ENVIRONMENTAL QUALITY AND CARRYING CAPACITY FOR AQUACULTURE IN MULROY BAY CO. DONEGAL

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1. INTRODUCTION

1.1 Introduction to the study

Concerns over recent expansion of the aquaculture sectors in Mulroy Bay have lead to several investigations on environmental quality within the area, including a recent report (in 2000) for the Irish Marine Institute by the Centre for Marine Resources and Mariculture (C-Mar, 2000). This study reported on changes in the physical and chemical environment within Muloy Bay over a two year data collection period, and from data collected by Marine Harvest Ireland, and related observed changes to aquaculture (notably salmon farming) outputs. The study also involved sediment bioassays using sediment collected in Mulroy Bay and a variety of test organisms to investigate whether any toxicants were present that may affect the benthic community.

The current study was commissioned by the Marine Institute to draw together the results of long term monitoring of water and sediment quality, and hydrography, within Mulroy Bay collected by the Institute of Aquaculture (IoA), Stirling, on behalf on Marine Harvest Ireland since 1986. The study examines trends in the physical and chemical environment within Mulroy Bay (predominantly using IoA data) since recording began, and relates results to the present activities within the bay. The aims of the study are:

- to produce a carrying capacity model for Mulroy Bay as a whole
- to produce maps of waste dispersion from fish cage sites
- to assess the environmental sustainability of current aquaculture activities in Mulroy Bay.

1.2 Description of Mulroy Bay and Surrounding Area.

Mulroy Bay has been described as a fully marine, fjordic inlet (C-Mar, 2000), situated adjacent to the Fanad peninsula on the Northern coast of Co. Donegal, Ireland. The Fanad peninsula is a dramatic outcrop of land that is bounded on its Eastern side by Lough Swilly, on its western side by the sheltered waters of Mulroy Bay, and to the north by the Atlantic Ocean (Figure 1.1).

For a relatively small land area, Fanad peninsula is rich in scenic variety that is largely unspoilt. The lack of trees, or their stunted and warped growth, except in the more sheltered parts of the peninsula, can be explained by frequent strong winds, mainly from the south and south-west. Fanad has fewer hours of sunshine than other parts of the country and its frost free period compares unfavourably with western coastal areas.

Dotted with historical landmarks and old forts, Fanad also boasts the County Genealogical Centre; several caravan sites; a community centre; a hostel; numerous primary schools[1] and churches; good quality guest houses and B&B accommodation. The Portsalon Golf Club (18 holes) is situated on one of the most scenic golf courses in Donegal with commanding views of Ballymastocker Bay, in Loch Swilly, and what is reputed to be one of the most beautiful beaches in the world.

[1] Seven of these schools have only one teacher each. Secondary school children travel by bus to Milford.

An extensive aquaculture industry is based in Mulroy Bay, consisting predominantly of salmon, mussel, scallop and clam production. The bay has also become one of the most important scallop spat collection points in Ireland. Like many areas in which aquaculture is being developed, Mulroy Bay and the Fanad peninsula are characterised by high quality landscape and environmental attributes, with typically rural features, a relatively traditional social context, and a generally low level of physical and economic development. The high quality, relatively unspoiled environment, important as it is for salmon farming, is equally important in the wider context, where there is a generally expressed concern for maintaining traditional features and natural environments.

Mulroy Bay contains a number of deep bays, separated by shallow sills, and can be split in to three main areas: Broadwater, Northwater and the Narrows (as shown in Figure 1.1). Tidal waters generally flow in to and out of Mulroy Bay through the Narrows, where current speeds and tidal ranges are often high. Current flows within the Northwater and Broadwater areas are often reduced and tidal ranges are low, due primarily to the observed "back-up" of water leaving the bay through the narrows. This causes a lag in high and low tidal periods compared with the mouth of the bay (IoA unpub. data). Several small streams drain in to the bay, notably at the southern end of Broadwater (near to Millford) where a halocline is often present, especially after periods of heavy rainfall (IoA unpub. data). In recent years, several small areas of new housing have been developed in the vicinity of Mulroy Bay, along with some expansion of the tourist industry.

Figure 1.1 Landsat image of Mulroy Bay, showing its convoluted nature and the individual basins which form the water body.

1.3 Aquaculture in Mulroy Bay

A variety of aquaculture producers currently operate in Mulroy Bay, including Atlantic salmon producers (Marine Harvest Ireland), mussel long-liners (McIlwain Mussels, Hugh Willhare, Martin Callaghan, Martine Cole, Brian Boyce and Brian Blaney), Pacific oyster producers (Eamon Coll, Taighte Mara Tir Lachan Teo), Manilla clam producers (Pascal and Holmes) and scallop producers (N.W. Shellfish, Deegagh Point Shellfish, Northwater Co-op). Production levels of Atlantic salmon within Mulroy Bay have remained fairly constant at approximately 800 to 900 tonnes per annum over the last five to ten years. Dramatic increases have occurred in shellfish production, with figures rising from 250 tonnes per annum to approximately 1000 tonnes per annum between 1994 and 1999 (C-Mar, 2000). Expansions by mussel producers have accounted for the majority of this increase, so that today, large areas of Broadwater in Mulroy Bay are covered by mussel lines.

1.3.1 Atlantic Salmon Production

The largest producer of Atlantic salmon in Co. Donegal is Marine Harvest Ireland Ltd. This company operates sites in both Mulroy Bay (headquarters) and neighbouring Lough Swilly. In terms of the relatively recent history of modern fish farming, the company has had a quite a long and sustained involvement in the area (IoA, unpublished data). Marine Harvest Ireland (known as Fanad Fisheries) was formed as a limited liability company to farm salmonids in Donegal in 1978. In 1979, with the financial support of six Irish shareholders, rainbow trout were stocked in marine cages at Kindrum, in Mulroy Bay. After a year it became clear that the project was not viable due to poor stock performance and poor market price. In 1981 a joint venture agreement was established with A/s Mowi of Bergen, providing access to the proven Mowi salmon strain and its advanced technology, as well as secure financing. The first salmon were harvested from Mulroy Bay in 1984, generating revenue, and stimulating further growth. Further equity dilution over subsequent years lead to 100% Mowi ownership. In 1996 Fanad Fisheries was bought by the Norwegian company, Norsk Hydro, as part of its expansion into Irish and Scottish salmon industries (the company becoming Hydro Seafood Fanad). However, in 2000 the company was acquired by Nutreco (the “parent” of Marine Harvest companies). As a result the company is now Marine Harvest Ireland.

Two freshwater smolt units have been developed by Marine Harvest Ireland at Kindrum (1983/4) and Altan (1986/7). Targets for on-growing production at sites within Mulroy Bay were completed by 1986, after buying out the ESB and Salmara sites within the bay. Further expansion was initiated through a pilot trial for salmon production in neighbouring Lough Swilly which started in 1985. The expansion in a number of sites has allowed regular fallowing, increased separation of year-classes and the protection of Fanad's disease free status.

1.3.2 Shellfish Production

A variety of shellfish species are currently cultivated in Mulroy Bay, i.e. mussels, oysters, scallops and clams. Scallop cultivation throughout Donegal concentrates on the collection of spat in Mulroy Bay, which has been described as one of the best scallop spatting grounds in Europe (Marine Institute, 1999). Currently three companies collect scallop spat and two scallop farms operate within Mulroy Bay.

