly increased area of seabed. In Scottish sea lochs there is likely to be a limit on the use of such designs due to the proximity of cages to the shoreline and interference with boating and other water uses. Also, in Scotland, fish farmers are restricted to specific leased areas of seabed that are incompatible with the use of SPM facilities.

It is generally accepted that in low to moderate hydrographic regimes the area most affected by deposition is that directly under the cage array or cage block. It is reasonable to hypothesize that any movement in cage position would result in feed and faeces being spread more thinly over a broader area, as the effective "area under a cage" is increased.

6.1.4 Modelling and GIS

The Institute of Aquaculture has been at the forefront in development of a particulate waste dispersion model integrated within a Geographic Information System (GIS) framework (Perez *et al*, 2001). GIS has long been established as an excellent tool for facility site selection (Church, 2002) using spatial analytical approaches with the overlay of thematic data layers, relating to land function and use, to form an image or graphical output that identifies appropriate sites. This technology is now widely used in aquaculture site selection (Ross, 1998; Nath *et al*, 2000) and is equally relevant for the siting of a range of aquaculture products and structures such as fish, bivalves, ponds or cages (Congleton *et al* 1999; Arnold *et al*, 2000; Gongora, 2003). Scale is an important consideration in the development of site selection and more often will provide map outputs based on high resolution remote sensing at spatial scales of tens of kilometres.

GIS is not a modelling environment but a "computer-based system for the acquisition, storage, analysis and display of geographic data" (Eastman, 1999). However, GIS software is built with the capability to integrate high level programming tools, in order to run new applications, which are then processed with automated spatial assessment and interpolation within a GIS framework. The output, after processing, is the production of raster-based images or other graphical information that reflect the particular application. The system easily handles spatial resolution down to 1m^2 and is therefore an excellent tool for farm level particulate dispersion modelling. Validation of such models with field data is important to ensure the model outputs reflect what is actually happening in the field (GESAMP, 1991) and to establish agreement between observations and predictions.

6.1.5 Modelling Procedure

The GIS model used in this study was based upon original work by Perez (1997) and Perez *et al* (2002) with further development by Brooker (2002). The model was developed to estimate the distribution of sediment carbon and consists of the following key parameters:

- The program for the dispersion model was coded using the high level programming language Pascal (Borland DELPHI 3 software, Borland Software Corporation, California, USA) integrated into IDRISI32 GIS software (Clark Labs, Massachusetts, USA) using the IDRISI Application Programming Interface (API). IDRISI32 spatially assesses and interpolates the data input and generates three pictorial raster-based images estimating the distribution of waste feed, waste faeces and total waste respectively.
- Data for mass balance calculations, cage block generation (including cage co-ordinates, number, orientation, distances between cages) and calculation of the distribution of feed and faecal particles are input via an easy to follow dialogue box (Appendix 5) within a waste dispersion module. This greatly simplifies data entry for the inexperienced user and significantly reduces the overall set-up time. It also limits the potential errors associated with a transfer of data between software packages.
- The program works by dividing the carbon content of feed and faeces, calculated from mass balance, equally between the number of cages specified

and then sub-dividing this quantity between each hydrographic measurement, creating a "packet" of waste to be distributed based on a specific current measurment. Each packet of waste is distributed and deposited within a grid cell based on a random settling velocity, water depth (bathymetry) and time-specific current speed and direction. The starting point for each packet of food and faeces is randomly assigned within the limits of the cage dimensions, assuming that feed input and faecal production by the fish are evenly spread within the confines of the cage. It eliminates the assumption that all particles are produced or distributed from the centre of the cage. Distribution of the particles commences at the net depth, removing the need to correct for differences in water speed inside and outside the cage (Inoue, 1972). Waste particles are dispersed in three dimensions based on the water currents until the seabed is reached, with X and Y components of the distribution calculated using the equations of Gowen *et al*, (1989).

- Separate settling velocities for feed and faecal particles are used with optional application of variability around mean values using a Monte Carlo based simulation technique. At each input a settling velocity is randomly assigned to each packet of waste feed and faeces, from the settling velocity distribution (Chen *et al*, 1999b).
- Variable bathymetry is included by extracting water depths from digital Admiralty Charts covering the modelling area in a 50m x 50m cell grid (each cell = $10m^2$). Half the tidal range is added to the water depth in each grid cell to adjust to mean water depth. The model assumes that water depth is positive with negative values representing height of land above sea level. To allow the model to run correctly all cells containing negative values (i.e. land) in the map are set to zero. The waste packet is dispersed in 1m-depth intervals and a comparison made between this depth and overall water depth indicated on the bathymetric map. When the particle depth is the same as the water depth the iteration stops and the quantity of feed or faeces being modelled at the time is assigned to that grid cell, before the distribution of the next packet of waste begins. Vertical and horizontal resolution on movement is 1m.

- Data assigned to specific grid cells is then interpolated in IDRISI32 using the filters and correction factors applied by Perez *et al* (2002).

6.1.6 Aims of this study

Modelling is increasingly being used to assess environmental impacts and to generate questions that can then be tested through a combined modelling and field approach. The aim of this study is to use a GIS-based dispersion model to assess whether predicted deposition, based on production criteria, is significantly lower at a fish farm site that uses an adaptive feeding system compared that predicted from a traditional hand fed site.

The objectives of this study were to:

1. Assess differences in the settlement of waste particulates under hand feeding and adaptive feeding methods by comparing predictive model outputs from a GIS-based waste dispersion model.
2. Measure the extent to which fish farm cages move as a result of changes in the tide.
3. Incorporate this movement of cages in an updated version of the GIS-based fish farm waste dispersion model and test the hypothesis that cage movement results in a reduction in the peak deposition under fish farm cages by comparing the predicted deposition between the static cage model and the moving cage model.
4. Validate the revised model by comparing predicted deposition of waste from the dispersion model against observed sedimentation under and around fish farms.
5. Contrast the updated Institute of Aquaculture GIS-based waste dispersion model with the industry standard model, DEPOMOD (Cromey *et al*, 2002) with reference to collected sediment trap data.

6.2 Materials and Methods

The approach used incorporated a comparison of fish farm waste dispersion using an updated version of a GIS-based dispersion model (Brooker, 2002) first developed by Perez *et al*, (1997; 2002). Firstly, cage movement was assessed at Portavadie fish farm, the model was updated to incorporate cage movement and comparison was made between the static cage and moving cage versions. Validation of the updated model was made using observed sedimentation of carbon at Portavadie fish farm (chapter 5) compared against predictions from the GIS model. Secondly, the updated model was used to assess implications for waste dispersal under the two feeding regimes. In particular Rubha Stillaig fish farm, which used traditional hand feeding methods, was compared against Portavadie, a site where adaptive feeding technology was used. Finally, DEPOMOD is the industry standard model used in regulation but has undergone only a limited validation in the published literature (Cromey *et al*, 2002). Thus collection of sediment trap data (Chapter 5) provided the opportunity to test DEPOMOD model (version 1.5) predictions with field data and to contrast this industry model with the updated GIS-based model.

6.2.1 Cage movement

The movement of a single 22m-diameter Polar Circle cage was measured on 4 occasions (16th October 2002, 23rd October 2002, 29th October 2002 and 5th November 2002) at Portavadie fish farm on the West Coast of Scotland.

Measurements were taken using a Wild TC1010 Total Station theodolite equipped with a Leica electronic distance-measuring device (Leica AG, Heerbrugg, Switzerland). "Snapshots" were taken of 2 reflectors, one on each side of the cage, every 10 minutes for 8 hours, reflecting the available daylight required for measurement but covering the feeding periods. Two reflectors were used, on opposite sides of the cage, to ensure each side of the cage moved simultaneously and changes in distance were not caused by rotation only.

The data composed a horizontal angle, vertical angle and slope distance from the point of origin on the shore. Data was converted into Eastings (Es) and Northings (Ns) values (in metres) using Leica's LISCAD Plus Surveying and Engineering Environment software version 4.0 (Leica AG, Switzerland and LIStech, Boronia, Victoria, Australia). The first reading on each collection was converted to point (0,0) Es and Ns respectively and each subsequent measurement was relative to this origin.

6.2.2 Comparison between predicted deposition from static cage model and moving cage model

The predicted carbon deposition was assessed using a GIS-based dispersion model. Comparison between the static cage model and moving cage model was based on 15-days of production at Portavadie fish farm, for the period August 16th – 31st 2001.

IDRISI32 limits the number of data points that can be modelled at any one time to a maximum 2200, which relates specifically to the number of hydrographic readings taken. Thus, when modelling full 15-day production (1075 recordings) the model was able to run 2 cages at one time (2 x 1075 = 2150 data points). Thus 6 equal runs were required to complete the 12 cages at the site. In the model, cages are assigned from the centre of the raster-image grid generated (Brooker, 2002), so to avoid the cage positioning of each of the runs overlapping one another, 6 off-set 500m x 500m bathymetric maps were produced for Portavadie, one for each of the required runs, representing the actual position of the cages in relation to the bathymetry. The final outputs, in the form of a 500m x 500m grid, resulted from the addition of the individual raster-images re-sampled using IDRISI32 image processing sub-routines, to create single 500 x 500 images representing the distribution of waste feed, waste faeces and total waste, respectively, for the whole site. Analysis of the raster-images was concentrated on cage 8 and on the transect corresponding to the collection of sediment trap data described in Chapter 5.

6.2.2.1 Mass balance for deposition model runs

A mass balance was used to calculate the amount of carbon waste entering the marine environment from a fish farm (Figure 6.1), with data input via dialogue boxes presented in Appendix 5. The daily quantity of feed added to experimental cage 8 (Figure 2.1) was provided by the fish farmer. Production (= increase in fish growth) in cage 8, between the start and end of the experimental period, was determined from growth curves and feeding algorithms within a CAS Adaptive Feeding System (Aquasmart UK Limited, Inverness), used at the site to distribute and monitor feed intake (Chapter 2). Production in the single cage was 3.964 tonnes (t) for a feed input of 4.360 t giving a feed conversion ratio (FCR) of 1.1. In keeping with other models (e.g. Cromey *et al*, 2002; Perez *et al*, 2002) all cages at the site were assumed to have the same biomass and feed input and model runs used multiples (e.g. 2 x 3.964 tonnes) for mass balance calculations with images added together as described in 6.2.2. Assumed site production during the 15 days was 47.568 t.

Four sizes of feed, supplied by EWOS Limited (Bathgate, Scotland), were used over all experimental periods. Carbon content of sample feed pellets, shown in Table 6.1, was assessed on samples dried in an oven at 90°C for 24 hours and measured using a Perkin Elmer 2400 Series II CHNS/O Autoanalyser with integrated AD-4 Auto-microbalance as described in Chapter 3. During this trial a mixture of small and medium pellets were fed to the fish each day so average carbon content and settling velocity (see below) was incorporated in to the model.

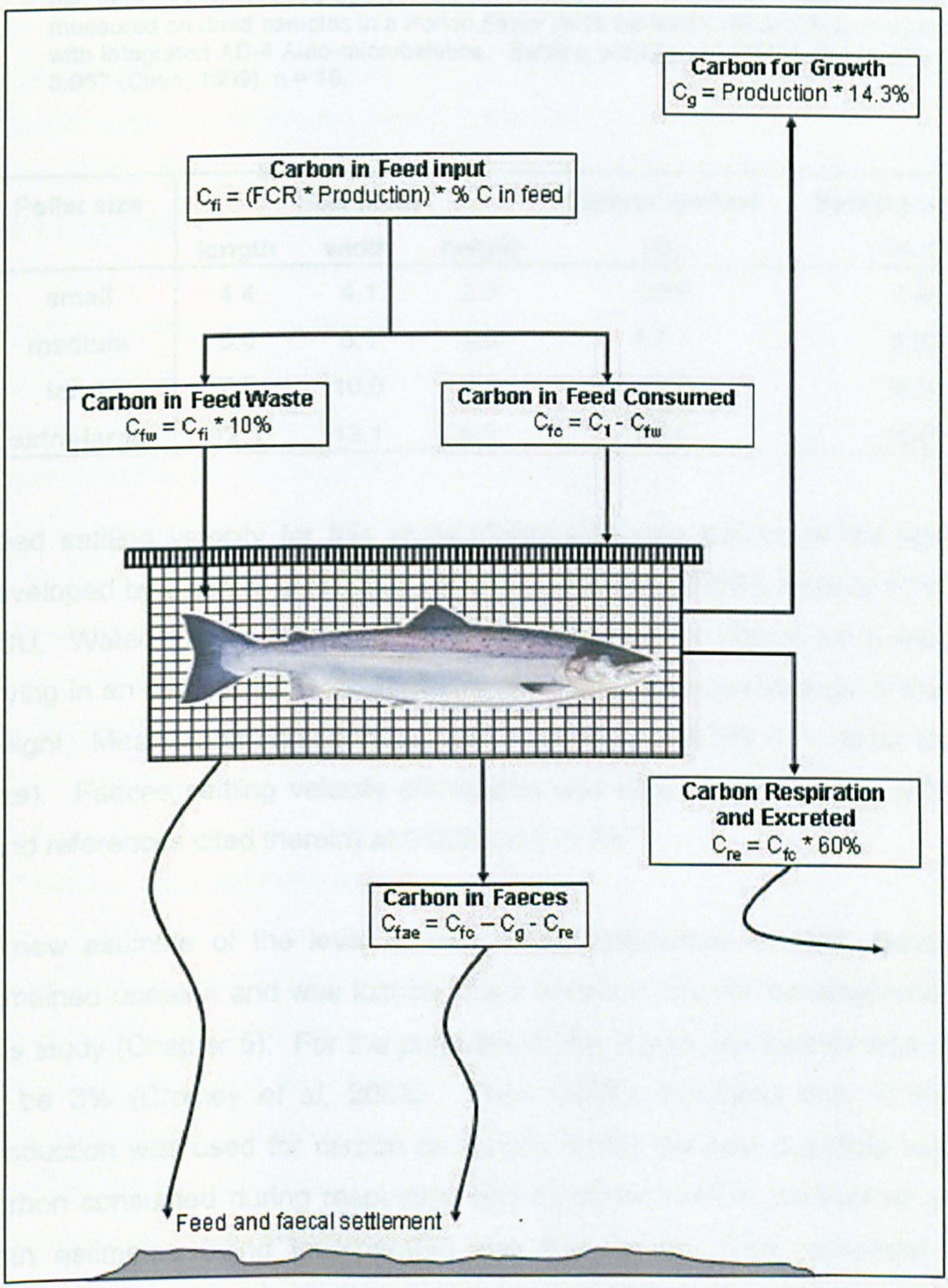


Figure 6.1: Mass balance calculations for carbon waste, generated from unconsumed feed and faecal material, in Atlantic salmon cage culture. Adapted from Perez *et al*, 2002.

Table 6.1: Characteristics of feed pellets used in experimental trials at Portavadie fish farm. Feed supplied by EWOS Limited (Bathgate, Scotland). Carbon content measured on dried samples in a Perkin Elmer 2400 SeriesII CHNS/O Autoanalyser with integrated AD-4 Auto-microbalance. Settling velocity = 0.9125.pellet length + 3.967 (Chen, 1999). n = 10.

Pellet size	Size (mm)			Carbon content	Settling velocity
	length	width	height	(%)	(cm s^{-1})
small	4.4	4.1	2.6	50.8	7.98
medium	5.0	6.1	3.3	51.1	8.53
large	7.5	10.0	6.2	49.5	10.81
extra-large	12.1	13.1	8.5	52.5	15.01

Feed settling velocity for this study (Table 6.1) was based on the relationship developed by Chen *et al* (1999a; 1999b) for standard EWOS diets at 10°C and 33 PSU. Water content of the feed was calculated as the difference in weight after drying in an oven at 90°C for 24 hours, calculated as a percentage of the original weight. Mean water content of all feed sizes used was 5% (n = 10 for each feed size). Faeces settling velocity distribution was taken from Cromey *et al* (2002) (and references cited therein) at 0.032 ± 0.011 ms^{-1}.

A new estimate of the level of feed waste (proportion of food delivered that remained uneaten and was lost as direct waste) could not be determined during this study (Chapter 5). For the purposes of the model, feed waste was assumed to be 3% (Cromey *et al*, 2003). Chen (2000) estimated that 14.3% of the production was used for carbon as growth, whilst the best available estimate of carbon consumed during respiration and excretion is 60% (Gowen *et al*, 1991); both estimates being incorporated into the model. The remainder (carbon consumed - carbon growth - carbon respired/excreted) was assumed to be faeces.

6.2.2.2 Cage movement model

Cage movement was incorporated in to the GIS waste dispersion module as an optional function. Cage movement measurements taken on 23rd October 2002 were used in the model as representative of average movement experienced over the 4 cage movement trials (6.2.1 above).

Data from a single reflector was integrated in to the model as a comma delimited text file (.csv) file with easting and northing values (m) in separate columns, imported from Excel into IDRISI. Resolution on the distance measurements was better than 0.001m although the modelled outputs have a resolution of $1m^2$ based on the grid generation within the model. Cage movement data was therefore rounded to the nearest metre. Cage movement data was extrapolated and collated directly with hydrographic data, such that the time intervals and number of observations (data points) were equal. All remaining cages at both Portavadie and Rubha Stillaig sites were assumed to move by the same amount, having the same anchoring and exposure to wind, wave and tidal effects.

6.2.2.3 Hydrography for model runs

Two Valeport BFM106 direct recording current metres (Valeport, Dartmouth, Devon) were deployed for 1 spring/neap tidal cycle as described in Chapter 2. Data were imported into the model as a single file covering 15-days data, representing the collection periods for sediment trap data in August 2001 at Portavadie fish farm (Chapter 5). Data was imported into the model as comma delimited (.csv) files containing one column for current speed (ms^{-1}) rounded to 3 decimal places (dp) and one for direction (degrees) to 1 dp.

6.2.3 Comparison of waste dispersion at Portavadie (adaptive feeding) and Rubha Stillaig (hand feeding) using a GIS-based modelling approach

Comparison was made between predicted dispersion model outputs for Portavadie and Rubha Stillaig sites, representing the two feeding methods being compared (adaptive feeding and hand feeding respectively). The GIS-based dispersion model cannot take account of the feeding method *per se*, but differences were compared based on the mass balance data included within the model under each of the feeding regimes. All predictive model runs incorporated cage movement. Data was analyzed for full 15-day production runs with two cages run simultaneously with the resultant 6 images added together to form the final images as described in section 6.2.2. Comparison was made on faecal waste only.

6.2.3.1 Mass Balance for comparison model runs

Data from both Portavadie and Rubha Stillaig fish farms were used in the cage movement model. The daily quantity of feed added to experimental cage 8 at Portavadie and cage 11 at Rubha Stillaig was provided by the fish farmer. The estimated increase in fish growth (production) in cage 8 during each of the periods assessed was derived as described in section 6.2.2.1. Data for Rubha Stillaig was estimated from feed conversion ratios and feed input provided by Lighthouse of Scotland Limited. The estimates of FCR provided by Lighthouse included mortalities, so FCR was adjusted (-10%; Fowler, pers. com.) to take account of this. Production, total feed input and FCR for each of the 15-day sampling periods is shown in Table 6.2. All cages at each site were assumed to have the same corresponding biomass and feed input within each period, to estimate site-wide production during each of the trials as shown in Table 6.2.

A combination of feed pellets sizes were used throughout each of the trials as specified in table 6.2. Where mixtures were used the average carbon content and settling velocity (Table 6.1) were incorporated into the model. Remaining mass balance input data is specified in section 6.2.2.1. Hydrography for comparative runs was as described in section 6.2.2.3.

Data extracted from the model runs under the respective cages (cage 8 and cage 11 at Portavadie and Rubha Stillaig respectively) and along the respective transects were standardized per tonne of growth within each of the periods, by dividing the predicted output from the model for each cage by the production figures (in tonnes) within that cage. Production in trial cages during each of the trial dates is specified in table 6.2. The predicted deposition at stations within sites, generated from single runs of the model, were compared between sites using a 2-sample t-test, after checking for normality, to test for significant differences.

Table 6.2: Mass balance data used in waste dispersion model for 15-day trial periods at Portavadie and Rubha Stillaig fish farms.

Trial date	Production Trial cage (kg)	Feed Input (kg)	Feed Size	FCR	Cages (n)	Site production (tonnes)
Portavadie						
August 2001	3964	4360	S / M	1.10	12	47.568
February 2002	2983	3460	L	1.16	12	35.796
April 2002	2814	3152	L / XL	1.12	12	26.208
Rhuba Stillaig						
February 2002	1802	3280	M	1.82	20	36.040
April 2002	2640	4330	M / L	1.64	20	52.800
September 2002	5868	7805	L / XL	1.33	20	117.360

6.2.4 Cage movement model validation

Predicted outputs from the GIS-based model were compared against observed sedimentation measured in the field, for model validation purposes. The method and mass balance data for validation runs was as described in section 6.2.3 (and sections described therein). Validation was conducted for faecal material only with analysis of the raster-images concentrated on cage 8 at Portavadie and cage 11 at Rubha Stillaig and along their respective transects. Accuracy was measured as an absolute value using equation 3 (Cromey *et al*, 2002).

$$\sum (((\text{observed-predicted}) / \text{observed}) * 100) / n \qquad (3)$$

where n = number of observation for all stations.

6.2.5 DEPOMOD model simulations

Simulations were conducted using version 1.5 of the DEPOMOD software, which was kindly provided by Dunstaffnage Marine Laboratory. Model predictions were generated for Portavadie fish farm, based on the feed input for August 2001, February 2002 and April 2002. Feed input per cage per day for model simulations was 282.4kg, 230.6kg and 218.8kg respectively (mean of Table 2.1). Grid generation was created through AutoDEPOMOD and subsequently imported to DEPOMOD v1.5. Grid resolution was set at 25m and simulation runs were conducted using the Partrack module only. As material collected in sediment traps was not subject to subsequent re-suspension, it was not included in model simulations. Outputs, in the form of a contour image, were generated through Surfer™ software, version 7 (Golden Software, Colorado, USA).

The water and carbon content of feed, feed and faecal settling velocity and bathymetry were the same as was used in the IoA GIS dispersion model (section 6.2.2.1). Hydrography was incorporated as a single dataset with current speed and direction averaged from the top and bottom current meters (Chapter 2) and the depth set at 11m (= (water depth-net depth)/2). Currents speed and direction data was averaged over one hour and the length of the hydrographic record was therefore 360 hours. Horizontal dispersion coefficients (k_x and k_y) and vertical dispersion coefficient (k_z) in the turbulence model were set to model default values of 0.1 m^2 s^{-1} and 0.001 m^2 s^{-1} respectively (Cromey *et al*, 2002). Particle starting positions were at mid-depth in the cage centre and trajectories were calculated every 60s. As the model was validated for faecal output only (Cromey *et al*, 2002), direct feed losses were assumed to be zero with 100% of the food ingested. Water content and digestibility was 5% and 85% respectively and thus 14.25% of the food fed was associated with faecal particles. The number of particles in the model was set at the model default value ($10*10^4$, Cromey *et al*, 2002).

The estimated faecal deposition for each period (After Cromey *et al*, 2002), in g C m^{-2} yr^{-1}, was scaled to g C m^{-2} 15-$days^{-1}$ for comparison against sediment trap data (Chapter 5). In the event that sediment traps did not sit directly on a grid node, then a deposition value was interpolated from surrounding nodes (Cromey *et al*, 2002). Accuracy was measured using equation 12.

6.3 Results

6.3.1 Cage movement

Cage movement data collected on the 5th November 2002 was rejected due to poor light resulting in less than 8 hours of data being collected. The extent of the movement on the remaining dates is shown in Figure 6.2.

The position of the measuring device varied between each of the trial dates and the starting position of the cages was arbitrarily set at (0,0) for each data collection. The important feature was the extent of the movement overall, on each of the dates. Maximal variation occurred on 29th October at 10.1m and 7.7m, easting and northing, respectively, when tidal range was 1.67m. Tidal range on all dates was broadly similar (1.61m and 1.87m on 16th and 23rd respectively) but the wind on the 29th was stronger and may account for the higher movement during this period, although wind speed and direction was not measured. Wind on other days was negligible. Overall the movement of the cages was random, depending on the state of the tide.

The area under the cage received the highest deposition of waste feed and faeces (See Chapter 5). Figure 6.3 shows the increase in this area as a result of measured cage movement on 23rd October 2002. The "area under the cage" was increased by 72% from 380m^2 to 655m^2. The spatial starting position and relative settlement position of waste feed and faecal material within the cage would therefore vary with the rise and fall of the tide and changes in wind direction and speed. This is not presently taken into account in available predictive fish farm waste dispersion models.

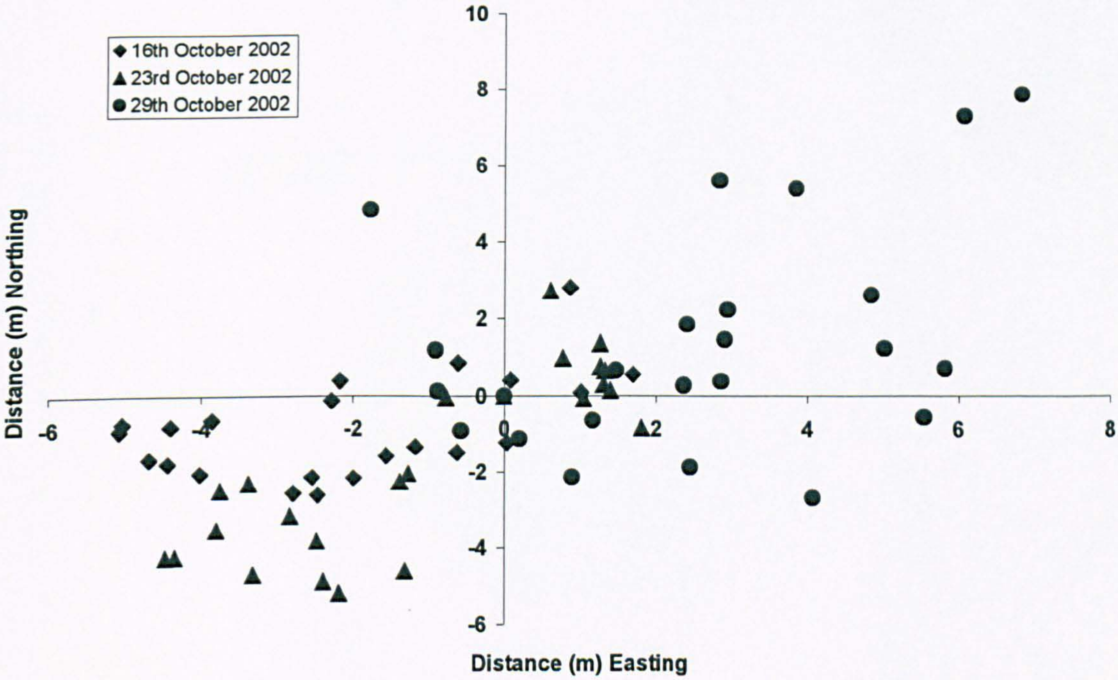


Figure 6.2: Position of the edge of a 22m-diameter Polar Circle marine cage relative to first reading, set to (0,0) on each of the trial dates; measured using a theodilite (see text 6.2.1) every 20 minutes over 8-hours.

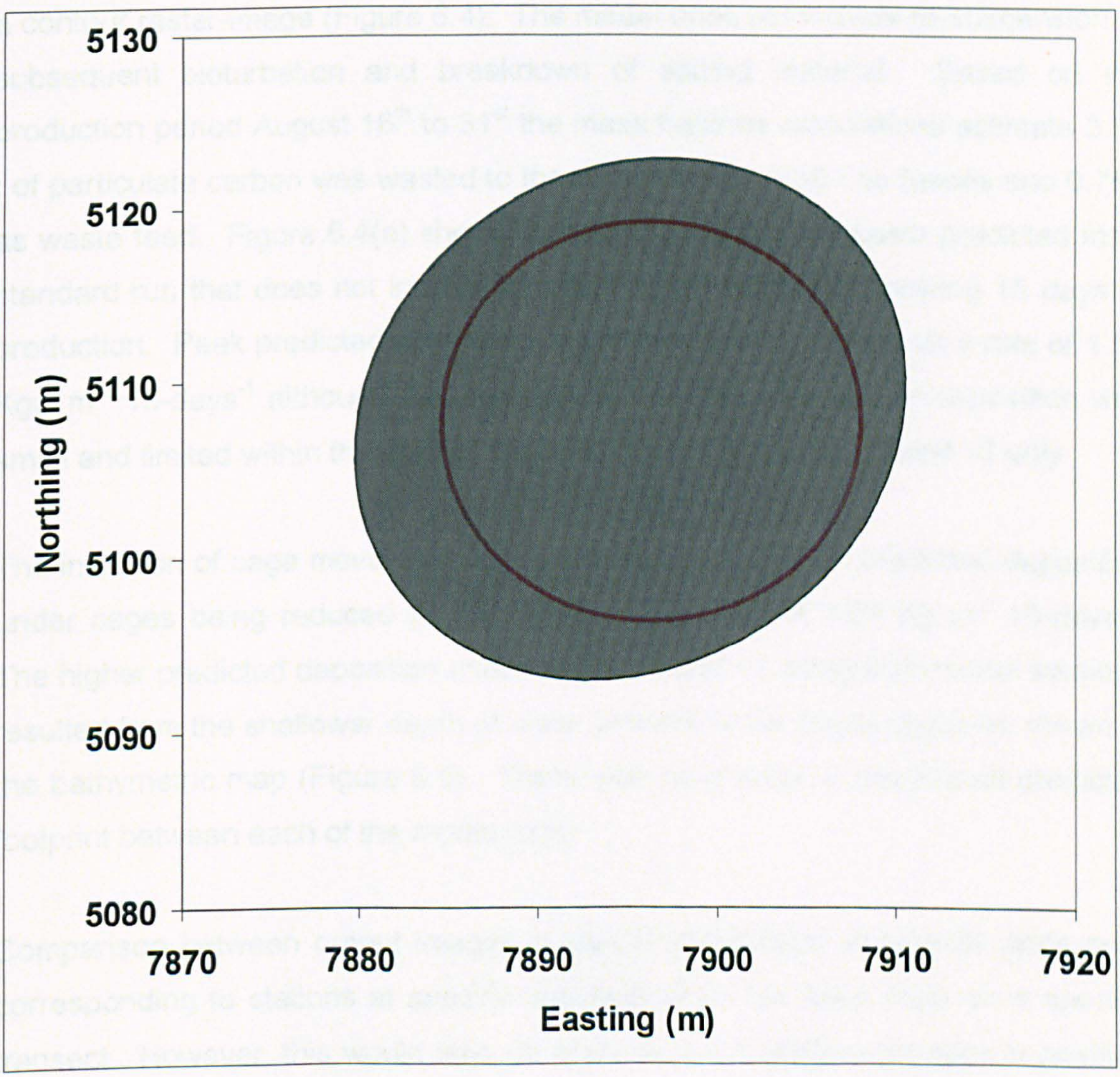


Figure 6.3: Representation of the additional area of seabed covered by a 22m-diameter Polar Circle marine cage as a result of measured movement of the cage on 23rd October 2002. Red circle represents cage starting position.

6.3.2 Comparison between predicted deposition from static cage model and moving cage model

The modelled output of predicted carbon settlement to the seabed is in the form of a contour raster-image (Figure 6.4). The model does not include re-suspension or subsequent bioturbation and breakdown of settled material. Based on the production period August 16th to 31st the mass balance calculations estimate 3.84 t of particulate carbon was wasted to the environment, 3.06 t as faeces and 0.78 t as waste feed. Figure 6.4(a) shows the distribution of total waste predicted for a standard run that does not incorporate cage movement and covering 15 days of production. Peak predicted deposition occurred under the cages at a rate of 1.55 KgC m^{-2} 15-days^{-1} although the area affected by this high rate of deposition was small and limited within the area of seabed covered by cages 11 and 12 only.

The inclusion of cage movement within the model resulted in predicted deposition under cages being reduced (Figure 6.4(b)) to a peak of 1.07 Kg m^{-2} 15-days^{-1}. The higher predicted deposition under cages 11 and 12 using both model versions resulted from the shallower depth of water present under these cages as shown in the bathymetric map (Figure 6.5). There was no change in the overall predicted footprint between each of the model runs.

Comparison between output images could be undertaken at specific grids cells corresponding to stations at specific distances from the cage edge on a specific transect. However, this would take no account of the relative changes in position of the stations, relative to the cage, as a result of the movement experienced by cages. Thus, Table 6.3 shows the average deposition within a 7m-diameter from the cage centre starting position and 4.5m-diameter around other stations along the transect, to take account of the relative movement.

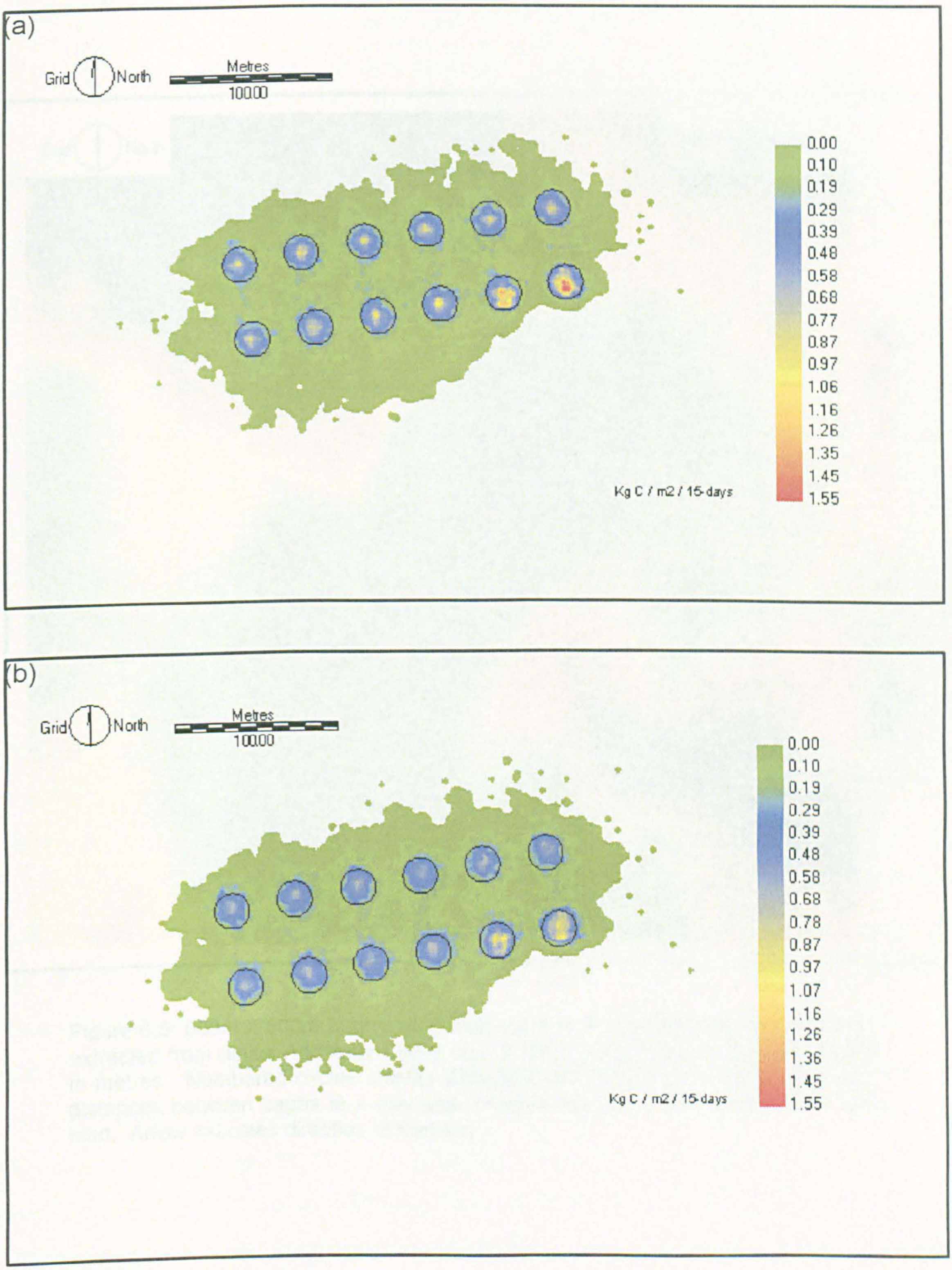


Figure 6.4: Contour rastor-image for Portavadie fish farm showing predicted total carbon settlement to the sediment, using GIS dispersion model, for the production period August 16th – 31st 2001. (a) static cages model (b) moving cages model. Production = 46.08 t, Feed Conversion Ratio = 1.1.

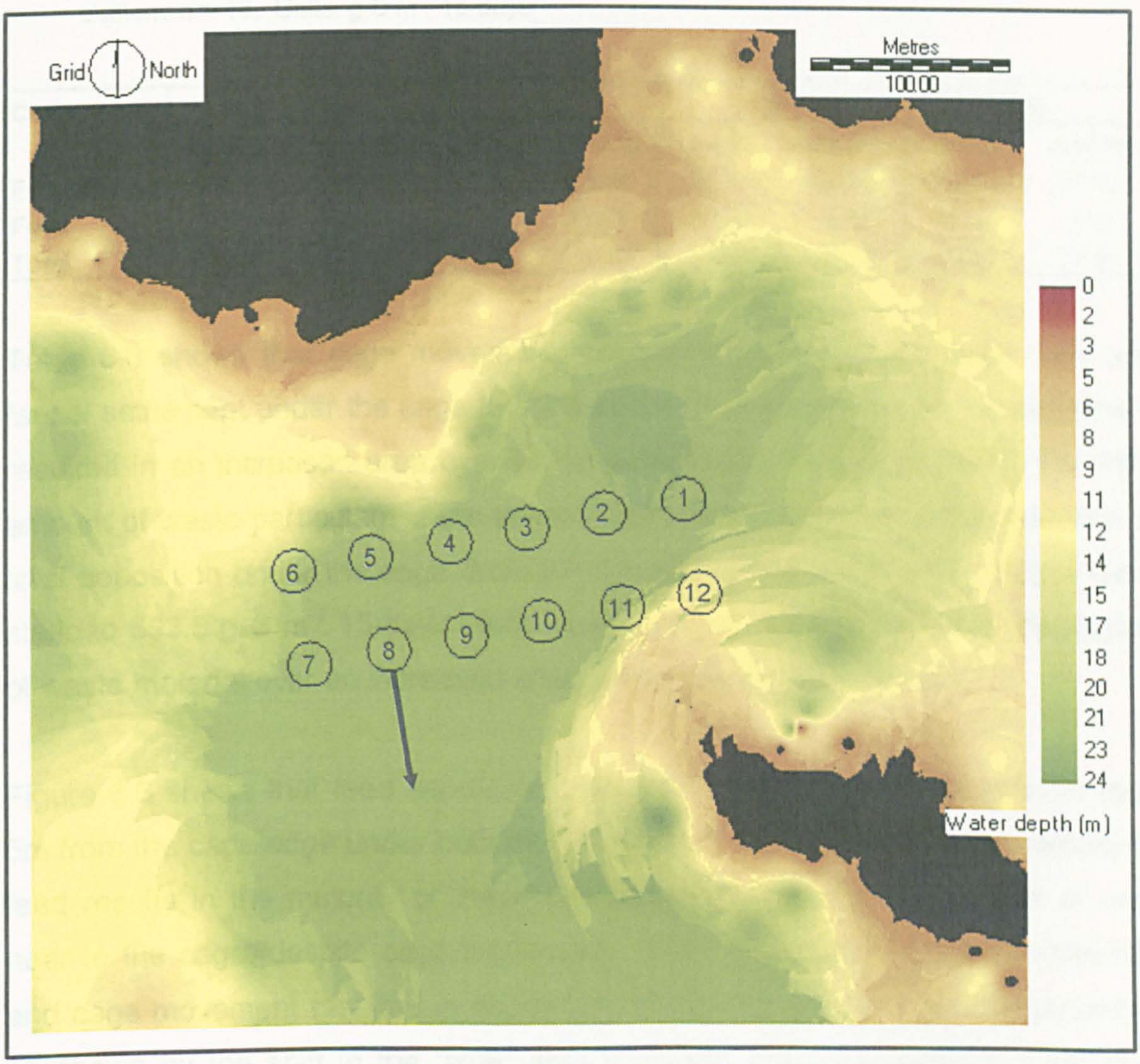


Figure 6.5: 500m x 500m bathymetric map showing Portavadie fish farm location, extracted from digital Admiralty Charts plus 2.16m to adjust for mean water depth in metres. Numbered circles identify 22m-diameter Polar Circle cages, with size, distances between cages in a row and between rows to scale. Black areas are land. Arrow indicates direction of transect.