Most of the shellfish waters around Donegal's coast have been designated as Grade A shellfish waters, allowing the production of very high quality mussels, oysters and scallops for export with a minimum requirement for depuration. Mussel cultivation within Mulroy Bay began in 1979, following seed collection trials, in Broadwater, and although initial development of the industry was slow, the bay currently supports six mussel farms that produce over 800 tonnes of mussels in total (C-Mar, 2000).

The development of commercial oyster beds in Mulroy Bay is currently limited to three sites, with only a few tonnes being harvested each year. Similarly the production of Manila clams is in its infancy, with only one site currently under production.

1.4 Other aquatic resource uses in Mulroy Bay

1.4.1 Housing and sewage

There are several villages around Mulroy Bay, and many individual houses, all of which discharge either treated or untreated sewage into the bay. Sewage and inputs of nutrients from the surrounds of Mulroy Bay (Donegal County Council, pers. comm.) suggest that inputs are minimal having a total value of only 1320 person equivalents (PE). This may be broken down into 100 PE from a septic tank outfall input in Carrickart which discharges into the narrows, 100 PE secondary treated sewage discharging into the bay from Kerrykeel, 200 PE untreated sewage discharging into the bay from Kerrykeel and 20 PE from a septic tank near Milford. This data gives no indication either on the estimations used for the calculation of PE or inputs from other villages and areas such as caravan parks around the bay. This therefore would be expected to be a severe underestimation of the inputs from human sources going into the bay.

In addition, there is no information on the seasonal variability of this data. Donegal is an important tourist area within Ireland, and it should be expected that a considerable increase in the inputs of human waste nutrients would occur during the summer months. However, data from the study, undertaken by the Letterkenny Institute of Technology between June and August 1997 (Slater *et al.,* 2001), showed that levels of nurtrients were not particularly high in the bay during these months.

1.4.2 Other industries

Considerable amounts of agricultural farming is undertaken on the periphery of Mulroy Bay and could be expected to add considerable amounts of nutrients to the bay due to runoff into rivers. This can be estimated by using data on the nitrogen and phosphorus rich fertilisers used in the various catchments surrounding the bay. This has not been done during this survey due lack of reliable data. However, environmental monitoring of overall nutrients within Mulroy Bay, reported in Chapter 3, has necessarily included any terrestrial based anthropogenic inputs and thus have taken them into account when considering changes in environmental quality.

Boat building and cleaning is undertaken in the Mulroy Bay narrows, at Meevagh Boatyard. This is likely to input toxic substances such as heavy metals (including cadmium, chromium, copper and mercury) and solvents into the bay, giving potential effects to shellfish and fish resources, but not likely to input nutrients which may effect the overall nutrient capacity of the system.

1.5 Socio-economic infrastructure

Perhaps the four most important industries in Co. Donegal are marine fisheries, aquaculture, tourism and textiles. A report produced by the Marine Institute in 1999 stated that Co. Donegal can be viewed as Ireland's premier sea fishing county, and also possesses a significant fish processing industry with an annual turnover of IR£112m (Marine Institute 1999, after O'Callaghan and Associates, 1998). The relatively large textile industry in the county provides high levels of employment in certain areas, but this tends to be focused mainly around centres of higher population density, due to the high labour requirement, meaning that rural and remote communities rarely benefit from this sector. It has also been stated that due to the non-dependence of the industry on indigenous resources (other than labour), it does not guarantee employment for the future since re-location may occur. In contrast, the marine fisheries and developing aquaculture sector have been shown to provide significant employment and economic benefits to Donegal's rural economy, while using the natural resources the county has to offer. These industries could provide more long-term economic stability to the areas in which they exist. The higher than average unemployment figures in Co. Donegal, combined with apparent under-development of marine resources, prompted a recent government initiative called the "Donegal Marine Action Plan " (DMAP) that focused on the development of the county's marine based industries. Financial investment of over IR£85m has been allocated to promote and develop the aquaculture, sea fisheries and marine technology sectors, with an aim of improving local economies and future job prospects (Marine Institute, 1999).

As described earlier, several types of commercial aquaculture are practised in Donegal, including finfish culture (mainly salmon) and shellfish production. Seaweed collection and processing are also practised on the west coast of Donegal. Mulroy Bay in particular is home to one of the largest finfish producers in Donegal, namely Marine Harvest Ireland Ltd, as well as several shellfish producers, all of which provide employment to the local area.

Marine Harvest Ireland is, in economic terms, the biggest aquaculture company in the Mulroy Bay and Fanad areas, providing high numbers of jobs at both its fish farm sites and new processing plant at Rinmore Point. In 1999, Marine Harvest Ireland (as Hydro Seafood Fanad) had an annual turn over of IR£12m, and contributed IR£2.1m in wages to the local economy (Marine Institute Report, 1999). Aside from direct benefits to employees, the company also brings benefits to both local and national goods providers and services (Marine Institute, 1999). Figures published in 1999 showed that the company employed over 165 staff on full time contracts, and several part time staff, offering many of their work force relatively flexible working hours. This has proved to be important for the maintenance of the local rural economy, since many staff (estimated to be approximately 35% of the male workforce) are also part time farmers. Similarly, many female staff working in the company's processing plants work on a flexible part time basis since they are unavailable for work outside of the home or school hours (Marine Institute, 1999).

Aside from salmon farming, shellfish aquaculture is also an important source of income and employment in Donegal, with around 40 co-ops or companies operating throughout the county, 13 of which operate in Mulroy Bay. Shellfish production in the county ranges from mussel cultivation (worth IR£340,000 in 1998/9), Pacific oyster production (valued at IR£606,250 in 1998/9) native scallop and oyster cultivation (valued at IR£405,000 in 1998/9) to native clam collection (no figures) (Marine Institute, 1999). In total, the shellfish industry employs over 125 people throughout the coastal regions of Donegal, including Mulroy Bay, where almost 1000 tonnes of produce were harvested in 1999 / 2000 (C-Mar, 2000).

2. PHYSICAL ENVIRONMENT

Removal of wastes from aquaculture and the replenishment of "used" water with fresh clean oxygenated water is essential for both the ability of the environment to sustain waste input and for the aquaculture resource. This process is mainly concerned with the movement of water within the environment. In coastal marine waters there are two primary processes which give rise to water movement; tides and waves. The latter is a product of wind speed and direction and the distance over the water (fetch) which it can input frictional energy. This is difficult to both predict and calculate and is often considered only a modifying factor in most coastal environments.

Water movement due to tidal influence and subsequent flushing are considered the most important mechanism in coastal systems in NE Europe for removing and diluting aquaculture waste (Perez*, et al*, in press). These are often predictable and can be measured directly using both simple and sophisticated instrumentation.

Mulroy Bay is highly convoluted and distinctly divided into areas by "narrows" or restrictions to water flow. These areas are known by different names; the Outer Bay, the Narrows, Broadwater and Northwater. These distinct areas are likely to have different hydrographic regimen and thus water movements. Therefore this naming method will be adopted to divide Mulroy Bay into distinct regions, for the purposes on water flow calculation and characterisation. Ultimately, they will all be important in the overall capacity of Mulroy Bay for aquaculture but are likely to have their own independent capacities, which are based around their own characteristic water movements. However, as the areas are interconnected, waste and water flows from one area to the others. This may transport nutrients from one area to another thus utilising the environmental services of the new area to process (dilute) wastes and therefore adding to the environmental capacity.