Table 6.3: Average predicted deposition under and at specified distances from cage 8 at Portavadie fish farm. Predictions from rastor-images generated using GIS dispersion model, with model runs assuming cage were static and moving. Based on production and mass balance calculations for the period August 16th – 31st 2001. Number of $1m^2$ cells averaged under cage (n) = 38, at remaining stations n = 16. Units: g C m^{-2} 15-$days^{-1}$.

Component	Under cage		5m		15m		25m	
	static	moving	static	moving	static	moving	static	moving
Faeces	480.71	426.60	115.04	129.04	59.71	58.76	74.01	27.45
Feed	216.81	166.89	38.77	21.81	1.94	1.04	0.23	0.19
Total	679.51	593.50	153.81	150.85	61.65	59.80	74.24	27.65

Table 6.3 shows that cage movement reduced the average predicted feed and faecal settlement under the cage by 23% and 11% respectively, as the movement resulted in an increased area over which particulates were deposited. The total amount of waste particulate is the same in both model outputs so the reduction in total deposition under the cage, from 679.51 g C m^{-2} 15-$days^{-1}$ when cages were static to 593.5 g C m^{-2} 15-$days^{-1}$ with moving cages, reflects the wider dispersion of waste material over an increased area, as suggested in Figure 6.3.

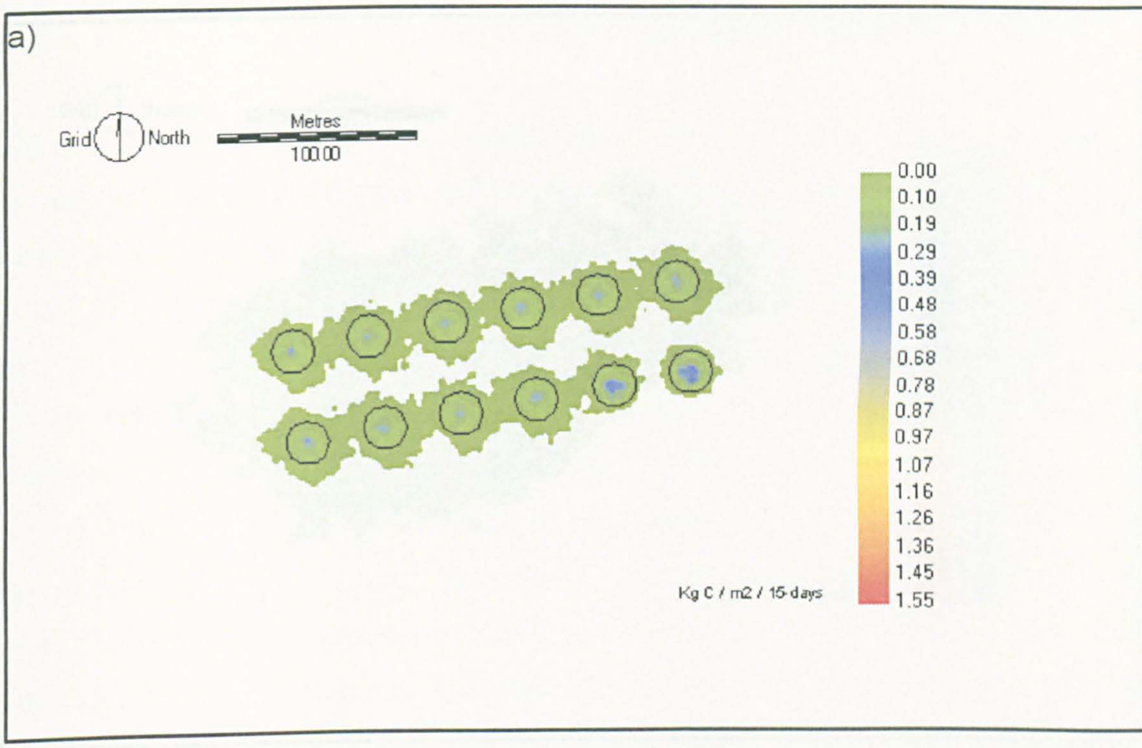
Figure 6.6 shows that feed deposition had little impact at distances greater than 5m from the cage edge under both model versions. The higher settling velocity of feed results in the majority of these particulates being deposited under or very near to the cage, despite cage movement. The combination of current direction and cage movement resulted in deposition increasing slightly in a NNE direction, as shown by the shift in the "blue" area in Figure 6.4(b), representing the total deposition (feed + faeces). This explains why the feed component of settlement at 5m distance decreased (Table 6.3), which appeared to deposit feed on the opposite side of the cage in a SSE direction, as shown in Figure 6.6. The faecal component increased at the 5m station (Table 6.3) and results from the lower settling velocity for faeces, allowing time in the model for the quantity that would have previously been predicted for deposition under the cage to be spread more evenly in all directions despite the cage movement. This is reflected in the lack of a difference in the faecal components of the two model outputs, shown in Figure 6.7.

Table 6.3 shows a large decrease in predicted faecal deposition with the moving cage model, compared to the static cage model, at the 25m station. This was thought to be an artefact of the interpolation process within the IDRISI32 software. The number of data points at this distance from the cage would be fewer and patchier than positions nearer to the cage, where settlement would fill more grid cells. IDRISI has fewer points between which to interpolate and as a result predicted deposition can vary. This was potentially exacerbated by differences in the random starting position and settling velocity applied to the particular packages of waste between the model runs, combined with cage movement that resulted in an increased distribution in a NNE direction.

The reduction in deposition under the cage, which given the same total deposition from the mass balance calculations results in an increase in deposition outside of the cage area, is also indicated by the changes in deposition under the whole of cage 8 as shown in Table 6.4 where faeces, feed and total deposition was reduced as a result of cage movement. Predicted deposition was calculated by applying a mask over the cage dimensions in IDRISI and adding each grid cell together in Microsoft Excel™.

Table 6.4: Total (g C 383-m^{-2}) and average (g C m^{-2}) predicted settlement under experimental cage 8 at Portavadie fish farm. Predictions from raster-images generated using GIS dispersion model assuming static and moving cages, based on production and mass balance for the period August 16th – 31st 2001. Polar Circle cage size 22m diameter, representing 383m^2.

Component	Static		Moving	
	total	average	total	average
Faeces	117.84	0.307	112.24	0.293
Feed	41.70	0.109	37.26	0.097
Total	159.53	0.417	149.51	0.390



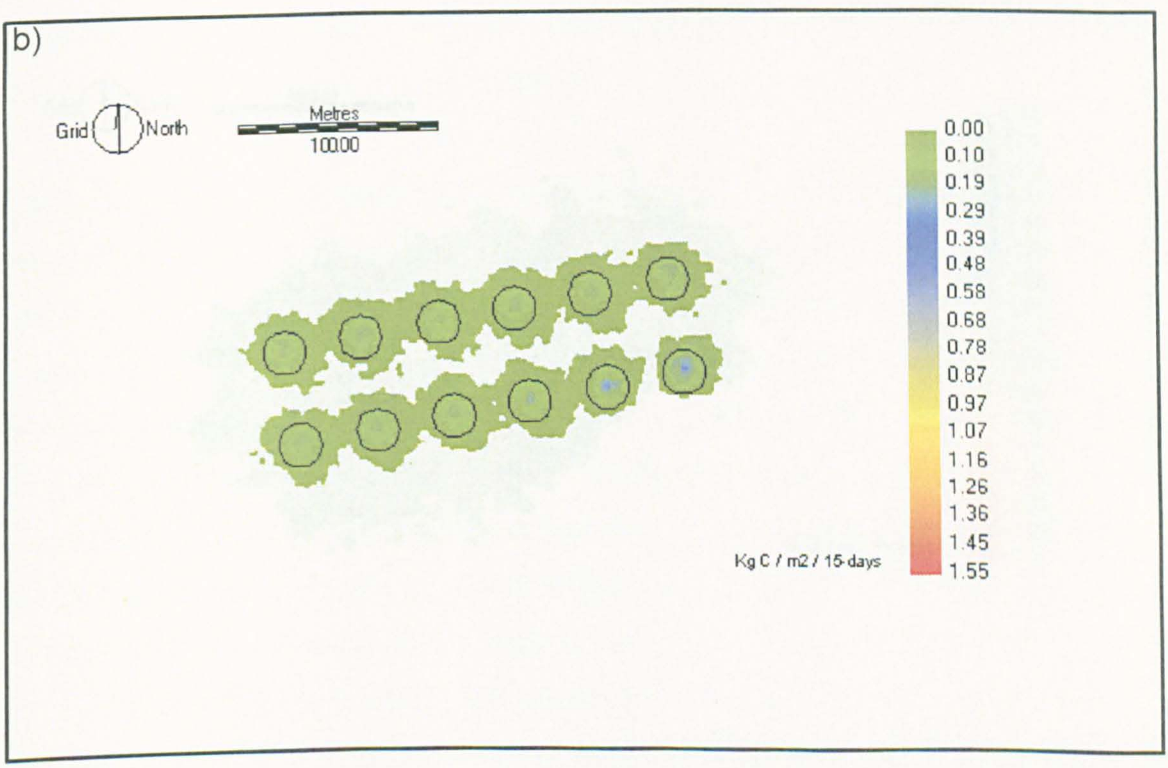


Figure 6.6: Contour rastor-image for Portavadie fish farm showing predicted feed carbon settlement to the sediment, using GIS dispersion model, for the production period August 16th – 31st 2001. (a) static cages model (b) moving cages model. Production = 46.08 t, Feed Conversion Ratio = 1.1. Assumed feed waste = 3% (0.78 tonnes).

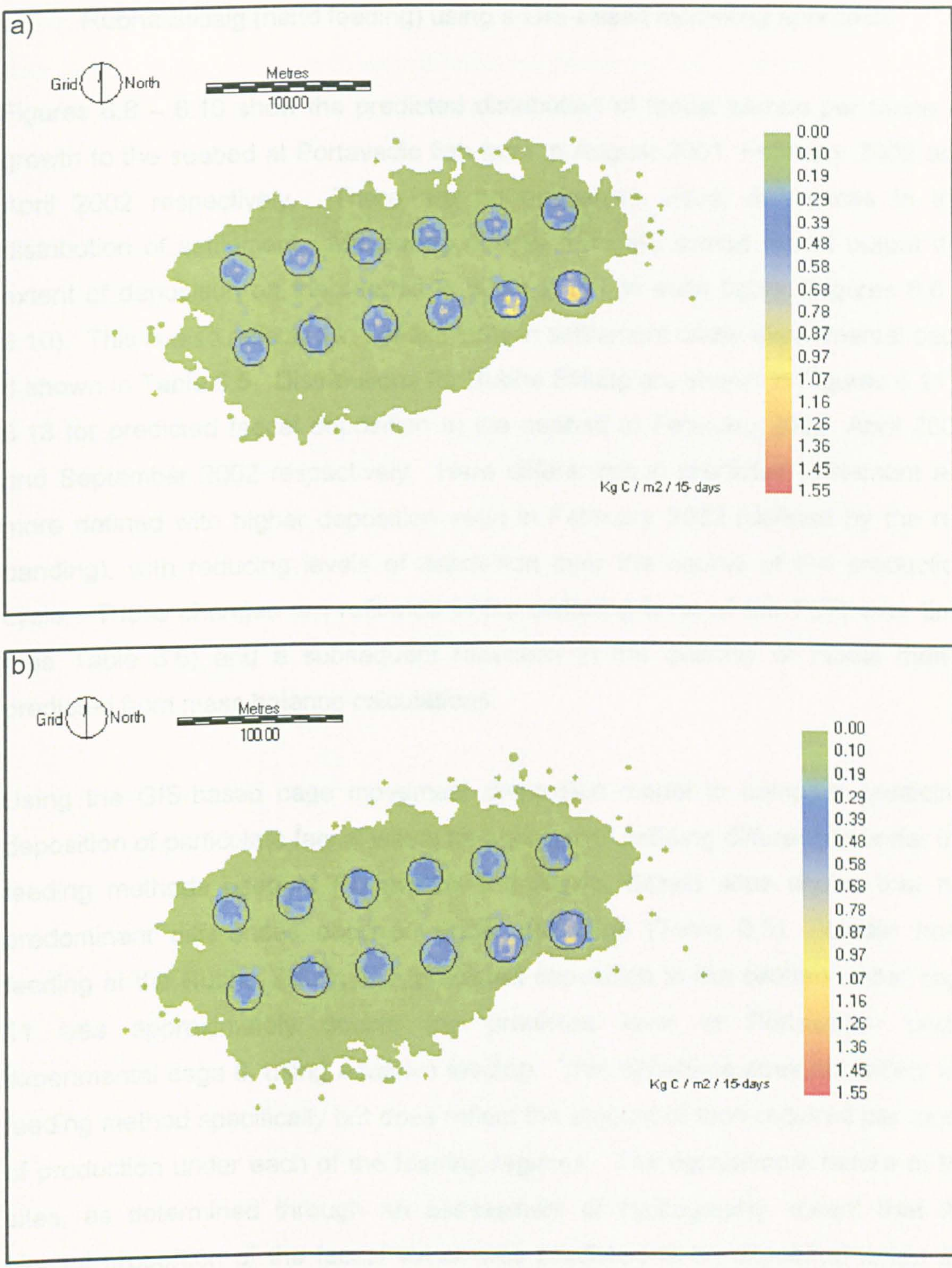


Figure 6.7: Contour rastor-image for Portavadie fish farm showing predicted faecal carbon settlement to the sediment, using GIS dispersion model, for the production period August 16th – 31st 2001. (a) static cages model (b) moving cages model. Production = 46.08 t, Feed Conversion Ratio = 1.1. Assumed faecal waste = 3.06 tonnes.

6.3.3 Comparison of waste dispersion at Portavadie (adaptive feeding) and Rubha Stillaig (hand feeding) using a GIS-based modelling approach

Figures 6.8 – 6.10 show the predicted distribution of faecal carbon per tonne of growth to the seabed at Portavadie fish farm in August 2001, February 2002 and April 2002 respectively. There are no noticeable visual differences in the distribution of settlement. Measuring directly from the scaled model output the extent of deposition on the seabed is 302m x 151 in each figure (Figures 6.8 – 6.10). This is also reflected in the similarity in settlement under experimental cage 8 shown in Table 6.5. Distributions for Rubha Stillaig are shown in Figures 6.11 – 6.13 for predicted faecal deposition to the seabed in February 2002, April 2002 and September 2002 respectively. Here differences in predicted settlement are more defined with higher deposition seen in February 2002 (defined by the red banding), with reducing levels of deposition over the course of the production cycle. These changes are reflected in the reducing level of the FCR over time (see Table 6.6) and a subsequent reduction in the quantity of faecal matter predicted from mass balance calculations.

Using the GIS-based cage movement dispersion model to compare predictive deposition of particulate faecal waste as a means of defining differences under the feeding methods used at Portavadie and Rubha Stillaig sites shows that the predominant differences occurred under the cage (Table 6.5). Under hand feeding at the Rubha Stillaig site, predicted deposition to the seabed under cage 11 was approximately double the predicted level at Portavadie, under experimental cage 8, using adaptive feeding. This difference does not reflect the feeding method specifically but does reflect the amount of food required per tonne of production under each of the feeding regimes. The depositional nature of the sites, as determined through an assessment of hydrography, meant that the highest proportion of the faecal waste was predicted to be deposited under the cage. Using a 2-sample t-test to compare the predicted settlement (g C m^{-2} 15-$days^{-1}$ t^{-1}) under the cages between sites, however, showed the confidence interval between the samples was large and the sample size was low resulting in no significant difference between sites at the stations under the respective cages

(T = -2.36, df = 2, p = 0.142). At remaining stations there was a higher degree of similarity between the sites (p = >0.20) so that overall the comparative modelling approach identified no significant differences between the adaptive feeding and hand feeding regimes based on the respective food inputs and feed conversions seen.

Table 6.5: Predicted deposition of faecal waste material standardized per tonne of production. Predictions from raster-images generated using a GIS dispersion model, incorporating cage movement and based on mass balance for 15-days production. Station distance = distance from cage edge (m). Number of cells in raster-images averaged under cage (n) = 38, at remaining stations n = 16. Units g C m^{-2} 15-$days^{-1}$ t^{-1}.

Collection	Under Cage	5m Station	15m Station	25m Station
Portavadie				
August	111.09	33.60	15.30	7.15
February	101.52	43.71	16.66	7.93
April	114.77	21.84	13.86	14.12
Average	**109.13**	**33.05**	**15.27**	**9.73**
Rubha Stillaig				
February	280.10	33.02	20.71	11.52
April	203.90	33.66	23.39	18.20
September	137.30	17.94	12.79	13.40
Average	**207.10**	**28.21**	**18.96**	**14.37**

In more general terms the contour images at Rubha Stillaig show a varying predicted distribution during each of the model runs, as indicated the maximum east-west distance, being 253m, 219m and 200m in Figures 6.11, 6.12 and 6.13 respectively. The constriction in dispersion on the eastern side of the Rubha Stillaig cages reflects the shallower water depths on that side. This distribution is also much broader than at Portavadie, where the maximum east-west distance is 151m (e.g. Figure 6.8). Differences in dispersion between Portavadie and Rubha Stillaig sites result from a combination of increased levels of predicted faecal waste and slightly deeper water present at the Rubha Stillaig site. Within the model, the increased depth (bathymetry) at Rubha Stillag increases the horizontal distribution of the waste, subject to the hydrographic regime.

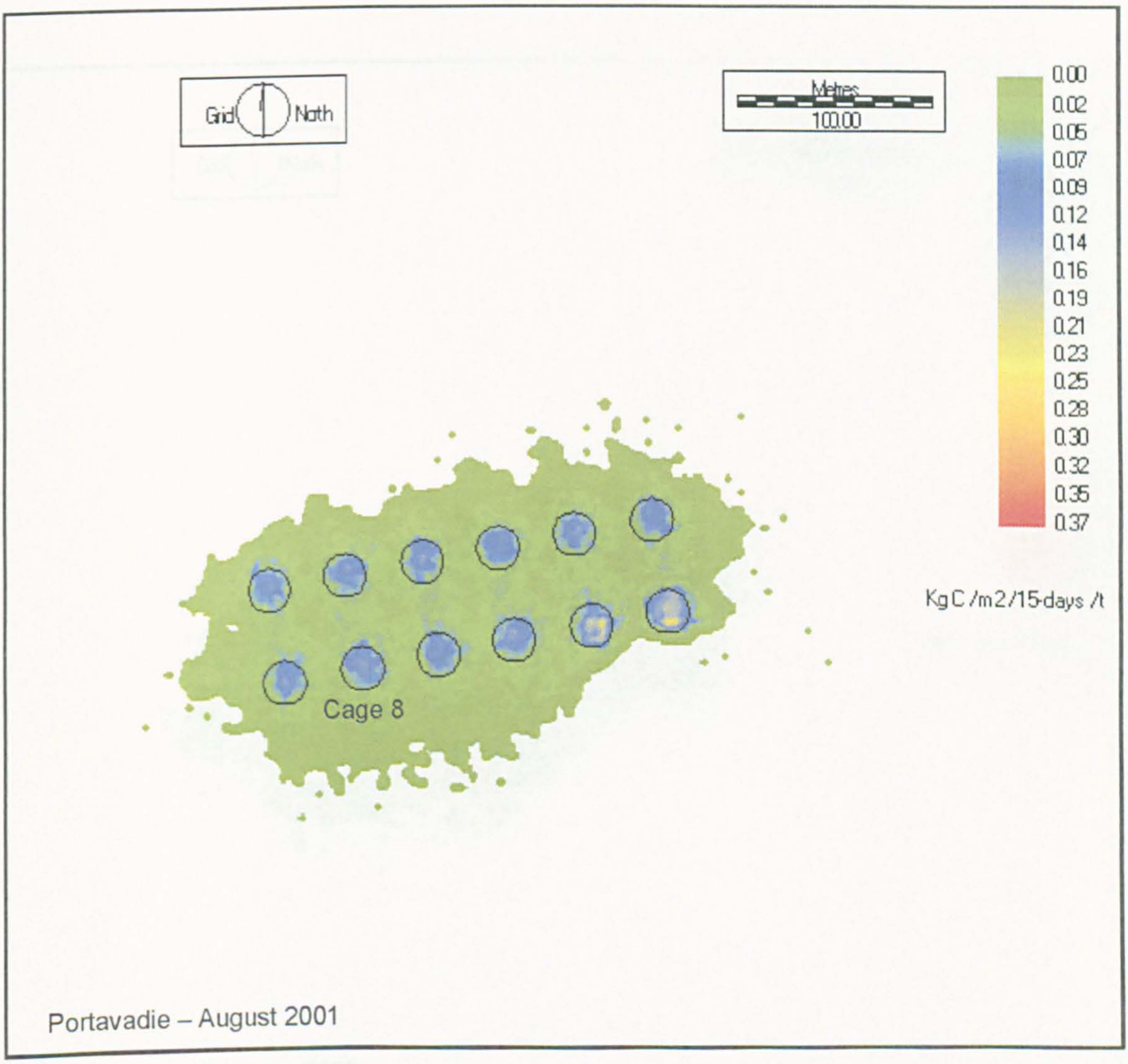


Figure 6.8: Contour rastor-image for Portavadie fish farm showing predicted faecal carbon settlement to the sediment per tonne of production, using GIS dispersion model (cage movement version), for August 2001.

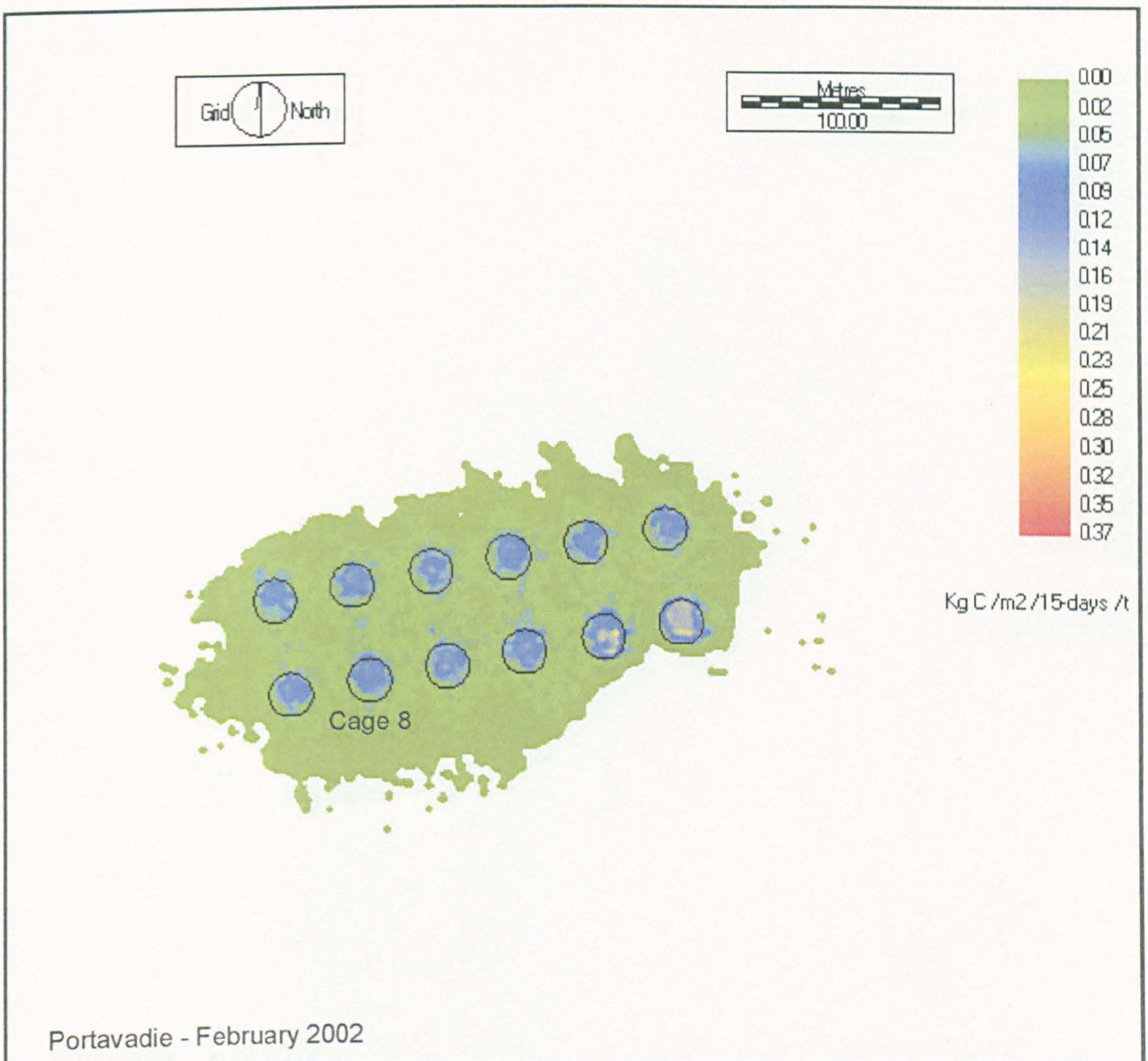


Figure 6.9: Contour rastor-image for Portavadie fish farm showing predicted faecal carbon settlement to the sediment per tonne of production, using GIS dispersion model (cage movement version), for February 2002.

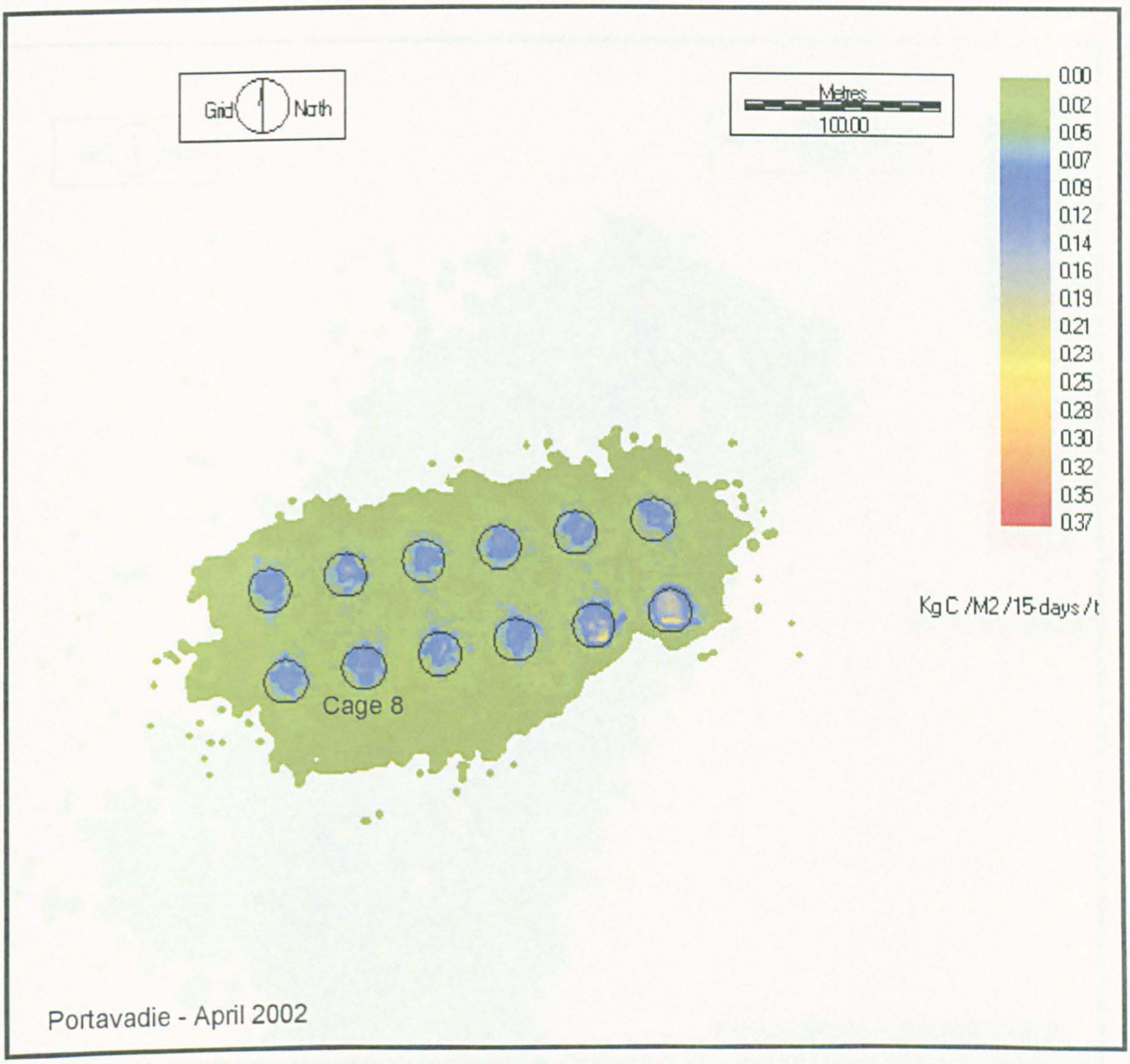


Figure 6.10: Contour rastor-image for Portavadie fish farm showing predicted faecal carbon settlement to the sediment per tonne of production, using GIS dispersion model (cage movement version), for April 2002.

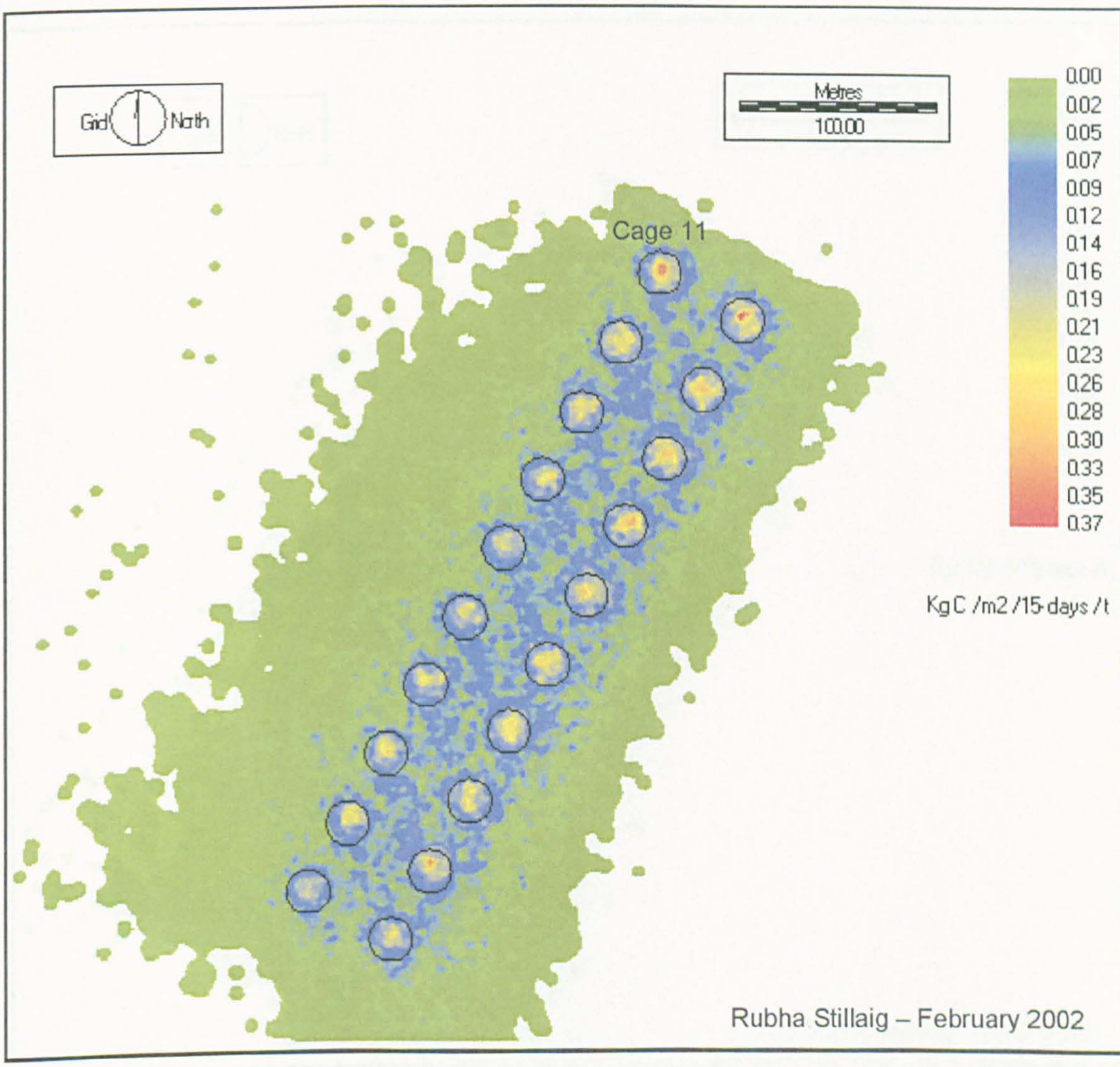


Figure 6.11: Contour rastor-image for Rubha Stillaig fish farm showing predicted faecal carbon settlement to the sediment per tonne of production, using GIS dispersion model (cage movement version), for February 2002.

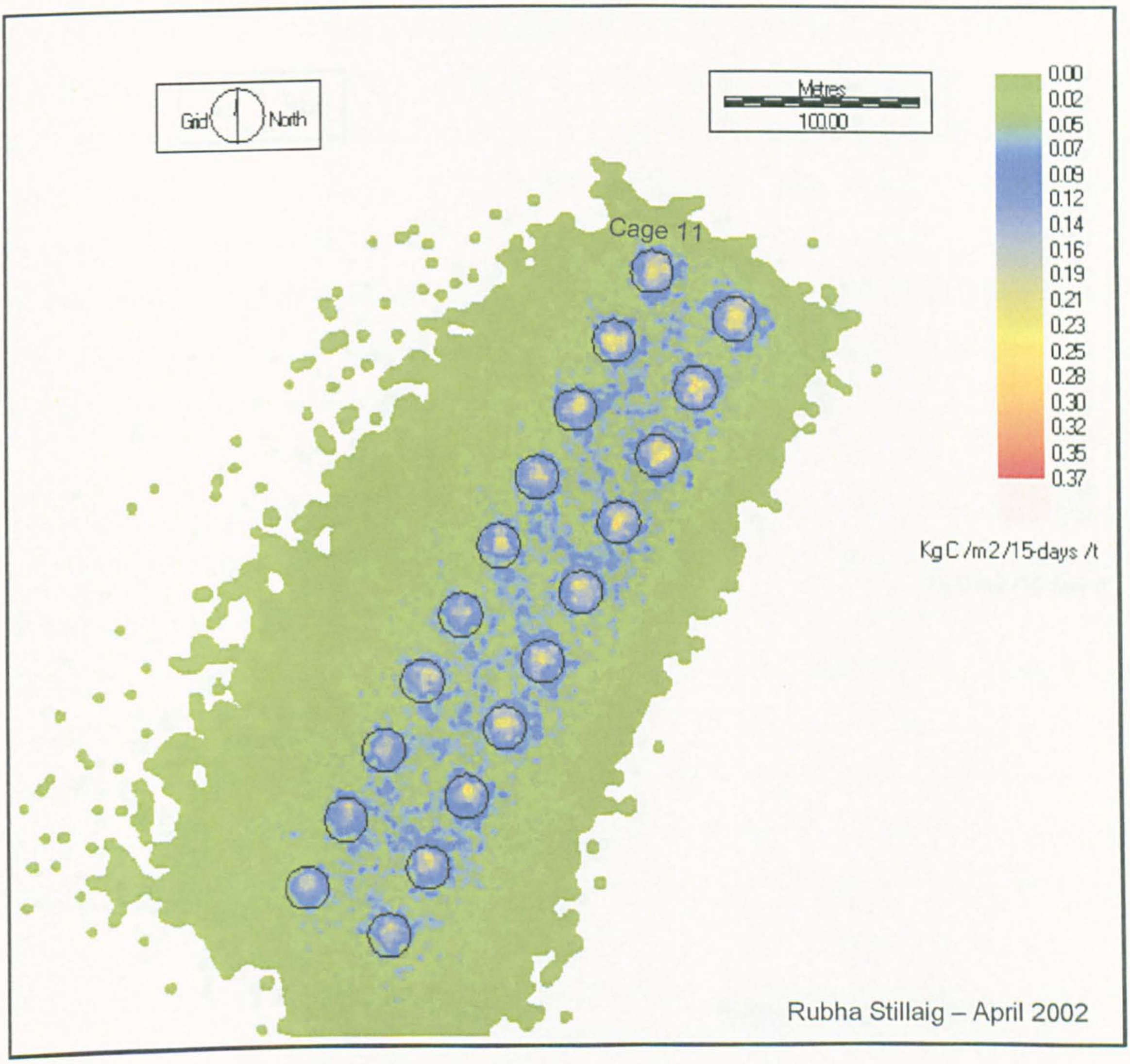


Figure 6.12: Contour rastor-image for Rubha Stillaig fish farm showing predicted faecal carbon settlement to the sediment per tonne of production, using GIS dispersion model (cage movement version), for April 2002.