2.1 Hydrographic characteristics and water flow of Mulroy Bay

Hydrographic data has been collected for the eight fish cage sites within Mulroy Bay (Figure 2.1) between 1996 and 2000 (IoA, 1996, IoA, 1997, IoA, 1998, IoA, 1999 & IoA, 2000). These data were collected over a minimum of 15 days to collect information over a complete half lunar cycle (i.e a single spring tide/neap tide cycle). Measurement was made using direct self-recording meters (Valeport BFM 105), which employ an impeller for which the revolutions per time are counted and current speed measured. The current meters swivel freely on the moorings in the direction of the current and this is recorded using an "on-board" flux-gate compass. Measurement was made over an averaging period of 1 minute every 20 minutes during the deployment period. To allow for variable tidal flow with depth, currents were measured near the surface (approx. 3 m from lowest tide during deployment) and 3 m from the seabed.

Wind data was recorded using a weather station. This was not used directly in calculation of water movement but its impact on flow was observed. These data were used for modelling the dispersion of particulate wastes (see Chapter 4). Data was rejected from these models if the wind strength was too strong.

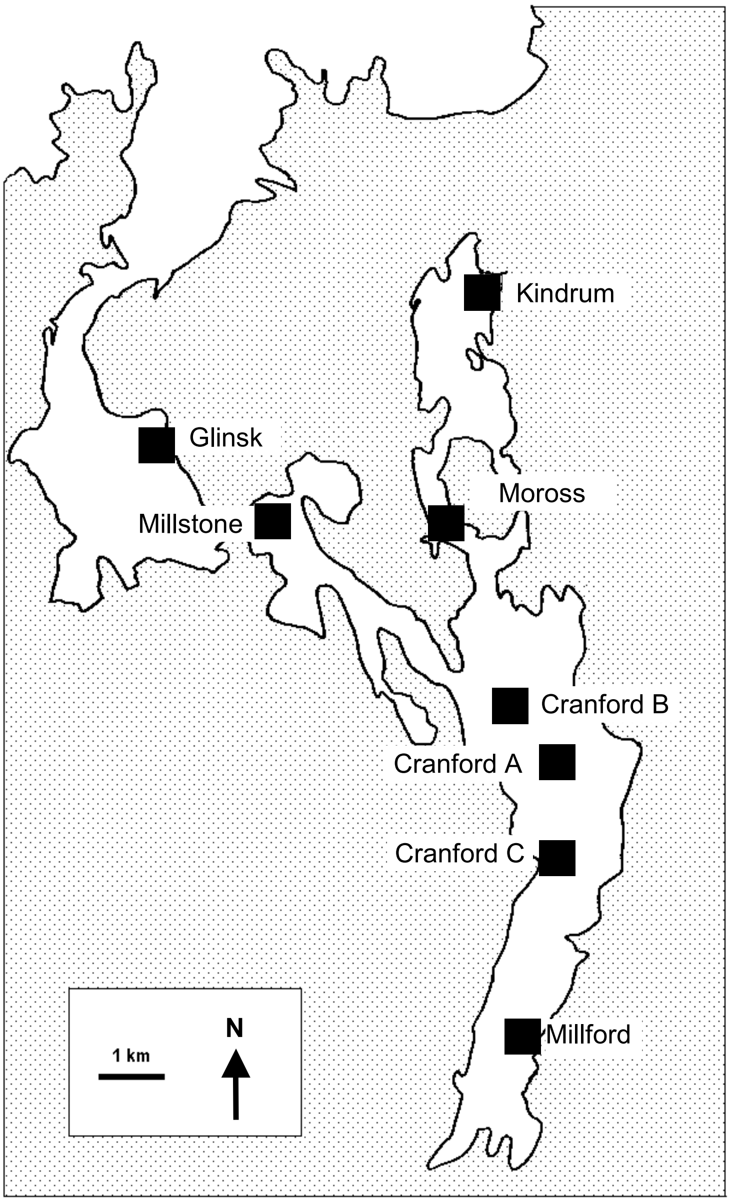

Figure 2.1 Mulroy Bay and approximate positions of the fish production sites.

The distribution of the sea cage sites throughout Mulroy Bay ensured that current characteristics were obtained for each of the distinct basins or areas. This enabled the characterisation of the water flow in each area. To do this scatterplots were created, which are a function of direction and current speed and residual current flow, and these are given below. Descriptions and conclusions were made for each area, within the context of the bay as a whole.

Northwater

Two cage sites are located within the Northwater; Kindrum at the northern end and Moross within the narrows at the southern end. Hydrographic current flow is given in Figures 2.2 to 2.3 for Kindrum and 2.4 to 2.5 for Moross.

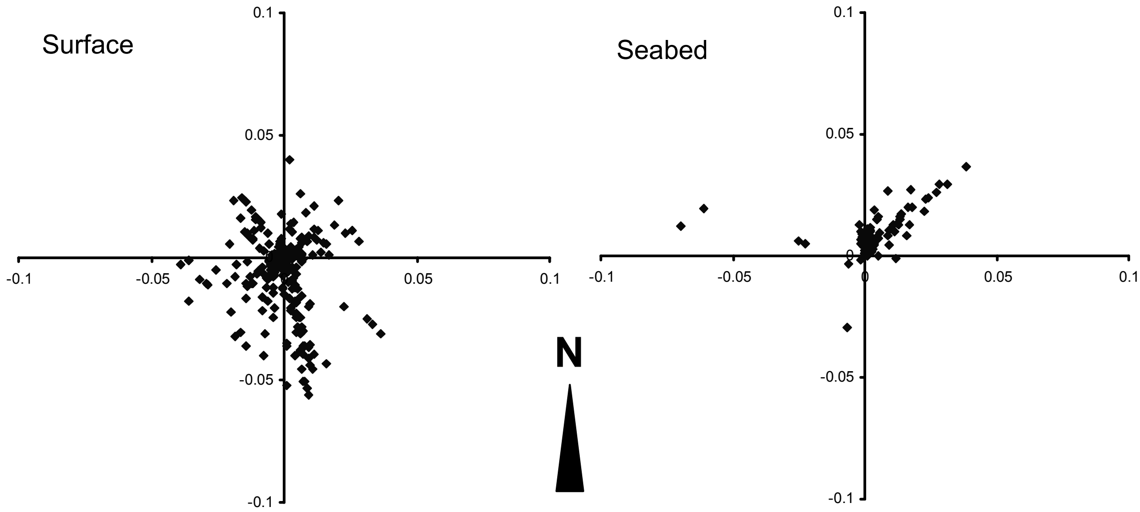

Figure 2.2 *Scatter plot for surface and seabed currents at Kindrum (13 to 28 Sept 2000). Axis units are in m/s.*

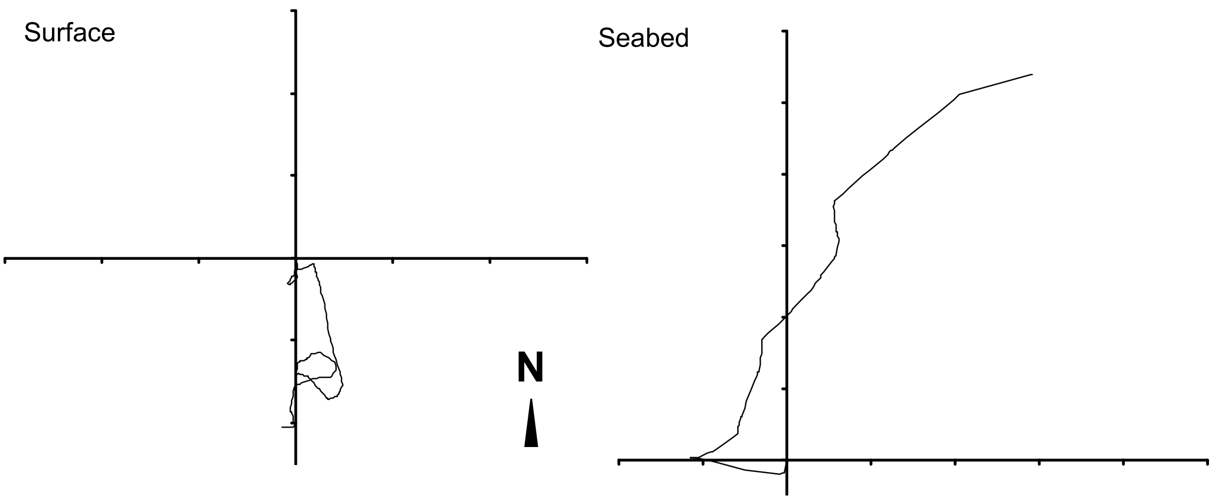

Figure 2.3 *Cumulative vector plots showing residual currents at Kindrum (13 to 28 Sept 2000). Axis units are: Surface = 1 km, Seabed = 200m.*