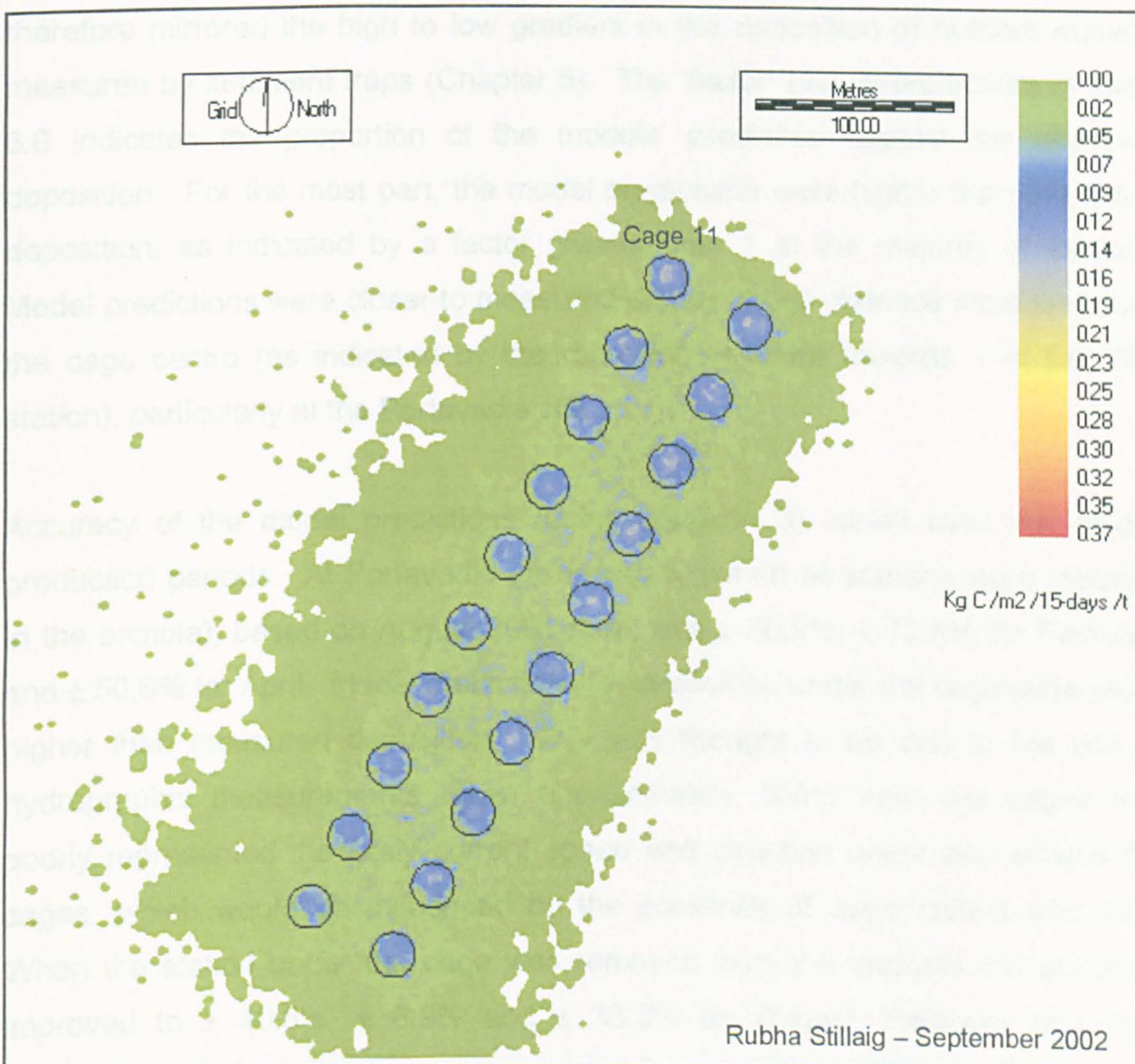


Figure 6.13: Contour rastor-image for Rubha Stillaig fish farm showing predicted faecal carbon settlement to the sediment per tonne of production, using GIS dispersion model (cage movement version), for September 2002.

6.3.4 Cage movement model validation

Validation of the cage movement GIS dispersion model was assessed against the sedimentation of particulate faecal material only. The faecal deposition predicted using the model was shown to be high under the cage and to reduce with increased distance from the cage edge up to 25m (Table 6.6). The predictions therefore mirrored the high to low gradient in the deposition of nutrient material measured by sediment traps (Chapter 5). The 'factor' (actual/prediction) in Table 6.6 indicates the proportion of the models' prediction against the observed deposition. For the most part, the model predictions were higher than the actual deposition, as indicated by a factor greater than 1 at the majority of stations. Model predictions were closer to measured deposition as distance increased from the cage centre (as indicated by the reduction in factor towards 1 at the 25m station), particularly at the Portavadie site.

Accuracy of the model predictions (using equation 3) varied over the 15-day production periods. At Portavadie the accuracy (when all stations were included in the sample), based on August 2001 data, was ± 50.9%, ± 72.8% for February and ± 50.6% for April. Model predictions for deposition under the cage were much higher than measured deposition. This was thought to be due to the use of hydrographic measurements taken approximately 100m from the cages that poorly represented the likely current speed and direction under and around the cages, which would be influenced by the presence of cage collars and nets. When the station under the cage was removed from the analysis the accuracy improved to ± 40.5%, ± 8.9% and ± 33.3% for August, February and April respectively. Summarizing the data for Portavadie resulted in an average predictive accuracy of ± 58.1%, when all stations were included, improving to ± 27.6% when the over-predictions under the cage were removed.

Table 6.6: Comparison of 15-day observed verses predicted faecal particulate carbon deposition for model validation. Actual deposition measured using sediment traps at stations along a transect from cage 8 at Portavadie and cage 11 at Rubha Stillaig, collected every 3-days over a 15-day period each month. Predictions from raster-images generated using a GIS dispersion model, incorporating cage movement and based on mass balance for 15-days production in tonnes. FCR = Feed Conversion Ratio. Station distance = distance from cage edge (m). Factor = actual/predicted. Number of cells in raster-images averaged under cage (n) = 38, at remaining stations n = 16. Units: g C m^{-2} 15-$days^{-1}$.

Collection	Prod'n	FCR	Under Cage			5m Station			15m Station			25m Station		
Portavadie			**Actual**	**Predicted**	**Factor**	**Actual**	**Predicted**	**Factor**	**Actual**	**Predicted**	**Factor**	**Actual**	**Predicted**	**Factor**
August	3.84	1.10	234.27	426.60	1.82	75.75	129.04	1.70	41.04	58.76	1.43	29.79	27.45	0.92
February	3.06	1.16	85.20	310.66	3.65	120.82	133.75	1.11	55.61	50.97	0.92	22.54	24.26	1.08
April	2.82	1.12	159.64	323.29	2.03	109.50	61.59	0.56	61.73	39.08	0.63	49.46	39.83	0.81
Average			**159.70**	**353.52**	**2.50**	**102.02**	**108.13**	**1.12**	**52.79**	**49.60**	**0.99**	**33.93**	**30.51**	**0.93**
Rubha Stillaig														
February	1.80	1.64	167.25	504.09	3.01	23.55	59.43	2.52	10.09	37.28	3.69	8.81	20.73	2.35
April	2.70	1.48	112.69	550.44	4.88	23.06	90.88	3.94	9.94	63.16	6.35	13.73	49.13	3.58
September	6.23	1.20	117.84	855.30	7.26	37.99	111.79	2.94	36.11	79.67	2.21	26.10	83.74	3.21
Average			**132.59**	**636.61**	**5.05**	**28.20**	**87.37**	**3.14**	**18.71**	**60.04**	**4.09**	**16.21**	**51.20**	**3.05**

Accuracy in model predictions for Rubha Stillaig was poor, with all factors above 2.5. Summarizing the data for Rubha Stillaig resulted in an average predictive accuracy of ± 256.6%, when all stations were included. Higher FCRs at Rubha Stillaig resulted in higher levels of predicted faecal waste, generated through the mass balance calculations. However, this higher predicted deposition was not supported by the observed deposition at the site. FCR is fundamental to the level of predicted waste through the mass balance calculations and given the reasonable accuracy of the model using Portavadie data, it suggests that the FCRs' for Rubha Stillaig were over-estimated.

6.3.5 DEPOMOD model simulations

Figures 6.14, 6.15 and 6.16 show the DEPOMOD model predictions for annual deposition of waste faecal material based on the food input during August 2001, February 2002 and April 2002 respectively at Portavadie fish farm. Model predictions ranged from 10g C m^{-2} yr^{-1} to 2500g C m^{-2} yr^{-1} directly under the cages. The contours on all outputs show the 365 and 700 g C m^{-2} yr^{-1} limits of deposition, representing the equivalent of 1 and 2g C m^{-2} d^{-1}. The closeness of the lines suggests a rapid reduction in particulate settlement with distance from the cages and reflects the depositional nature of the site. There is no displacement of faecal waste in any particular direction in line with the lack of residual current in any direction. It was unclear why the 10g C m^{-2} yr^{-1} contour partly covers land, but may be due to the large grid cell resolution (25m x 25m) and/or a failure in the model to recognize heights above 0m.

Table 6.7 provides a comparison between observed faecal deposition and predicted faecal carbon deposition from the DEPOMOD dispersion model and includes associated factors that indicate the proportion of the model prediction to the observed deposition. Predicted values from the model were scaled down to represent 15-day deposition to enable direct comparison with sediment trap data. The predicted model deposition gradient from the trap under the cage (0m) out to 25m from the cage edge mirrored the higher to lower deposition measured by sediment traps. However, DEPOMOD under-predicted deposition at all stations on all dates when compared to sediment trap data as shown by factor values less than 1 at all stations (Table 6.6). Highest accuracy was measured in August 2001 where the average factor was 0.81, giving an accuracy of ± 19.3%, with the lowest factor in April 2002 (0.48), giving an accuracy of ± 51.9%. Summarizing the data across all 3 sediment trap collection periods gave an average accuracy of ± 32.0% for model predictions.

Table 6.7: Comparison of 15-day observed verses predicted faecal particulate carbon deposition. Actual deposition measured using sediment traps at stations along a transect from cage 8 at Portavadie, collected every 3-days over a 15-day period each month. Predictions from contour plots generated using DEPOMOD dispersion model, with annual deposition scaled down to represent 15-days production. Units: g C m^{-2} 15-$days^{-1}$.

Collection	Prod'n	Under Cage			5m Station			15m Station			25m Station		
		Actual	Predicted	Factor	Actual	Predicted	Factor	Actual	Predicted	Factor	Actual	Predicted	Factor
August	3.84	234.27	102.85	0.44	75.75	71.8	0.95	41.04	40.75	0.99	29.79	25.81	0.87
February	3.06	85.20	83.93	0.99	120.82	58.63	0.49	55.61	33.28	0.60	22.54	21.08	0.94
April	2.82	159.64	79.69	0.50	109.50	55.63	0.51	61.73	31.57	0.51	49.46	20.00	0.40
Average		**159.70**	**88.82**	**0.64**	**102.02**	**62.02**	**0.65**	**52.79**	**35.20**	**0.70**	**33.93**	**22.30**	**0.74**

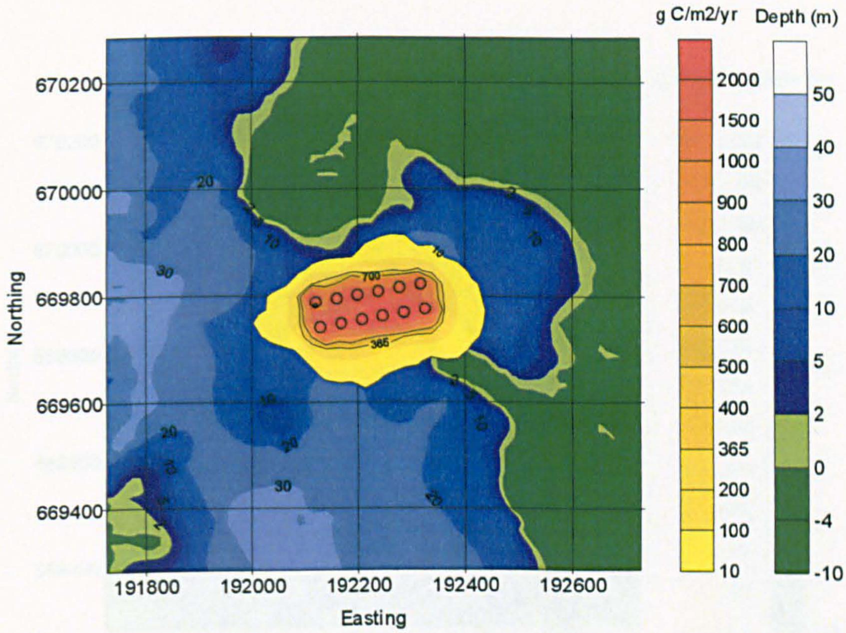


Figure 6.14: Contour image for Portavadie fish farm showing predicted annual faecal carbon settlement to the sediment, using DEPOMOD dispersion model, overlaying a 1km^2 bathymetric map, based on the food input for August 2001. See text for model parameter specifications.

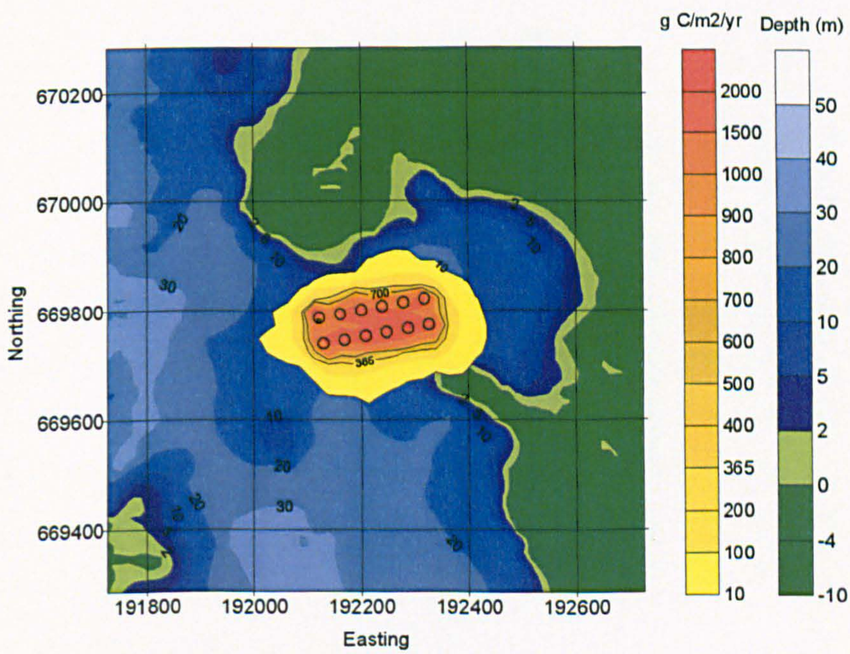


Figure 6.15: Contour image for Portavadie fish farm showing predicted annual faecal carbon settlement to the sediment, using DEPOMOD dispersion model, overlaying a 1km^2 bathymetric map, based on the food input for February 2002. See text for model parameter specifications.

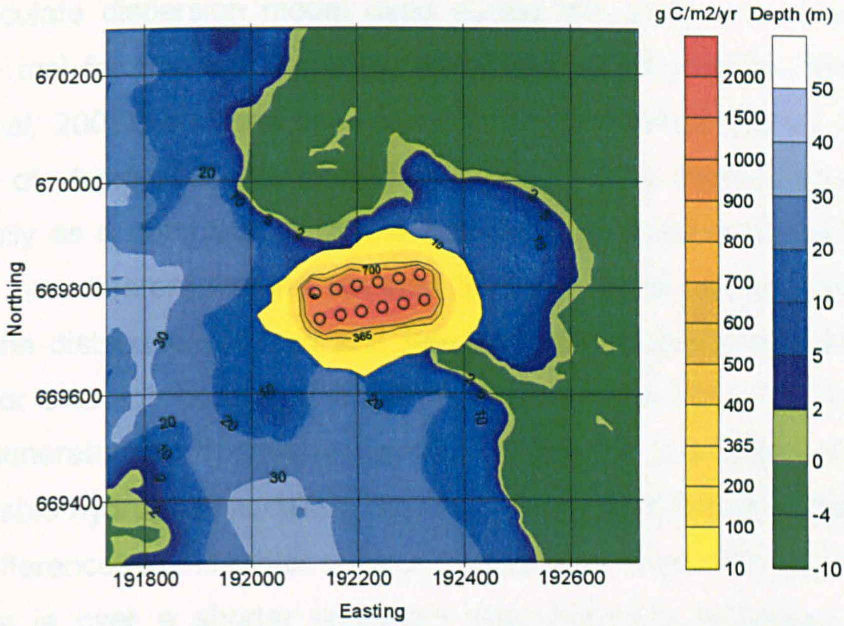


Figure 6.16: Contour image for Portavadie fish farm showing predicted annual faecal carbon settlement to the sediment, using DEPOMOD dispersion model, overlaying a 1km^2 bathymetric map, based on the food input for April 2002. See text for model parameter specifications.

6.4 Discussion

The particulate dispersion model used during this study was developed as a predictive tool for use by regulating authorities and managers (Brooker, 2002; Perez *et al*, 2002), and has been used in Environmental Impact Assessments (Institute of Aquaculture, unpublished data). The model was also used successfully as a comparative tool to assess differences in deposition between sites that use different feeding systems to deliver food. Specifically the model predicts the distribution of feed and faecal carbon waste on the seabed, either annually or over the course of a full production cycle (18 – 24 months). The outputs generated for this study covered 15-days of production commensurate with available hydrographic data (Chapter 2), the period being sufficiently long to identify differences in the raster contour images over time. Although modelling 15 days data is over a shorter timescale than originally envisaged (i.e. annual production), model outputs are valid by virtue of the robustness in model design that allows variable data and timescales to be simulated. Importantly for validation purposes, model outputs could be compared directly to sediment trap data (Chapter 5).

Irrespective of their complexity, computer based models are simplified representations of the processes, variables and relationships that function in the natural environment. Since their inception (Gowen *et al*, 1989), particulate waste dispersion models have undergone various transformations as the influences on where particulate waste is deposited on the seabed have become apparent and a means of modelling these influences has been determined (Silvert, 1992; 1994; McDonald *et al*, 1996; Hevia *et al*, 1996; Pereira, 1997; Chen *et al*, 1999a; 1999b; Cromey *et al*, 2002). Variable bathymetry, random settling velocity, random particle starting position and estimates of waste through mass balance used in the above models are all included in this GIS model (Brooker, 2002). Further, this study has shown that the movement of cages has a relatively small, but nonetheless, perceptible influence on the deposition of particulate farm waste, even where tidal range was small. This is particularly true for the area under the cage.

6.4.1 Cage movement

This study has shown that the physical influence of cage movement can be mathematically incorporated into a dispersion model but that its use is only appropriate when the model has a spatial scale that can register the movement. In this study movement from an arbitrary starting position was measured at up to 10m over the course of 8 hours, with movement driven by current speed and direction over the course of the tide. Thus models that use greater than 10m spatial resolution (Dudley *et al*, 2000; Cromey *et al*, 2002) would not benefit from the introduction of this level of movement, as the grid cell size in those models would be too large to register changes in deposition. Tidal height variation at Portavadie had not reached its maximum during the measurement phase, however, and larger movement might be expected at the extremes of the spring/neap tidal cycle and also at deeper water sites depending on the tension of the moorings.

Movement data was collected over a relatively short period and was extrapolated for integration in to the GIS model but was assumed to represent the total movement over a whole spring/neap cycle, in a similar way to hydrographic data over a 15-day period is assumed to represent the annual cycle (SEPA, 2001). A greater accuracy would be gained if position was assessed over a full 15-day cycle at defined intervals, such as every 10 or 20 minutes, in a similar way to hydrography measurements. This might be achieved with the use of either GPS or differential GPS, where readings could be recorded via data-logging. However, this would require better accuracy than may presently be available commercially as it will depend of the availability of satellites to provide the required accuracy.

The GIS model has a spatial resolution of 1m that allowed the extent of the measured movement to be integrated fully into the model and for the effect to be measurable through the data and images generated. The validity of applying cage movement to dispersion models has clearly been demonstrated. The total area of seabed, on to which material from the fish farm was deposited, remained unaffected by the inclusion of cage movement. Cage movement, however, results in a reduction in the peak deposition under fish farm cages when using the moving cage model.

The dynamic interaction of cages on the environment (Silvert and Sowles, 1996) is acknowledged within the modelling processes, which resulted in a re-distribution of carbon settlement, lower predicted peak values and a reduction in the predicted particulate settlement directly under cages. This shows that the inclusion of cage movement in waste dispersion models is an important parameter in determining the magnitude and extent of particulate settlement, especially at distances close to a fish cage.

6.4.2 Comparing deposition under two feeding regimes

The GIS dispersion model used in this study is a research model that had not previously been used to compare outputs from sites under different feeding regimes. Contour images reflected the feeding methods indirectly through differences in FCR that after standardization (to per tonne of growth) showed no significant differences in predicted faecal deposition under each of the feeding types. Use of the faecal portion of the output only in model development is not uncommon (Cromey *et al*, 2002) and was assessed in this study because the majority of the sediment trap collections, spanning 8 weeks of sampling, contained faecal material only as indicated by the carbon content. The lack of a difference using a modelling approach confirms previous assertions made about the use of adaptive feeding technology over hand feeding and the effect on the environment using a biological approach (Chapter 4) and sediment traps (Chapter 5).