Currents within Kindrum Bay at the northern end of Northwater were slow with an average current speed over a spring/neap tidal cycle of 0.004 m/s and 0.001 m/s at the surface and seabed respectively. This may be considered as quiescent waters during the majority of the tidal cycle. Even so, there was an apparent difference in current direction with depth at Kindrum. The surface currents were primarily to the south whereas the seabed currents were very slow but to the north-east. Their residuals were very weak suggesting a slow turnover, flushing time. This also indicated that there was likely to be little movement of waste material and a potential build up of particulate waste beneath cages and little movement of dissolved nutrients. However, the site is deep (approximately 42 m) and may provide an opportunity for some further dispersal of solid waste falling to the seabed. The slowness of the currents is unlikely to lead to re-suspension of settled material.

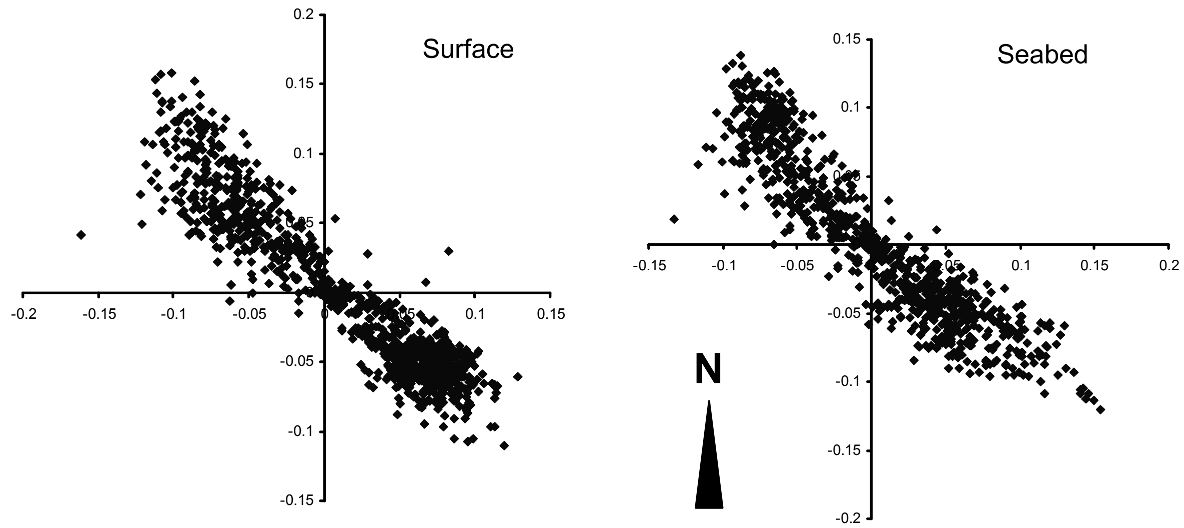

Figure 2.4 Scatter plot for of speed and direction surface and seabed currents at Moross (13 to 28 Sept 2000). Axis units are in m/s.

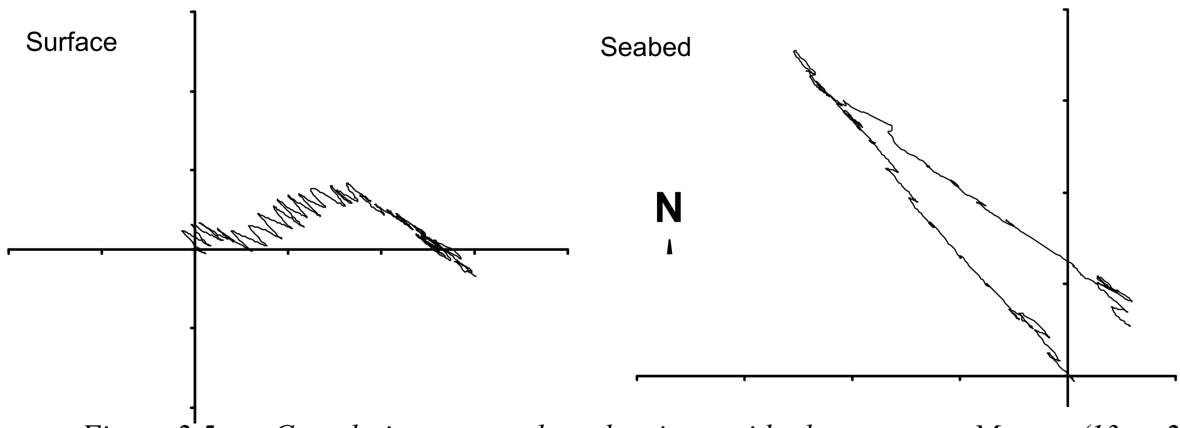

Figure 2.5 Cumulative vector plots showing residual currents at Moross (13 to 28 Sept 2000). Axis units are 5 km graduations.

The current characteristics at Moross, within the narrow channel at the entrance of the Northwater were different and more distinct (see Figure 2.4 and 2.5). The mean current speeds ranged from 0.075 m/s at the surface to 0.062 near the seabed. These were considerably faster than those at the northern end, suggesting the potential for more replenishment of water and distribution of waste material. The currents flowed distinctly parallel to the shore, though the residuals were different at the surface than the seabed, showing an easterly and north-west to south-east direction respectively.

These results suggest that there is certainly a potential for build-up of waste material in Kindrum Bay, but that the southern area within Northwater would support a greater dilution and distribution. However, any distribution from this area would be into the northern end of the Broadwater, adding to nutrients and increasing oxygen demand, within this area of water.

Broadwater

The Broadwater is the largest area within the Mulroy Bay complex, and is occupied by the largest concentration of aquaculture. Four fish cage sites occupy this area, three of which have collected hydrographic data; Cranford A, Cranford B and Cranford C (see Figure 2.1). These were distributed along the main axis of the bay and give a good indication of the water flow characteristics for the mid and northern bay. The very shallow, southern area within the Broadwater has no collected current data as it largely dries on extreme low spring tides.

Cranford B was the most northerly of the three cage blocks within the Cranford area of Broadwater. It was situated just north of the entrance to the narrows. The mean current speeds were 0.08 and 0.06 m/s at surface and seabed respectively, indicating some dispersive capacity. The scatter plots in Figure 2.6 indicate that they were primarily parallel to the nearest shoreline in a north/south direction. The residuals flows though were different with depth, having a strong residual to the south at the surface and an indistinct residual toward the east near the seabed. This suggests that the surface waters were most important for "emptying" of the Broadwater/Northwater at this point.

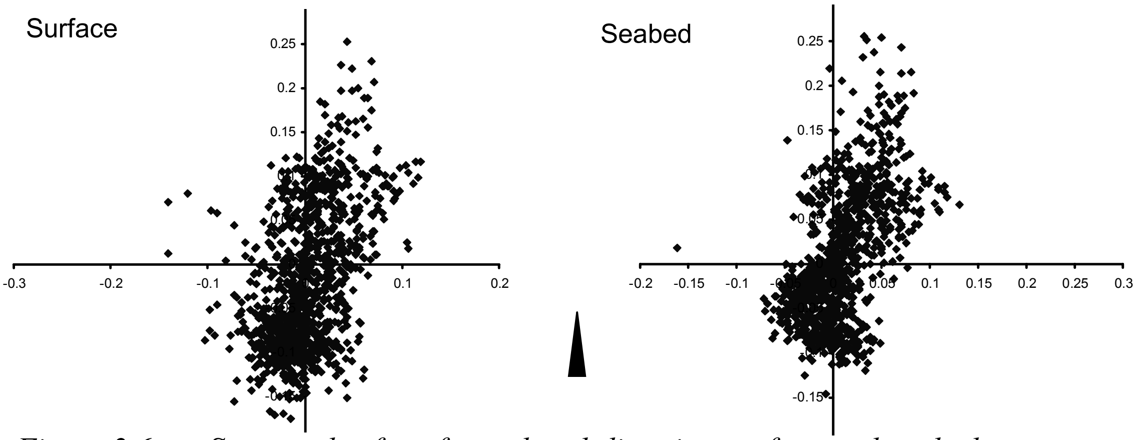

Figure 2.6 Scatter plot for of speed and direction surface and seabed currents at Cranford B (13 to 28 Sept 2000). Axis units are in m/s.

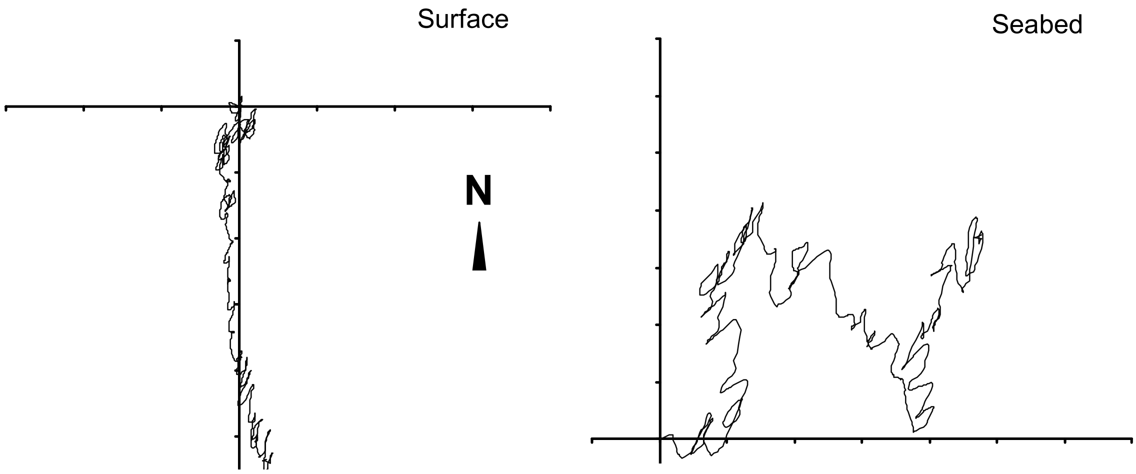

Figure 2.7 Cumulative vector plots showing residual currents at Cranford B (13 to 28 Sept 2000). Axis units are; surface = 5 km, and seabed = 2km, graduations.

Cranford A was situated to the south of the entrance to the Narrows (see Figure 2.1). The characteristics of water flow are given in Figures 2.8 and 2.9. Though currents were more diverse at this site they flowed primarily to the west and in particular, on the surface to the north-west. Seabed currents were less distinct. These results suggest that the main current flow here tended to be out of the basin towards the Narrows. This was shown clearly by the residual current flows at the surface, thus suggesting much of the waste from cages at this position would be taken quickly from Broadwater and having little potential for retention. However, current flows at this site were relatively slow (compared to Moross, for example) with average speeds being 0.04 m/s and 0.01 m/s at the surface and seabed respectively.

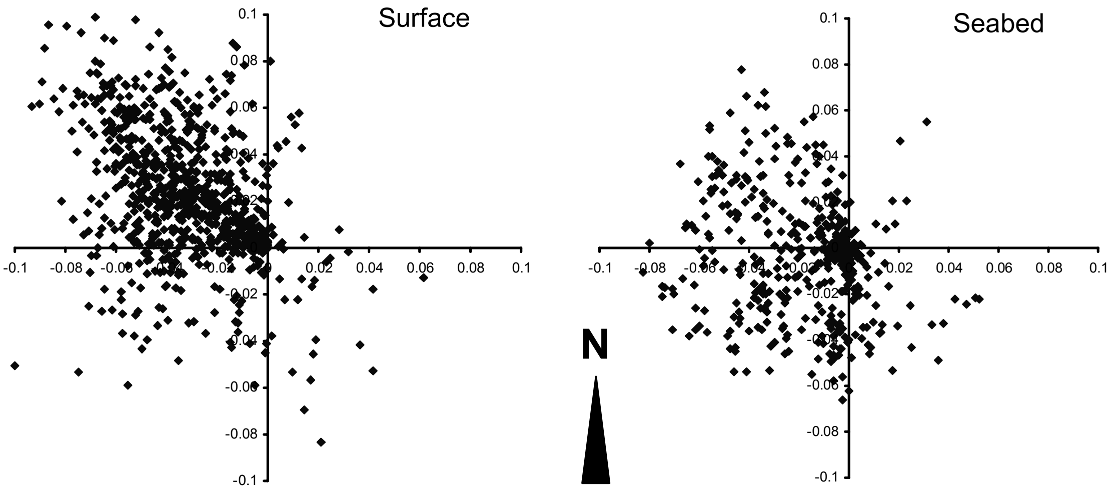

Figure 2.8 Scatter plot for of speed and direction surface and seabed currents at Cranford A (13 to 28 Sept 2000). Axis units are in m/s.

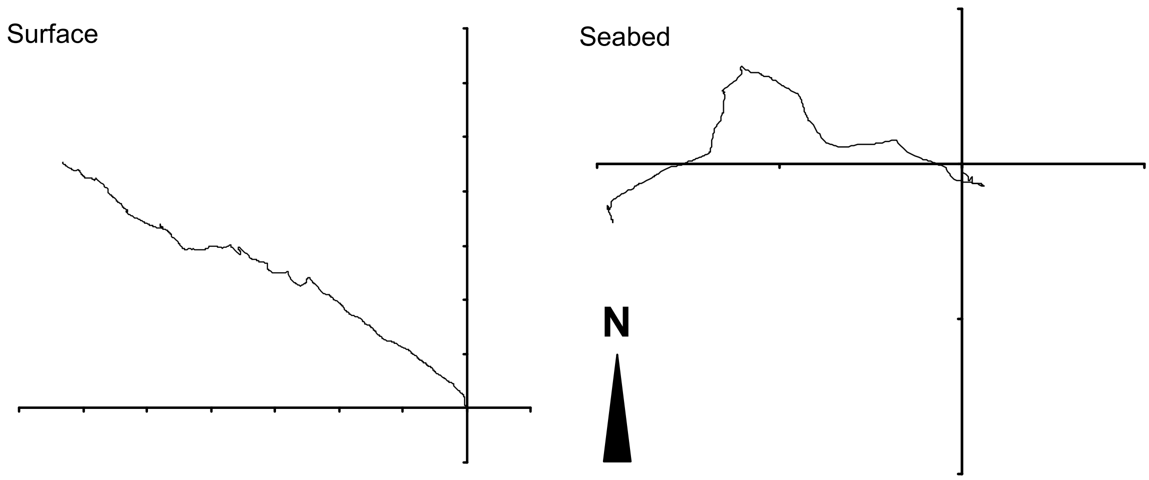

Figure 2.9 Cumulative vector plots showing residual currents at Cranford A (13 to 28 Sept 2000). Axis units are 5 km graduations.