Although the model was used primarily to assess differences in faecal waste the outputs include a feed element. In this model there is a difficulty in assuming that the feed element of the model raster-images is an accurate depiction of the likely settlement at the farm sites. Feed loss is a transient process within cage culture and infinitely depends upon physical, biological and feeding characteristics at a farm site. The model assumes that feed loss occurs uniformly across all hydrographic measurements for example, but in reality feed loss is limited to feeding periods only. Also, the quality of staff feeding the fish to satiation, the stress on the fish in any one day, the prevailing weather conditions, tidal speed through the spring-neap cycle, water quality, water temperature variation with season and level of parasite infestation will all influence feed loss over varying temporal scales (Kadri *et al*, 1996). Therefore the feed element of the model output was thought to be unreliable for comparative purposes and also could not be validated anyway, due to lack of feed deposition data from Chapter 5. More reliable estimates may shortly be available from IoA, who are conducting whole net exclusion experiments to determine the quantity of feed lost at salmon farms that may also give some indication of feed loss over time that could then be incorporated in to the model to eliminate the assumption that feed loss is uniform (Reynolds, pers comm.).

Accounting for the difference in the number of cages present at each of the farms under investigation, the spatial distribution at Rubha Stillaig would appear larger than at the Portavaide site but was thought to result from bathymetric differences rather than the particular feeding type used. Importantly, the predicted spatial extent of the deposition at both sites was similar to reported field studies (Hall *et al*, 1990; Weston, 1990; Henderson and Ross, 1995; Karakassis *et al*, 1999; Kempf *et al*, 2002) and other modelling approaches (Cromey *et al*, 2002), limited to 50 - 80m from the cage edge.

6.4.3 Validation of predicted dispersion with observed sedimentation and future development

Model validation is an important function within model development, assessing agreement between predictions from the model with data collected in the field (GESAMP, 1991), whilst at the same time clarifying the assumptions and functional relationships. The GIS model provided a strong correlation to actual deposition at the Portavadie site, where predictions agreed well with field data, giving accuracy as high as ± 27.6 % when over-prediction of deposition under the cage was excluded. Overall, predictions and observations were a similar order of magnitude and the degree of accuracy reflected the variability seen at all stations in sediment trap collection data over the 6 weeks of sampling. Model predictions followed a similar pattern to field data, with decreasing deposition at increasing distances from the cage edge and there was no apparent patchiness in the interpolated raster-image.

Validation for Rubha Stillaig was poor, which would have invalidated any differences found under the different feeding regimes, had any been found. This poor validation was in part due the failure of currents meters at the site (Chapter 3) and the need to use hydrographic data from Portavadie to represent currents at Rubha Stillaig. Rhuba Stillaig is known to have a higher exposure to wind and to turbulent mixing than the more sheltered Portavadie site, which would have affected deposition (Silvert and Sowles, 1996), but neither of these was assessed during this study. It is believed that hydrographic data for Rubha Stillaig would have made some difference to predicted deposition, but of itself cannot explain the large over-prediction seen. The poor validation at Rubha Stillaig also means that further sampling needs to take place to ensure model robustness and for the model to be applicable to fish farm sites in general.

The GIS model includes all parameters present in other models, except re-suspension, and with existing knowledge about dispersal and deposition, the over-prediction at Rubha Stillaig cannot be explained easily. It appears highly likely that the primary reason for the over-prediction relates to the FCR. The FCR's provided by the farm company were biological "estimates", which were adjusted by -10% to reflect the fact that a proportion of the fish ate food but subsequent became mortalities (= economic FCR) (Fowler, pers. Comm.), for use in the model. Even accounting for this it was thought that the estimated FCR's for Rubha Stillaig were too high. This view is supported by the similar biomass of fish, similar feed input and importantly by the similarity in the amount of solids and carbon being deposited at the two sites (Chapter 5).

The relationship between FCR and waste estimates is a difficult concept (Talbot and Hole, 1994) because feeding is also linked to other factors, such as fish health, water temperature and weather. FCR is particularly important, however, because it is a highly sensitive parameter within the GIS model (Brooker, 2002) that can make a large difference to the predicted level of feed and faecal waste and, therefore, to the overall settlement pattern. Certainly a more representative estimate of FCR would have been gained if growth was calculated by weighing a random sample of fish from each cage at the start and end of the sediment trap collection periods, with a sufficiently large sample to take full account of size variations within cages. In suggesting this there is still a danger that if large fish were selected at the start and smaller fish at the end, or *vice versa*, FCR may be greatly under- or over-estimated.

If it is assumed that no errors were present in field collected data, subsequent measurement of sediment trap contents and model input data then differences between predicted and observed sedimentation may have been due to processes that are not included in the model, such as losses from leaching and post-depositional movement through saltation (Chen, 1999) and re-suspension (Cromey *et al*, 2002[b]; Stewart and Grant, 2002). Perez *et al*, (2002) attempted to overcome some elements of these by using filters as part of the interpolation process, but the relevance of the filters and their applicability to near-field and far-field distribution of waste (i.e. under cage and not under cage) was not tested during this study and may be a source of error. There is also a reliance on hydrographic data (current speed and direction) that takes no account of shear stresses between water layers, such as prior to and post-slack water, eddies and wind generated movement that adds to turbulent mixing and affects the dispersion.

There are also elements that are not currently included in any commercially available or research models. Hydrographic data is measured within 100m of farm sites to represent current speed and direction through the farm. There is, however, an acknowledged reduction in current speed and alterations in direction as a result of the presence of nets (Inoue, 1972; Black, unpublished data) and fouling of nets over time. Fish may also play a part in distributing waste, by having a tendency to swim in circles that creates a vortex, giving rise to suction of water through the bottom of the net and movement away through the cage at shallow depths (Beveridge, 1996). Such influences may particularly affect the dispersion directly under cages, the area where the GIS dispersion model predictions are least accurate. Henderson *et al*, (2001) noted that all of these processes would need to be investigated to provide a comprehensive model, with data tested for sensitivity within the model. However, it would be worth noting that increasing the validation accuracy under certain conditions and at certain sites may limit the general applicability of the model to represent salmon farming as a whole, which must remain the ultimate goal of such a model (Silvert and Sowles, 1996).

In general validated models can provide cost effective alternatives to full monitoring and field collection. This is especially true for the Institute of Aquaculture's GIS-based model that provides easy data entry and a requirement for smaller data sets, which IDRISI or other GIS software packages are easily capable of interpolating. Predictive capability in the model provides multiple functions. It allows this model to be used as part of an Environmental Impact Assessment decision support process, in determining whether a site is acceptable for farming, under the banner of site selection (Perez *et al*, 2003). It is also useful during production, for monitoring and to assess the impact of proposed increases in production. Henderson *et al* (2001) recommended that models be used as a management tool as part of the decision making processes for setting quality standards and objectives but that as yet few models are used in this way. SEPA (2004) have recently put out to consultation the use of the DEPOMOD model to assist in the prediction of Allowable Zones of Effect (AZE's) and ITI scores at fish farms, for example, although the methodology has not yet been adopted. Although not presently included, the GIS model could also be used to predict recovery after cessation of farming, but would require a detailed understanding of recovery processes and timescales (e.g. Karakassis *et al*, 1999; Macleod *et al*, 2004). The model could be developed for particulate nitrogen and potentially for dissolved wastes although degradation coefficients and the 2 dimensional nature of dissolved waste movement would need to be included (Doglioli *et al*, 2004).

Although this GIS-based dispersion model provides the industry with valuable information that can be tested at the farm scale, further development and validation of the model would be required for integration into the wider functioning of a water body and an assessment of carrying capacity. Importantly, the GIS framework used as the basis for the model allows the integration of varying spatial scales within the same framework. This would be particularly important in the development of Coastal Zone Management Plans (CZMP) in which the dispersion model forms a layer (see Ross, 1998; Nath *et al* 2000) within the model framework that could provide a fully integrated decision support system for aquaculture development.

6.4.4 DEPOMOD

It was difficult to provide a direct comparison between the DEPOMOD model and the GIS-based dispersion model. The input data to the models was the same but differences arose in how the data was interpreted within the respective models, especially in estimating the predicted level of faecal waste. DEPOMOD uses a digestibility approach, assuming 10% of the feed is water and 85% of the dry weight is retained by the fish (see Cromey *et al*, 2002), whereas the GIS model uses a carbon mass balance approach (Figure 6.1), which resulted in slightly different quantities of waste being predicted under the two modelling types.

The inclusion of a feed loss element in the GIS model was necessary for calculating the quantity of faecal material produced, via the mass balance calculations. Validation occurred against the faecal portion of the modelled output that would have been over-estimated had zero feed loss been assumed in the mass balance. Although DEPOMOD calculates faeces in a different manner, through water content and digestibility, the fact that 100% of the feed was assumed to be eaten results in an over-estimation of predicted faecal carbon. This was not taken in to account during the Cromey *et al* (2002) validation. Within the DEPOMOD model 100% feed consumption was required, however, because a single model output is produced, being either total solids or total carbon. The IoA GIS model therefore has a distinct advantage, whereby feed and faeces are treated independently with separate raster-images generated. Feed loss can therefore rightly be included in the model, even though analysis was assessed against the faecal portion of the output only.

There were also differences in the predictive timescales (1 year vs 15-days) and in the grid generation size. The wide spacing between polar circle cages combined with the 1m grid size in the GIS model suggests that deposition between cages was relatively low. Such distinction was not possible using DEPOMOD with a 25m grid size where the area under and between cages was treated as a block with a uniformly high predicted deposition within the limits of the cage layout.

The estimated overall accuracy of the DEPOMOD model predictions, in this study, across all dates was ± 32%, which shows a large decrease in accuracy over the data presented in the literature, where Cromey *et al* (2002) achieved ± 13.2% at a similarly depositional site. It is of note that the plus/minus estimate of accuracy in the model (using Equation 3) was an artefact of the equation used to calculate the accuracy, which generates an absolute value but in fact DEPOMOD consistently under-estimated deposition both under the cages (near-field) and at distances away from the cage edge (far-field).

During the production period analyzed, changes in FCR, food input and growth result in variable faecal waste predictions over time and this was reflected in the reduced accuracy of the DEPOMOD model outputs present here. Cromey *et al* (2002) were unable to take account of such variability because the feed input they used was based on a limited sediment trap collection period. The sediment trap data collected and used for validation purposes was collected over 2-days that took no account of the likely variable faecal outputs over time as fish increased in size. In the Cromey *et al* (2002) study sediment trap deployment was also conducted over a small percentage of the total area likely to be affected by deposition of waste particulate material (i.e. under the cage only).

Chapter 7

General discussion

Over the last 25 years the cage culture of Atlantic salmon in Scotland has grown significantly. In tandem with this development there has been an on-going debate on the environmental impact of fish farms. Some advocate a *zero-tolerance policy* towards any form of impact (e.g. Folke *et al*, 1994) while pragmatists acknowledge that such impacts are an unfortunate but necessary consequence of cage culture. Technological developments have not always kept pace with the growth in production but an improved understanding of feeding behaviour, better husbandry and formulated feed have all acted to mitigate the environmental consequences of the growth in the industry. Distributing feed by hand is still practised widely but technological developments in this area now allow feed to be distributed remotely. However, to date there had been insufficient study into whether the use of adaptive feeding systems conferred any environmental benefit, an assessment of which was the main purpose of this study.

This study focused on the environmental implications of using an Akvasmart CAS Adaptive Feeding System; a set of feeding equipment linked via a radio transmitter to a computer that monitors and regulates food delivery, indirectly assesses food intake and, in real-time, adjusts feeding rate, ration and feeding to satiation. Such systems (Blyth *et al*, 1993; Juell *et al*, 1993; Ang and Petrell, 1998) are a natural evolution of feeding strategy and were borne out of an understanding of feeding behaviour (Noakes and Grant, 1992; Kadri *et al*, 1996) and how best to accommodate the meeting of a fish's appetite in an efficient manner (DeSilva and Anderson, 1995).

Investigations were carried by comparing Portavadie fish farm, equipped with a CAS adaptive feeding system (Akvasmart UK Limited, Inverness, Scotland) and a farm site at Rubha Stillaig, where hand feeding takes place. The most obvious manifestation of the siting of open cage culture is the deposition of nutrient-rich waste particulate material on to the seabed (e.g. Hargrave, 1994) and its effect on the composition and diversity of the benthos (e.g. Henderson and Ross, 1995). The work therefore focused on 3 key areas; 1) a comparative assessment of the quantity and nutrient composition of particulate waste material that emanates from the cages under the two feeding systems, affected by feeding, feed waste, faecal production and dispersion (physical approach); 2) a comparative assessment, between the two sites, of the benthic fauna that populate sediments under and around fish cages (biological approach); and 3) the use of a GIS-based waste dispersion model (Brooker, 2002; Perez *et al*, 2002) to compare the predicted dispersion and settlement of carbon under the two feeding regimes; with comparison against the current Scottish industry standard model, DEPOMOD (Cromey *et al*, 2002) (modelling approach).

7.1 Faecal and feed deposition

Sediment trap deployment and collection represented a physical approach for comparing waste deposition under the two feeding regimes being investigated. Sediment traps are used widely in oceanography but their use in more dynamic coastal waters, in which fish farms are located, has only rarely been reported. In this study they were successfully used to compare the composition and extent of faecal particulate settlement at each of the sites under investigation. Chapter 4 showed that rates of faecal deposition in particular were broadly similar under each of the feeding regimes, in terms of faecal solids deposited, their composition and overall rates of sedimentation.

The measured particulate waste (faceal solids under cage 190 ± 18.3 g FS t^{-1} at Portavadie and 268.8 ± 34.9 g FS t^{-1} decreasing with distance from the cage - Chapter 4) appeared similar to other sediment trap studies carried out at fish farms (Hargrave, 1994; Kupka-Hansen *et al,* 1991; Kempf *et al*, 2002). Direct comparison remained difficult due to the under reporting of specific fish biomass and food use within cages during those studies. There was, however, a failure to achieve the aim of measuring levels of waste derived from uneaten food, with food collected on only 3 occasions during the 6 weeks of sediment trap studies thought to under-represent that likely deposition during this period.

It was expected that levels of feed wastage could be determined during this study. Current published estimates of 5% to 15% food waste derive from data that were collected some time ago, remains highly variable and needs to be fully reevaluated. Cromey *et al* (2002) suggests 3% feed waste is a more realistic current estimate, based upon well argued but anecdotal evidence. It is well argued because current husbandry practice and the high cost of feed requires fish farm companies to closely monitor feed wastage, through the use of sub-surface cameras or uplift systems, but there is no firm data available to confirm that wastage has reduced to this level. Unfortunately, the configuration of the sediment traps used in this study proved to be insufficient for estimating feed waste. Cromey *et al* (2002) and Kempf *et al* (2002), for example, had successfully collected feed pellets during their respective studies but in retrospect the short duration of these studies was insufficient to re-evaluate estimated feed losses. A number of researchers had suggested that measurement of particulate waste was difficult in the field (Ackerfors and Enell, 1994; Cho and Bureau, 1997). In this study the sporadic nature of food input to cages, the high settling velocity of feed pellets and the limitation of the technology to reflect the variations of feed waste in both time (feeding periods) and space (wide distribution of large pellets with high settling velocity) meant that food waste estimation was and continues to be a difficult parameter to assess adequately using sediment traps.

Although sediment traps proved insufficient, the cone sensor on the adaptive feeding system was able to count pellets falling through it and could have been used as a basis for estimating feed losses under this feeding method. However, the cone is positioned at mid-depth within a cage (Figure 2.3) to assess pellet counts and to aid the decision to stop feeding, and not specifically to assess deposition on the seabed. In future work an additional similar cone, positioned at the base of the net (effectively measuring pellets released from the cage as waste) might prove useful in assessing waste pellet deposition.

Overall, this study was not able to enhance our understanding of feed losses under either the hand feeding or adaptive feeding regimes and current estimates must continue to be used. However, that feed loss under both of the feeding systems was lower than reported in some literature (e.g. Findlay and Watling, 1994) can be deduced from circumstantial evidence. In particular the lack of identifiable pellets in benthic grabs samples and on the video survey (Chapter 4) as well as the low occurrence of feed pellets in the sediment traps (Chapter 5) suggested that feed losses were minimal, adding to the anecdotal evidence presented by Cromey *et al* (2002). As a result, the lowest end of the available feed waste estimates (i.e. 3%) was used for modelling purposes (Chapter 6).

Importantly, studies to assess levels of particulate waste deposition must be of a sufficient duration to assess the full variation of waste settlement. The final aim of this sediment trap study was to repeat measurements to evaluate variations in settlement over time which showed that particulate waste output from cages was both highly variable and sporadic. 1 or 2-days data, as has been used elsewhere (Morrisey *et al*, 2000; Cromey *et al*, 2002; Kempf *et al*, 2002) is therefore wholly insufficient to realistically assess faecal and feed waste output. Faecal waste by its nature is produced more often and in larger volumes than feed waste and could have been studied for a shorter duration than was used in this study, which was supported by the similarity in faecal waste per tonne of growth over time. Measurement of uneaten feed waste would benefit from more intensive and longer term study (see section 7.6, below).

7.2 Implications for the benthos

An assessment of mass balance calculations used in Chapter 6 showed that faecal waste forms an increasingly high proportion of the environmental particulate loading around fish farms. The quantity of faecal waste is in general related to the digestibility of the feed (Cho and Bureau, 1998) but also reflects the quantity and size of fish present in the cage. Thus faecal waste deposition will not necessarily alter between farms that have a similar fish biomass, even where different feeding methods are used (Chapter 4), which in turn made a comparative assessment of benthic species composition under the different feeding regimes difficult.

The macrofaunal community at both farm sites showed a response similar to the findings reported by Henderson and Ross (1995) for other Scottish fish farm sites. They reported increases in abundance but reductions in diversity close to the cages, attributable to increases in the deposition of nutrient rich waste material.

Both Portavadie and Rubha Stillaig sites increased their proportion of nutrient tolerant species, such as *Capitella capitata* and *Ophryotrocha puerilis* through the course of this study, both of which continued to prove useful indicators of enrichment. *Capitella* sp. is traditionally courted as "the" indicator species for nutrient enrichment studies in coastal waters due to its ubiquitous nature (Pearson and Rosenberg, 1978; Chareonpanich *et al*, 1993; Tsutsumi, 1993; Felsing, 2003).

During this study other species have proved to be equally insightful. *Heteromastus filiformis*, and *Corophium* sp. (Bat and Raffaelli, 1998), which are known to tolerate slight increases in nutrients, were particularly useful. The presence of these animals in the sediments at Rubha Stillaig indicated that a nutrient increase had occurred even though sediment carbon levels and traditional univariate indices had failed to identify the change. Importantly, this level of understanding could not have been gained had the benthic samples not been identified to at least species level, where possible.

It was recognised that identification and analysis of benthic animals to species level is a skill that requires much time and effort. Fortunately during this study there was sufficient time available for such analysis. Whilst other researchers have found that identification to family level is sufficient to detect impacts (Warwick, 1993; Karakassis and Hatziyanni, 2000), this study has shown that a more detailed knowledge is required if degrees of impact are to be fully investigated. Species level identification might be particularly useful in post-fish farming monitoring strategies, as it would seem to allow more subtle changes (improvements) to be identified. In this study species level identification also showed subtle distinctions between the stations being measured, using multivariate techniques, that may have been lost had identification been done at a higher level.

Overall, species changes throughout this study were more evident at Rubha Stillaig, the hand fed site. At this site the benthic composition at all stations closely resembled the reference site at the start of trial but by the end both species composition and abundance were largely altered as a result of nutrient deposition, which was indicated by increased sediment carbon levels (5.3.1). At Portavadie, composition altered to a lesser extent and the observed changes in abundance could equally have resulted from natural variability than to any deterioration in sediment quality. Sediment data should have been collected at Portavadie in August 2001 that would have identified any increase in nutrient loading. The increases observed in the abundance of *Capitella* sp. would improve the sediment in the longer term, due to their ability to turnover high quantities of organic material relatively quickly. *Capitella capitata* is known to increase rates of mineralization by 87% (Heilskov and Holmer, 2001), that would increase the turnover of carbon and burial processes, improving the nutrient content of sediments.