Cranford C was the longest used site within the Cranford area for cage production. However, it was also the smallest producing site. It was situated towards the south of the area near the Cranford Pier. The characteristics of current flow for surface and seabed at this site are given in Figures 2.10 and 2.11. These illustrate the current speed and direction and the residual flow over the current meter deployment period.

The currents at Cranford C site flowed primarily to the south and are stronger on the surface than near the seabed. This is confirmed by the clear residual flows to the south over the deployment, with that at the surface being considerably quicker than at the seabed. However, the overall currents here were characteristic of slow flows with the average surface and seabed currents being 0.03 m/s and 0.02 cm/s respectively. These currents are unlikely to be dispersive and move cage wastes, which are likely to settle locally.

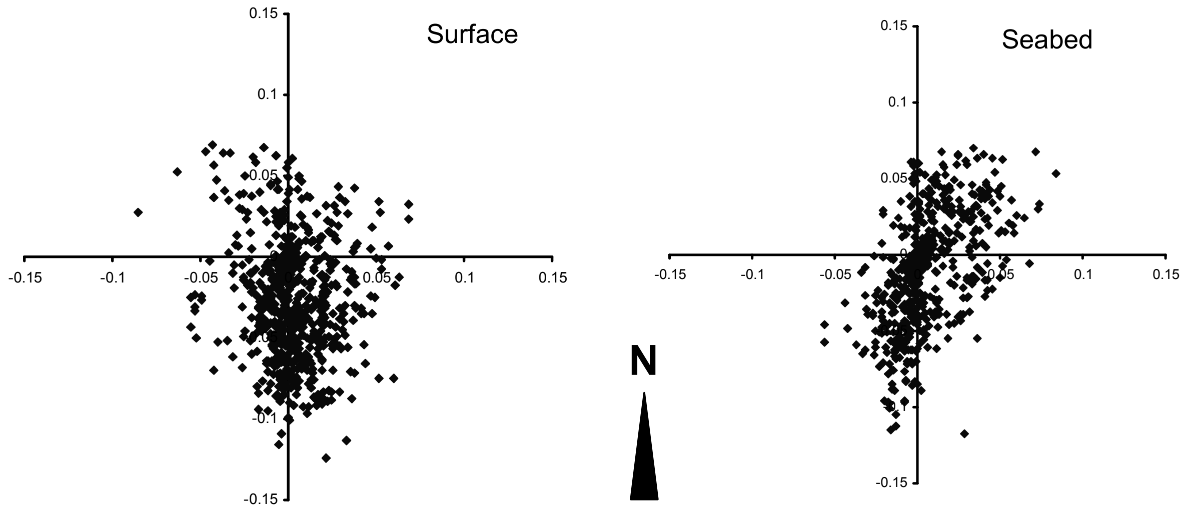

Figure 2.10 Scatter plot for of speed and direction surface and seabed currents at Cranford C (13 to 28 Sept 2000). Axis units are in m/s.

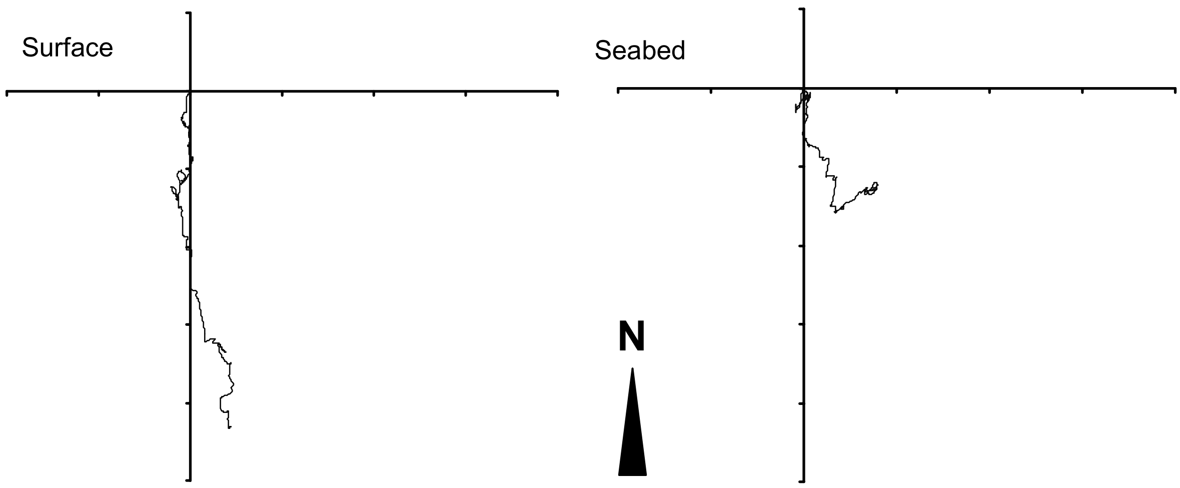

Figure 2.11 Cumulative vector plots showing residual currents at Cranford C (13 to 28 Sept 2000). Axis units are 5 km graduations.

Narrows

The Narrows is a thin channel of water through which the waters of the Northwater and Broadwater must flow in order to empty to the Atlantic Ocean. The narrows are such that this cannot be fully achieved during a single tide as backing up occurs. However, the nature of the narrows ensures a fast current flow through this area. There were two fish farm sites operating within the narrows.Only data for a single site, at Millstone, is given here to illustrate the nature of the currents in this area of the Bay. This site was situated within a shallow bay to the north of the narrows. Current characteristics are plotted in Figures 2.12 and 2.13. These indicate very strong currents primarily to the west at both surface and seabed. This is confirmed with a strong residual in this direction. The average surface and seabed currents were 0.10 m/s and 0.09 m/s respectively, testifying to the highly dispersive nature of the site. The direction of the dispersion is important as the data suggests that waste will be consistently taken towards the ocean, rather than back into the waters of Broadwater. Though not given here, the same is true for the current characteristics at the fish farm site in Glinsk Bay.

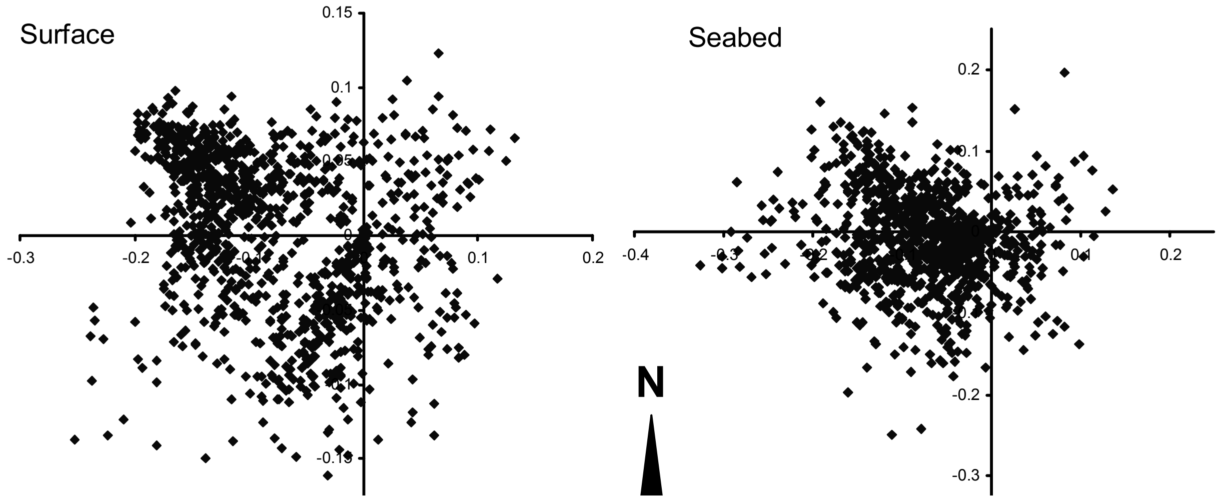

Figure 2.12 Scatter plot of speed and direction for surface and seabed currents at Millstone (30 June to 14 August 1997). Axis units are in m/s.

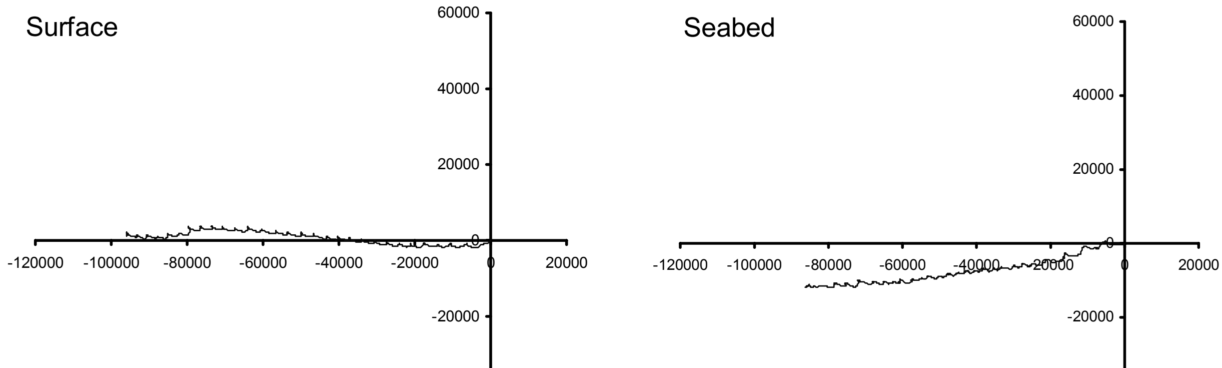

Figure 2.13 Cumulative vector plots showing residual currents at Millstone (30 June to 14 August 1997). Axis units are 5 km graduations.

The residual directions for the fish farm sites within the Mulroy Bay area are summarised in Figure 2.14. These give the basic directions of residual flow for the surface and seabed currents but not relative speeds. However, it serves as an indication of the likely overall directions of dispersion of wastes as a result of currents throughout a spring/neap tidal cycle within the Bay.

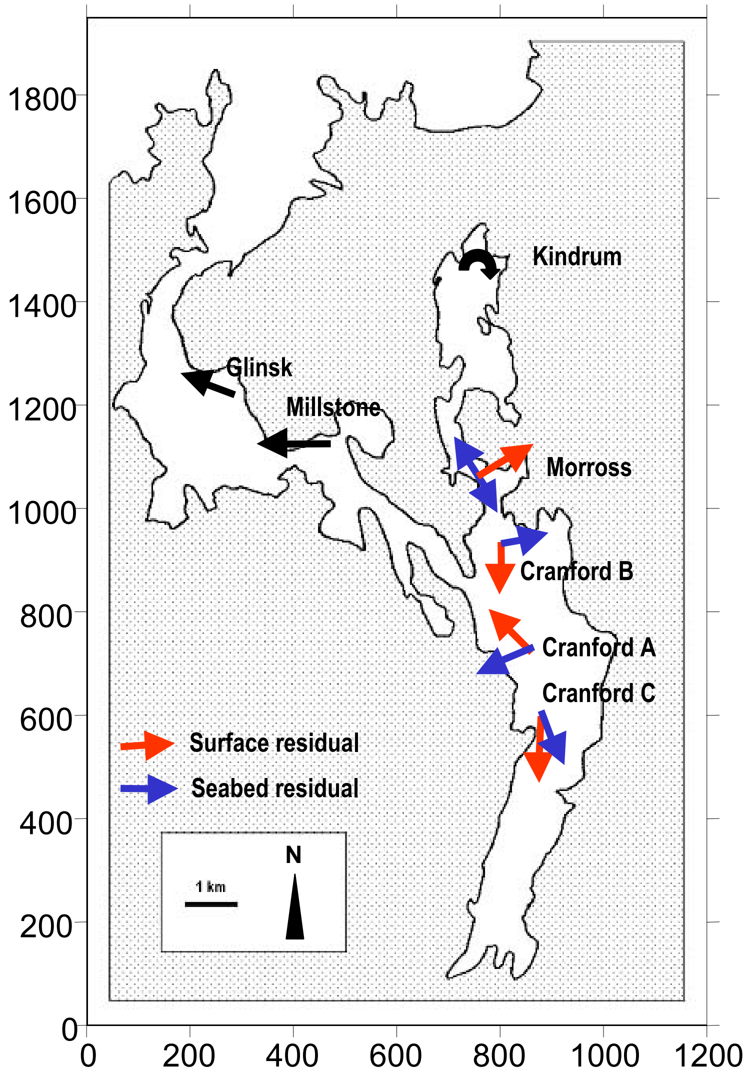

Figure 2.14 Summary of residual current directions for surface and seabed flow over a spring/neap tidal cycle at each fish farm location within Mulroy Bay. Black arrows indicate where residuals were in the same direction throughout the water column.

The measured currents within the bay show clearly the differences between the various areas of the bay and thus the requirements for treating these as distinct areas for calculation of flows and exchange rates. The Narrows generally had a very fast current flow and in the positions measured tended to flow out of the system towards the ocean. The strong residuals in this direction throughout the whole water column, indicated that both dissolved and particulate waste material was taken, in general, to the west and thus not did increase nutrient loading on the inner bay area (Broadwater and Northwater).

Currents within Kindrum Bay were very quiescent in terms of speeds and residuals, suggesting that there was the strong possibility of nutrient retention, particularly in terms of distribution of solid wastes from the cages. The Moross channel, to the south of the Northwater, had a distinct "in and out" current flow parallel to the shore but variable residual, suggesting that while dissolved nutrients were likely to be dispersed into Broadwater or Kindrum Bay, particulate material was likely to be incorporated locally to the seabed. There was, however, some capacity for resuspension of settled material at this site at the accepted threshold speed, at which settled material is moved up into the water column, which was attained for 25.2% of the current meter deployment period.