In part, a lack of distinction between sites in the benthic study (Chapter 5), which was unable to identify an equilibrium community structure, reflected the short timescale available for a comprehensive analysis. Clearly the changes that occurred at each site were as a consequence of nutrient enrichment but differences between sites could not be linked directly to the different feeding systems employed at each site. Benthic studies of this type would ideally require sufficient temporal scale to ensure that any measured improvement in benthos could be ascribed to the use of the feeding system rather than natural variation brought about by other factors. Food availability, for example, is particularly important in the context of fish farms where faecal material provides a relatively constant source of food and nutrients to the seabed under both feeding regimes. The uniform settlement of faeces alone might explain why benthic composition did not vary between sites during this study. Benthic studies by their nature tend to be longer-term activities, because it is important to gain an understanding of the underlying variability in benthic populations (Ervik *et al*, 1998) before ascribing alterations in structure to specific changes in nutrient loading. It is worth noting, for future similar work (e.g. other fish species), that even longer-term investigations may fail to show alterations to benthic populations that could specifically be attributable to the use of adaptive feeding systems, because of the uniform nutrient loading from faeces described earlier.

The assessment of benthic communities was carried out at two farms that had undergone previous production cycles for differing periods of time, which resulted in different starting sediment conditions. Despite the short timescale and the different sediment conditions the data gathered suggests that benthic populations do not seem to inherently benefit from the use of adaptive feeding systems *per se* over a well managed hand feeding strategy. However, the shorter on-growing period resulting from the use of the adaptive feeding system could provide a tangible benefit to the benthos if the time gained was used constructively, for fallowing.

Fallowing provides an opportunity for the benthos under cages to turnover the sediment, without additional waste being continually added, and is thus of environmental benefit. Sediment is able to recover to some degree before production is re-commenced. In a study of 80 farm sites in Norway, Carroll *et al* (2003) suggested that fallowing was one of the key management factors affecting the sustainability of fish farming. A typical fallowing period in Scotland is 2 months in every 24-month production cycle. When using adaptive feeding systems the fallow period could and perhaps should be increased to take account of the fact that the on-growing period is 3 months shorter than under hand feeding. Failure to increase the minimum fallow period at adaptive feeding sites may encourage farmers to intensify production through a quicker turnover of on-growing periods and this may be detrimental to the sediments over the long-term.

However, it would be contentious to recommend that 5-6 months of fallowing be enforced, as it may be perceived that a penalty was attached to the use of the equipment, when under hand feeding farmers could continue with the minimum fallowing period. Also, whether this would affect a particular farm company depends on their fallowing strategy, where many sites are already fallowed for longer periods than the minimum due to management decisions or locally agreed arrangements. Overall, increasing the fallow period to a compulsory minimum number of weeks would encourage farmers to use feed and feeding methods that reduce the growth period and the Feed Conversion Ratio (FCR), to the benefit of the benthic population and sediment quality.

7.3 Considerations for modelling

Utilizing modelling to compare differences in predicted deposition between farms using different feeding methods was a novel and untried approach. The GIS-based model used was also a relatively new method of assessing particulate waste dispersion (Perez *et al*, 2002). Despite this, the high grid resolution available through GIS allowed assessment of predicted deposition both under the cage and at specific points within the grid (equal to the stations measured in Chapters 4 and 5) with a high degree of accuracy.

Subtle differences between the predicted depositions under each of the feeding types are identified in Chapter 6. More generally, standardization of the contour images to deposition per tonne of growth eliminated the variation caused by the large differences in FCR between the sites, so that overall there were no significant differences in predicted deposition per tonne of growth under each feeding type. In absolute terms deposition at Rubha Stillaig was much higher than Portavadie. However, the poor validation against the sediment trap data for Rubha Stillaig (6.3.4) meant the predicted deposition for that site may have been unreliable. In particular the high estimated FCR data used in the model was thought to be too high, based on both the lower FCR achieved at Portavadie and the similarity in sediment trap data between the two sites (Chapter 4).

FCR is the parameter that most notably affects the outputs from the GIS dispersion model, with the model being particularly sensitive to small changes in its magnitude (Brooker, 2002). There may be a tendency to misunderstand FCR and its implications for the amount of waste generated (Talbot and Hole, 1994). An FCR of 1.2 specifically means that 1.2kg (dry weight) of food was required to produce 1kg (wet weight) of fish and, therefore, it would be reasonable to assume that 0.2kg was released to the environment as particulate waste; and by inference lowering the FCR to 1.1 equating to 0.1kg lost to the environment. This assumption, however, is incorrect (Talbot and Hole, 1994), as the relationship between FCR and particulate waste loading is not a proportionate one.

Lopez-Avarado (1997) noted that a reduction in FCR of 50% in the 20 years leading up to 1997 resulted in an 80% reduction in waste discharges. Much of this reduction was achieved through the conversion from trash fish feed to a pelleted formulated feed and latterly through better feed formulation (Cho and Bureau, 1997; Sveier *et al*, 1999). A better understanding of feeding behaviour (Olla *et al*, 1992; Noakes and Grant, 1992; Kadri *et al*, 1996) and improved husbandry practice has also contributed to a reduced FCR. Thus, FCR is not linked directly with waste but fundamentally reflects the conversion of food to biomass which in turn is reliant on digestibility, fish health and maintenance and feeding strategy; that in turn are affected by abiotic factors such as sea temperature and weather. As the farming industry moves towards achieving a ratio of 1:1, the large environmental benefits that have resulted from improved feed formulation and production techniques, reported by Lopez-Avarado (1997), will be more difficult to come by.

Until this study, cage movement had been an acknowledged source of error in deposition models (Cromey *et al*, 2002) but the extent of movement and its effect on deposition had not been estimated. The coding used for the GIS model, that continues to be developed at the Institute of Aquaculture, was successfully enhanced to include cage movement in an attempt to eliminate this as a source of error. Although horizontal movement was less than 10m in any one direction, there was a perceptible effect on deposition. Most notably it resulted in lower settlement of particulate waste directly beneath cages than in a previous version of the model (23% less for feed and 11% less for faeces). It is important to recognise that reducing the predicted settlement of particulate waste under the cages increases the settlement further away and that the feed and faeces were not lost from the model output. The effect of cage movement was measurable within the GIS model due to the high grid resolution, a factor that is not available in many other dispersion models.

The GIS model used in this study continues to be developed but the data gathered using sediment traps has proved useful in validating the GIS model in its current form. Model development is a protracted task that requires the use of repeated datasets under different conditions to ensure robustness and its eventual applicability to fish farming as a whole under different environmental conditions. The calculated accuracy of the GIS model using Portavadie data was ± 58.1%. The accuracy reflected the wide variability in the sediment trap data and the fact that certain parameters are not currently included in the model, including feed loss variation with time, leaching rates, post-deposition movement and re-suspension (see Chapter 6). Future development of these elements for inclusion into the GIS-model and with the potential to integrate it as a layer within a carrying capacity or coastal zone management model will depend upon the availability of further datasets from different sites to assess its general applicability to fish farming as a whole.

It was inevitable that a new waste dispersion model would have to be compared against DEPOMOD (Cromey *et al*, 2002), the industry standard. Variations in the interpretation of data input made such a comparison difficult, although specific advantages of the GIS model are specified in Chapter 6. Using data from this study, it was suggested that the accuracy of DEPOMOD was much lower than previously thought. More specifically, in a comparison with field data, DEPOMOD was shown to under-estimate waste deposition at all stations measured.

If DEPOMOD was used to agree a new site or biomass increase then the likely predicted deposition of waste, on which any decision might be based, could also be under-estimated. This may have implications for the confidence that can be ascribed to DEPOMOD although it must be stressed that the data presented represents a limited assessment of the model and that further validation studies should be carried out.

7.4 Allowable Zone of Effect (AZE)

Allowable Zone of Effect or AZE is a concept embedded in the regulatory structure for Scottish fish farms (see Fernandez *et al*, 2000 for a review). Although not discussed previously it was important to raise it here as the outcomes of this study suggest there may be a need to review the extent of the AZE. The AZE is effectively a deposition zone around a fish farm within which the environmental impact of a farm is assessed against set quality standards and objectives, but where failure of some elements is permitted under the regulatory procedures (SEPA, 2001). At present the AZE is set at 25m in all directions from the cage edge for all fish farms in Scotland and requires that a minimum of 2 species are present under the fish cages, that at 25m distance the diversity and species richness are to be no less than 80% of the background level, that no afaunal zone is present and that carbon in sediment should not exceed 7% (SEPA, 2001).

In Chapter 4 biological data collected from cages under both feeding regimes indicated that the sediments were grossly impacted out to 25m from the cage edge, according to the criteria suggested by Henderson and Ross (1995). Using the same criteria, however, moderate to heavy impacts were also recorded at 50m, at both sites. All stations where biological samples were collected showed the benthic populations to be above the minimum standards set by the regulating authority except species abundance at 25m, which was below the 80% of background level as required under the regulations.

In addition, both the sediment trap data and predictions from the GIS-model contour images suggest that both sites were depositing particulate waste material on to the seabed beyond the current 25m AZE. It is important to note, in the context of a single AZE applying to all sites in Scotland, that both Portavadie and Rubha Stillaig were characterized as depositional sites with a lack of residual flow in any one direction. Sites that have a higher current speed are likely to be affected at greater distances than reported here.

Postscript to section 7.4 – Subsequent to production of this thesis, SEPA (2004) have submitted a consultation paper that details a proposal to generate site-specific AZE's, using DEPOMOD, integrated with predictive benthic indices (such as ITI score). This AZE would be used to generate standards as part of the regulation of the site. The proposal had yet to be accepted and did not form part of the regulatory procedures for fish farms at the time of writing.

Data collected during this study therefore suggests that the current AZE approach to impact monitoring should be reviewed, perhaps encompassing site specific criteria. Based on the present study the method of feed delivery (i.e. adaptive feeding system or hand feeding) would not appear to be a critical factor in setting the limits of the AZE. This statement must be cautionary, however, because levels of feed waste under each of the feeding types investigated could not be established during this study as outlined above. A review of the AZE criteria will become increasingly necessary as new production systems emerge for the culture of novel fish species, such as halibut, cod and other white fish that may have impacts that are different from Atlantic salmon.

7.5 Adaptive feeding systems and sustainability

Adaptive feeding systems have been shown to have a direct affect on production by reducing the FCR (Kadri, pers comm.) to as low as 0.95 (Telfer and Beveridge, 2001). Data collected during this study has shown that lower FCRs and a shorter production period were achieved at Portavadie fish farm than was achieved at the hand fed Rubha Stillaig site. Part of this improvement undoubtedly resulted from the adaptive systems' ability to feed in short bursts over extended periods and to better reflect Atlantic salmon feeding behaviour (Blyth *et al*, 1993; Kadri *et al*, 1996; Talbot *et al*, 1999) that improves growth; rather than short heavy bursts of feed under hand feeding. The use of an adaptive feeding system also removes the subjective decision over satiation from the farmer to a pellet detection system and software that objectively evaluates subsequent feeding decisions. That such systems are not used more comprehensively throughout Scotland may be related to farm size and the cost/benefit of these systems, which fell outside of the scope of this study.

From a wider environmental perspective, the lower FCRs generated using adaptive feeding systems means using less food per kilogram of production. This must be beneficial in the long term with a lower use of fish oil per kilo of production, for example. Such sustainable environmental benefit is relative, however, as the level of production continues to increase world-wide and the amount of formulated food used increases year on year in real terms, irrespective of the feeding mechanism used to distribute that food. Environmental benefit and improved long-term sustainability on a global scale will primarily accrue through continued changes in feed formulation, where fish oil is reduced or replaced effectively with alternative protein sources, such as plant material, with a smaller and more limited contribution from the choice of feeding system.

7.6 Future Work

Analysis of the effects of using adaptive feeding systems to feed fish at marine cage sites is at an early stage and would benefit from additional assessment. Two areas of study would be particularly useful, first an up-to-date assessment of food waste levels, and secondly an analysis of the environmental benefit of fallowing.

An up-to-date assessment of the levels of feed waste from cage culture is urgently required for modelling purposes, taking into account new developments in feed composition and modern husbandry techniques. The sediment trap method used here proved an inadequate method. Realistically, whole-net exclusion experiments would be the only reliable method available to assess waste, where all outputs over a set period are collected and analyzed. Exclusion experiments are not feasible on large cages however, due to problems with water exchange and oxygen depletion in cages, and the physical difficulties associated with drag and the manoeuvring of large and heavy tarpaulins; all reasons it was not attempted during this study with the two available sites. It would also be inappropriate to conduct very small scale trials in tanks, were secondary feeding will interfere with waste estimates.

Food waste estimates must rely on small scale experiments at fish cages in the open sea (e.g. maximum 5m x 5m x 5m deep), where the limitations identified above are reduced, and by extrapolating the data gained to full production quantities. Sediment traps may then be appropriate at full production sites as a means of validating the extrapolated data. Such studies may benefit from the design of a more appropriate sediment trap and a more intensive use of traps under and around single cages than was used in this study, with deployment along multiple-transects away from the cage. Clearly, if feed estimates were targeted in particular, there would be little point in extending the transects beyond 20m from the cage edge at the majority of in-shore sites, as the high settling velocity of feed means feed settles quickly and over short distances. Measurement out to this distance would also factor in any potential movement in the cages as a result of wind and tidal effects (Chapter 6).

It is suggested that an increased fallowing period could be applied where adaptive feeding systems are in use (Chapter 5), which may have an environmental benefit by allowing sediments to recover prior to the next production cycle. It might also be argued that continued production may be a better strategy, so that a significant population of *Capitella capitata* is maintained and sediment re-mineralization processes are maximized. Most research in this field has been conducted at farms that have ceased production altogether, to assess benthic recovery (Karakassis *et al*, 2000; Kraufvelin *et al*, 2001). There is, however, no understanding of exactly what benefit fallowing has to sediment under cages. An assessment of the changes in species and physio-chemical parameters during varying lengths of fallow period would be useful. Such data may then be used to assess whether an increase in the fallow period, which is afforded by using adaptive feeding systems by virtue of the shorter growth period, is an environmentally beneficial strategy. It would be equally important to assess a farm that has a continuous production strategy to evaluate the effect of having no fallowing period.

7.7 Conclusions

Overall, the physical, biological and modelling approach used during this thesis has shown that the use of adaptive feeding systems at fish farms cannot confer a tangible environmental improvement, although future work on feed losses using the system, identified above, may alter this statement. However, the use of adaptive feeding systems could form part of a sustainable farming strategy for fish farms. Specifically, the shorter growth period using the system in combination with the potential to increase the fallowing period, whilst maintaining current levels of production, should benefit the localised benthos by reducing the overall deposition of waste over a whole production cycle and by increasing the time available for recovery in between production cycles.

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Appendix 1 – Feed data collection sheet

Lighthouse Ltd - Portavadie Site **Cage No.**

Date

Date	Feed Type	Feed Size (mm)	Quantity Fed (kg) AM	Quantity Fed (kg) PM	Sea Temp (°C)	Weather	Any Comments
			n/a				Traps in today
							1st collection
							2nd Collection
							3rd Collection
							4th Collection
							Final Collection

Appendix 2 – Statistical Table for Chapter 4 – Sediment Trap Study

Table A2.1: 2-sample t-test for comparison between 4th root *transformed* data (except 25m = Box-Cox transformation λ = 0.337) for faecal solids deposited in sediment traps at specified stations for combined data from Portavadie and Rhuba Stillaig fish farm sites for collections made February 2002 and April 2002 at both sites. Faecal solids deposition adjusted for reference levels. T = test statistic, df = degrees of freedom, P = probability. Significance at p = < 0.05.

Station	T	df	p
Under	-0.59	24	0.563
5m	2.07	21	0.051
15m	0.30	16	0.766
25m	-1.24	20	0.228

Table A2.2: Spearmans correlation coefficients comparing faecal solids (FS), total carbon (TC) and total nitrogen (TN) for sediment trap samples collected at Portavadie and Rubha Stillaig fish farms. Respective data adjusted for reference levels. Probability in brackets. Significant values highlighted in red.

Portavadie				
Date	**Station**	**FS/TC**	**FS/TN**	**TC/TN**
August 2001	P_0	0.204	0.230	0.440
		(0.389)	(0.330)	(0.052)
	P_5	-0.123	-0.041	0.974
		(0.617)	(0.869)	(<0.001)
	P_{15}	-0.206	-0.324	0.879
		(0.461)	(0.238)	(<0.001)
	P_{25}	0.305	0.170	0.922
		(0.204)	(0.488)	(0.001)
February 2002	P_0	0.961	0.978	0.970
		(0.001)	(0.001)	(0.001)
	P_5	0.582	0.404	0.905
		(0.018)	(0.120)	(0.001)
	P_{15}	0.307	0.309	0.686
		(0.248)	(0.244)	(0.008)
	P_{25}	0.065	-0.577	0.529
		(0.810)	(0.019)	(0.035)
April 2001	P_0	0.912	0.846	0.959
		(0.001)	(0.001)	(0.001)
	P_5	0.625	0.493	0.964
		(0.013)	(0.062)	(0.001)
	P_{15}	0.686	0.379	0.848
		(0.003)	(0.147)	(0.001)
	P_{25}	0.595	0.088	0.703
		(0.019)	(0.809)	(0.002)

Rubha Stillaig				
Date	**Station**	**FS/TC**	**FS/TN**	**TC/TN**
February 2002	P_0	0.665	0.712	0.936
		(0.005)	(0.002)	(0.001)
	P_5	-0.026	-0.179	0.965
		(0.937)	(0.578)	(0.001)
	P_{15}	-0.327	-0.707	0.636
		(0.342)	(0.015)	(0.036)
	P_{25}	-0.365	-0.870	0.279
		(0.270)	(0.001)	(0.407)
April 2002	P_0	0.824	0.794	0.988
		(0.001)	(0.001)	(0.001)
	P_5	0.512	-0.054	0.455
		(0.043)	(0.844)	(0.077)
	P_{15}	-0.263	-0.792	0.330
		(0.435)	(0.004)	(0.322)
	P_{25}	-0.356	-0.693	0.421
		(.212)	(0.006)	(0.134)
September 2002	P_0	0.794	0.697	0.899
		(0.001)	(0.001)	(0.001)
	P_5	0.188	0.090	0.978
		(0.520)	(0.760)	(0.001)
	P_{15}	0.403	0.170	0.868
		(0.122)	(0.529)	(0.001)
	P_{25}	0.170	-0.625	0.896
		(0.529)	(0.010)	(0.001)

Table A2.3: 2-sample t-tests and Mann-Whitney U-test results comparing total carbon and total nitrogen for sediment trap samples collected at Portavadie and Rubha Stillaig fish farms in February and April 2002.

Carbon

Station	T	W	df	p
Under	-0.76	-	59	0.451
5m	5.39	-	56	<0.001
15m	-	1142.5	-	<0.001
25m	-	1139.0	-	<0.001

Nitrogen

Station	T	W	df	p
Under	-	856.5	-	0.215
5m	-	1137.5	-	0.002
15m	-	1093.5	-	<0.001
25m	-	1034.5	-	0.013

Table A2.4: 2-sample t-test results comparing carbon/nitrogen ratios for sediment trap samples collected at Portavadie and Rubha Stillaig fish farms in February and April 2002.

Station	T	df	p
Under	1.13	57	0.263
5m	3.95	46	<0.001
15m	4.06	51	<0.001
25m	2.85	53	0.006

Table A2.5: Factorial analysis of variance output based on a General Linear Model (GLM) analysis of regression of rates of change in faecal sedimentation rate with distance from cage centre for 3 collections of particulate deposition using sediment traps at Portavadie fish farm. Collection dates August 2001, February 2002 and April 2002.

(a) Analysis of variance table with date of collection, with distance as a covariate, after ln(x) transformation of faecal sedimentation rate. Seq = sequential, Adj = adjusted for entry order into the model, p = probability with significance at <0.05.

Source	df	SeqSS	AdjSS	AdjMS	F	p
Date	2	1.5387	0.4666	0.2333	0.58	0.564
Distance	1	21.173	19.797	19.797	49.29	<0.001
Interaction	2	1.851	1.851	0.9255	2.30	0.112
Error	45	18.0757	18.0757	0.4017		
Total	50	42.6384				

(b) Comparisons between regression slopes and intercepts and their respective group averages after ln(x) transformation of faecal sedimentation rate. p = probability with significance at <0.05.

Date	Intercept	T	p
Average	3.82 ± 0.16		
August 2001	4.03 ± 0.21	0.99	0.325
February 2002	3.63 ± 0.23	-0.88	0.427
April 2002	3.79 ± 0.23	0.11	0.916
	Slope		
Average	-0.047 ± 0.007		
August 2001	-0.065 ± 0.010	-2.06	0.045
February 2002	-0.050 ± 0.010	0.56	0.632
April 2002	-0.032 ± 0.010	1.50	0.141

Table A2.6: Factorial analysis of variance output based on a General Linear Model (GLM) analysis of regression of rates of change in faecal sedimentation rate with distance from cage centre for 3 collections of particulate deposition using sediment traps at Rhuba Stillaig fish farm. Collection dates February 2002, April 2002 and September 2002.

(a) Analysis of variance table with date of collection, with distance as a covariate, after ln(x) transformation of faecal sedimentation rate. Seq = sequential, Adj = adjusted for entry order into the model, p = probability with significance at <0.05.

Source	df	SeqSS	AdjSS	AdjMS	F	p
Date	2	4.7338	5.1044	2.5522	4.69	0.015
Distance	1	31.2401	32.1618	32.1618	59.05	<0.001
Interaction	2	1.7181	1.7181	0.8590	1.58	0.219
Error	39	21.2407	21.2407	0.5446		
Total	44	58.9327				

(b) Comparisons between regression slopes and intercepts and their respective group averages after ln(x) transformation of faecal sedimentation rate. p = probability with significance at <0.05.

Date	Intercept	T	p
Average	3.41 ± 0.19		
February 2002	3.92 ± 0.27	1.90	0.064
April 2002	3.66 ± 0.27	0.96	0.343
September 2002	2.64 ± 0.27	-2.86	0.031
	Slope		
Average	-0.062 ± 0.008		
February 2002	-0.070 ± 0.012	-0.68	0.502
April 2002	-0.074 ± 0.012	-1.01	0.321
September 2002	-0.042 ± 0.012	1.69	0.093

Table A2.7: Factorial analysis of variance output based on a General Linear Model (GLM) analysis of regression of rates of change in faecal sedimentation rate with distance from cage centre for pooled collections of particulate deposition using sediment traps at Portavadie and Rhuba Stillaig fish farms. Collection dates February 2002 and April 2002.

(a) Analysis of variance table with date of collection, with distance as a covariate, after ln(x) transformation of faecal sedimentation rate. Seq = sequential, Adj = adjusted for entry order into the model, p = probability with significance at <0.05.

Source	df	SeqSS	AdjSS	AdjMS	F	p
Site	1	5.752	0.004	0.004	0.01	0.933
Distance	1	31.465	32.142	32.142	65.33	<0.001
Interaction	1	3.125	3.125	3.125	6.35	0.015
Error	56	27.551	27.551	0.492		
Total	59	67.893				

(b) Comparisons between regression slopes and intercepts and their respective group averages after ln(x) transformation of faecal sedimentation rate. p = probability with significance at <0.05.

Date	Intercept	T	p
Average	3.70 ± 0.16		
Portavadie	3.72 ± 0.16	0.08	0.933
Rhuba Stillaig	3.68 ± 0.16	-0.08	0.933
	Slope		
Average	-0.054 ± 0.007		
Portavadie	-0.037 ± 0.007	2.52	0.015
Rhuba Stillaig	-0.071 ± 0.007	-2.52	0.015

Table A2.8: Factorial analysis of variance output based on a General Linear Model (GLM) analysis of regression of rates of change in total sedimentation rate with distance from cage centre for combined collections of particulate deposition using sediment traps at Portavadie and Rhuba Stillaig fish farms. Collection dates February 2002 and April 2002.

(a) Analysis of variance table with date of collection, with distance as a covariate, after ln(x) transformation of total sedimentation rate. Seq = sequential, Adj = adjusted for entry order into the model, p = probability with significance at <0.05.

Source	df	SeqSS	AdjSS	AdjMS	F	p
Site	1	10.907	0.555	0.555	0.70	0.407
Distance	1	47.615	48.352	48.352	60.81	<0.001
Interaction	1	2.562	2.562	2.562	3.22	0.078
Error	56	44.524	44.524	0.795		
Total	59	105.608				

(b) Comparisons between regression slopes and intercepts and their respective group averages after ln(x) transformation of total sedimentation rate. p = probability with significance at <0.05.

Date	Intercept	T	p
Average	2.29 ± 0.20		
Portavadie	2.46 ± 0.20	0.84	0.407
Rhuba Stillaig	2.13 ± 0.20	-0.84	0.407
	Slope		
Average	-0.067 ± 0.009		
Portavadie	-0.051 ± 0.009	1.80	0.078
Rhuba Stillaig	-0.082 ± 0.009	-1.80	0.078

Table A2.9: Estimated deposition of faecal carbon to the seabed from Portavadie fish farm using the method of Gillibrand *et al* (2002).

Dy (m)	Dx (m)	Area (m^2)	y = 11.7 e-0.051x	Deposition C (kg d^{-1})
5	10	157	9.07	1.42
16	32	1451	5.17	7.5
26	52	2639	3.11	8.21
36	72	3896	1.87	7.29
46	92	5152	1.12	5.77
51	102	3048	0.87	2.65
	Total	**16343**		**32.84**
Average predicted deposition (t d^{-1})				**0.184**
Estimated deposition (kg t^{-1})				**178.5**
Average deposition (g C m^{-2} d^{-1})				**2.01**

Table A2.10: Estimated deposition of faecal carbon to the seabed from Rubha Stillaig fish farm using the method of Gillibrand *et al* (2002).

Dy (m)	Dx (m)	Area (m^2)	y = 8.42 e-0.082x	Deposition C (kg d^{-1})
5	10	157	5.59	0.88
16	32	1451	2.27	3.29
26	52	2639	1.00	2.64
36	72	3896	0.44	1.71
40	80	1910	0.32	0.61
	Total	**10053**		**9.13**
Average predicted deposition (t d^{-1})				**0.143**
Estimated deposition (kg t^{-1})				**63.85**
Average deposition (g C m^{-2} d^{-1})				**0.91**

Appendix 3: Videographic survey of Portavadie and Rubha Stillaig fish farm in October 2002. Compact disc.

Dive Time and Distance

1) Portavadie (12/10/2002)

	Outward	Back	Notes
Start/end time	13:18.34	13: 37.29	start cage centre
Cage Edge	19.11	31.08	note large white marker
5m	19.42	30.51	*
10m	20.18	30.34	
15m	20.58	30.17	*
20m	21.34	30.00	
25m	22.13	29.42	*
30m	22.49	29.26	
35m	23.39	29.09	*
40m	24.16	28.50	
45m	24.58	28.28	
50m	25.38	28.05	*
End/start	13: 26.15	13: 27.46	

* denotes white markers, remainder are black tape,
All divisions are 5m apart, except cage centre to cage edge at 11m.

2) Rhuba Stillaig (12/10/2002)

	Outward	Back	Notes
Start/end time	14:58.05	15: 09.28	start cage centre
Cage Edge	58.55	07.04	note large white marker
5m	59.20	06.42	*
10m	59.45	06.18	
15m	15: 00.08	05.57	*
20m	00.29	05.37	
25m	00.48	05.17	*
30m	01.10	04.52	
35m	01.30	04.35	*
40m	01.51	04.17	start of *Beggiatoa* mats denoting previous location of site
45m	02.11	04.00	
50m	02.30	03.39	*
End/start	15: 02.53	15: 03.17	

* denotes white markers, remainder are black tape,
All divisions are 5m apart, except cage centre to cage edge at 11m.

3) Reference (13/10/2002)

	Outward	Back	Notes
Start/end time	11: 36.19	11: 50.22	0m
5m	36.34		*
10m	37.00		
15m	37.26		*
20m	37.50		
25m	38.15		*
30m	38.40		
35m	39.05		*
40m	39.30		
45m	39.53		
50m	40.20		*
End/start	11: 40.38	11: 42.05	

* denotes white markers, remainder are black tape,
All divisions are 5m apart.

Appendix 4 – Statistical tables for Chapter 5 – Benthic Analysis

Table A4.1: One-way ANOVA on untransformed (unless otherwise specified) species abundance for macrofauna in 5 replicate 0.025m^2 Van Veen grab samples taken at Portavadie and Reference site in August 2001 and April 2002. Reference site in August 2001 2 grabs only. ^{1} = excludes reference sites, ^{2} = includes reference sites and all data Log_{10} transformed. df = degrees of freedom, F = test statistic, P = probability. * = Significantly different

	df	F	P
Within year^1 2001	3	1.96	0.160
Within year^1 2002	3	9.57	0.001*
Between years at			
P_5	1	46.21	>0.001*
P_{15}	1	2.72	0.138
P_{25}	1	10.70	0.011*
P_{50}	1	3.37	0.104
C	1	0.29	0.614
Within year^2 2001	4	9.34	>0.001*
Within year^2 2002	4	31.27	>0.001*

Table A4.2: Kruskal-Wallis Test comparisons between station differences in median values of $\log_{10}$ transformed Shannon-Weiner Index between August 2001 and April 2002 for macrofaunal samples collected in 5 replicate Van Veen grabs at Portavadie fish farm and reference site. Station subscripts represent distance from cage edge in metres. C = Reference Site. * = significant.

Station	H Statistic	df	p
P_5	6.82	1	0.009*
P_{15}	4.84	1	0.028*
P_{25}	5.77	1	0.016*
P_{50}	5.31	1	0.021*
C	3.82	1	0.051

Table A4.3: Kruskal-Wallis test comparisons of arcsin√ transformed Shannon-Weiner Index values standardized as proportions of the reference values between stations at Portavadie (2002) and Rubha Stillaig (2003) fish farm sites in respective years. H = test statistic, df = degrees of freedom, p = probability significant at < 0.05.

Station	H	df	p
5m	6.82	1	0.009
15m	6.82	1	0.009
25m	5.77	1	0.016
50m	0.53	1	0.465

Appendix 5 – Cage Dispersion module dialogue boxes

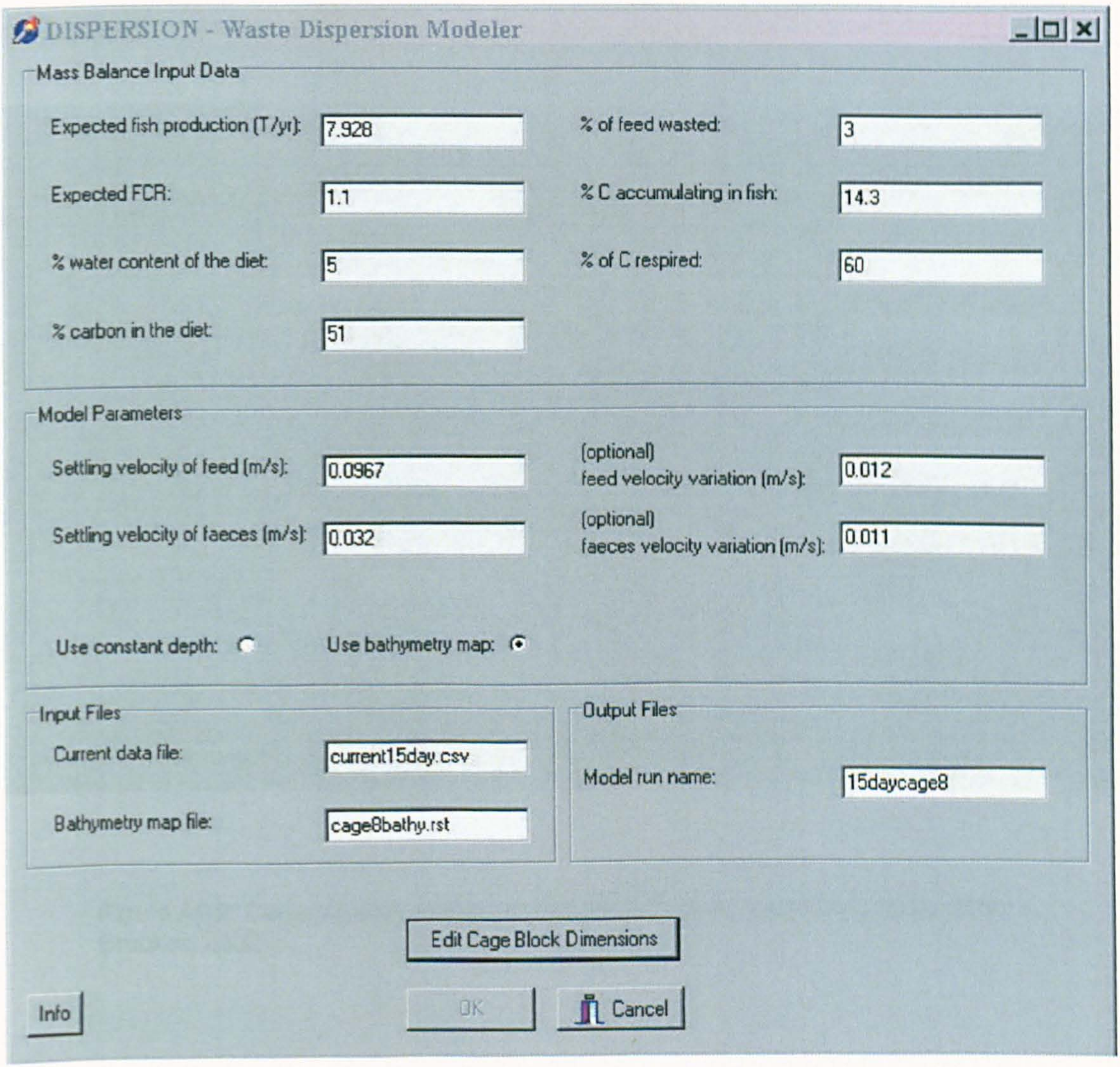

Figure A5.1: Parameter dialogue box for GIS waste dispersion model (After Brooker, 2002).

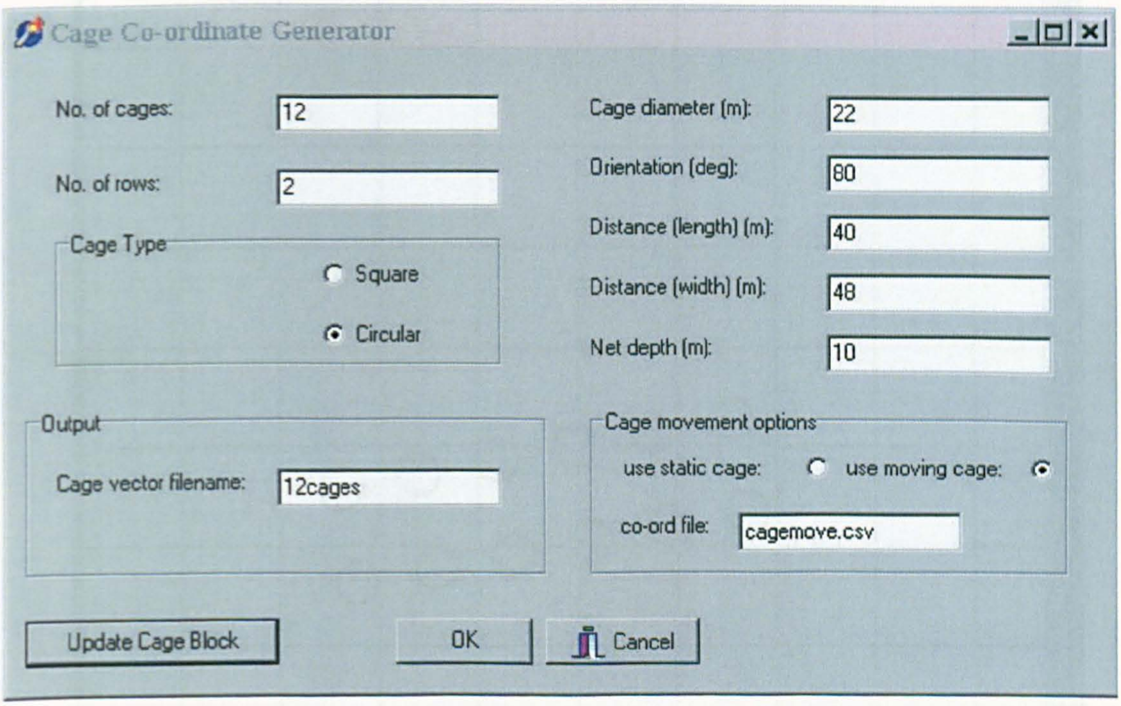

Figure A5.2: Cage-generator dialogue box for GIS waste dispersion model (After Brooker, 2002).

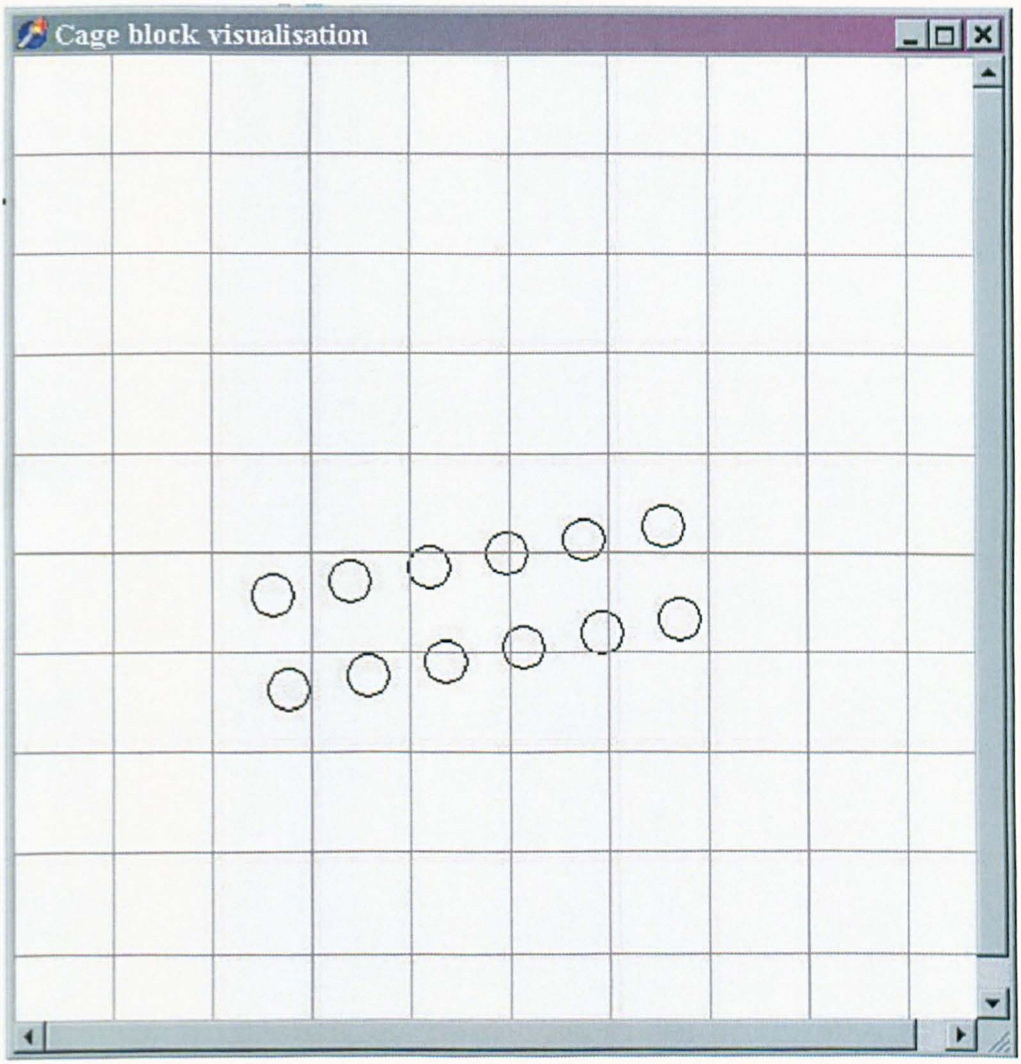

Figure A5.3: Cage block visualization box for GIS waste dispersion model (After Brooker, 2002)