Currents within Broadwater, in terms of general flow and residuals, as illustrated by the three Cranford sites, were largely governed by their position in relationship to the entrance to the narrows. Cranford B site indicated some dispersive capacity with current flow directions indicating a likelihood of movement of waste towards the Narrows and towards the Moross channel. The surface residual gave a southerly direction toward the narrows. This site showed a reasonable capacity to distribute waste material, initially within the water column and then due to resuspension, with the threshold speed being exceeded for 28.3% of the deployment period. Tidal currents at Cranford A and Cranford B sites were considerably slower and less directionally distinct. These indicate limited capacity for dispersal of cage wastes, including that due to resuspension. Thus it is likely that particulate material will be distributed very locally to the cages at these sites. However, the residual at Cranford A suggested that the majority of any dispersion occurring would be toward the narrows. There is no data on currents available for the southern area of Broadwater, near Milford. Observation and modelled data (C-Mar, 2000) indicated that these waters exhibited low flow characteristics and thus nutrient inputs and waste within these areas were likely to be retained for some period of time, and that solid waste will be distributed locally.

2.2 Flushing times and exchange rates for Mulroy Bay

Based on the hydrographic characteristics described above, the different areas within Mulroy Bay (see Figure 1.1) should be considered separately in terms of flushing and exchange rates. These measures can be used to calculate the retention of dissolved nutrients, dissolved oxygen turnover and particulate food availability within the embayments, thus allowing modifying factors such as replenishment of food and dissolved oxygen to be taken into account. The more localised dispersal of particulate material from fish cages can be modelled using hydrographic characteristics and production data (see Chapter 4).

Areas and tidal volumes have been estimated from digitised charts of Mulroy Bay and depth variation data collected by the Valeport current meters during deployment over a spring and neap tidal cycle. These data have been entered into the computer program Surfer™ (Golden Software Ltd) which calculated the area and mean volume at low water and high water within the various embayments. Using variation in volume the tidal volumes and thus the exchange volumes have been calculated. Results are given in Table 2.1 for each area within the loch system. From this the flushing time in "number of tides for complete water exchange" has been estimated.

	Broadwater	Northwater	Narrows	Outer Bay
Area (m^2)	19671653	5255609	17240178	37826700
Volume LW (m^3)	125429368	43709546	41950899	226960000
Volume tide (m^3)	38639700	9913060	24075200	68088100
Exchange (m^3/day)	77279400	19826120	48150400	136176200
Flushing time (no. of tides)	3.25	4.41	1.74	3.33

Table 2.1 Estimated areas, tidal volumes, exchange and flushing times for areas within Mulroy Bay.

These results clearly show the variation between the different areas within Mulroy Bay. Through tidal exchange, the Narrows had the quickest flows where water is completely exchanged over 1.74 tides. Broadwater and Northwater were considerably slower, exchanging water over 3.25 and 4.41 tides, respectively. In terms of water quality these results suggest that nutrients would be retained 2 times as long in Broadwater and 2.5 times as long in Northwater than in the Narrows. Oxygen renewal would also be this much slower in these areas due to reduced tidal water exchange.

However, water flushed from one region to another will add to the loading of that system and not give a true representation of the effective flushing time for the region to the ocean. The overall mean tidal exchange volume for the Inner Bay (Broadwater and Northwater) (T_{Inner}) to the ocean can be calculated by taking the tidal exchange of the Narrows into account by using:

$$T_{Inner} = (T_{Broadwater} + T_{Northwater}) - T_{Narrows} \text{ (Where T is tidal volume)}$$

From this approximate flushing time F (in tides) to the ocean may be calculated by:

$$F = (V_{Broadwater} + V_{Northwater})/T_{Inner} \text{ (Where V is volume at low water)}$$

And approximate flushing time (in tides) to the ocean for Broadwater and Northwater will be:

$$F_{Broadwater} = \frac{V_{Broadwater}}{T_{Inner}\left(\frac{T_{Broadwater}}{T_{Braodwater} + T_{Northwater}}\right)}$$

Therefore, using the above calculations the:

- Effective volume of exchange per tide between the inner bay and ocean (m3) = 24477560
- Flushing time between Broadwater and ocean (no of tides) = 6.91
- Flushing time between Northwater and ocean (no of tides) = 8.75

C-Mar (2000) indicated that the flushing rate for the Broadwater was 11.5 tides and for Northwater was 15 tides. These were considerably longer than those found during the present study. The difference may be due to the differences in the hydrographic data used. Here 15 days of data has been used from 5 sites throughout Mulroy Bay allowing spring and neap tides to be taken into account. The deployment time for current meters by C-Mar as this varied between 4 days and 1 month at five positions in the bay using a variety of equipment, including direct recording meters, ADCP and current drogues.

3. ENVIRONMENTAL QUALITY OF MULROY BAY

3.1 Introduction

Environmental quality in Mulroy Bay has been the subject of many previous and on-going studies on behalf of both environmental regulators and the aquaculturists that operate within the bay. The Institute of Aquaculture (University of Stirling) has collected water and sediment data in Mulroy Bay on behalf of Hydro Seafood Fanad since 1986 as part of the company's on-going environmental monitoring programme, and has recently completed hydrographic surveys at all of the existing salmon cage sites within the bay. The data collected over the years is presented in the following sections, where long-term trends and changes in water and sediment quality in Mulroy Bay will be described in order to help determine whether increased aquaculture production has had an overall impact on environmental quality.

3.2 Water Quality

3.2.1 Introduction

Several aspects of water quality were measured throughout Mulroy Bay since the Institute of Aquaculture began its annual monitoring surveys in 1986. These included the measurement of both physical (e.g. temperature) and chemical (dissolved oxygen (D.O.), pH, salinity, nitrite, nitrate, ammonia, dissolved reactive phosphorus (DRP) and chlorophyll 'a') parameters at various locations in the Bay (shown in Figure 3.1). Water quality has been assessed annually at 8 to 12 sites (longitude and latitude positions given in Table 3.1), at the same time each year (i.e. during the final week of July), thus allowing comparisons to be made between survey results from different years.

The temperature, D.O., salinity and pH of a water body, and fluctuations in these parameters, are important in determining the types of organisms that can exist. Temperatures are known to fluctuate seasonally in Mulroy Bay (as would be expected) with winter temperatures being lower than those during the summer (C-Mar, 2000). Similarly, it has been observed that salinity values are generally slightly lower during the winter than in the summer, with the occasional formation of haloclines (salinity stratification) in the Broadwater area, likely due to increased rainfall in winter (C-Mar, 2000). Dissolved oxygen concentrations are known to fluctuate with temperature, weather conditions and biochemical / biological demand (due to respiration of aquatic organisms), meaning that D.O. could become low in warm waters with a high density of organisms (e.g. near to fish cages or mussel lines, or near to sediments where bacterial loads are high) during the summer months.

Nitrogen and phosphorus are important requirements for phytoplankton growth in the marine environment and are largely responsible for its control. Levels of these nutrients vary with depth, but they are generally lower in the surface layer where plankton productivity is high. Higher values in surface waters are often recorded in shallow, coastal areas where there is some disturbance of the seabed, or where convection currents and up-welling bring deeper waters to the surface. Inputs from land run-off and rivers may also increase nutrient levels in coastal area.

Nitrogen is one of the most important, growth-limiting nutrients in the coastal marine environment, and has a major influence on phytoplankton productivity. It is present in seawater in various combined forms, the most common being ammonia, nitrate and nitrite. Nitrogen concentrations in the surface waters are usually low during the summer when plankton productivity is high. Concentrations are often high in deep waters and where currents bring these waters to the surface. Anthropogenic inputs, such as fertiliser run-off from surrounding farmland, or nutrients from aquaculture, can lead to significant increases in nitrogen in surface waters, increasing phytoplankton productivity. This in turn can lead to the development of algal blooms and associated low water quality. It is therefore important to monitor levels of nitrogenous inputs to receiving water, and their breakdown products, some of which may be toxic.

The major source of nitrogen from fish farms is ammonia. This is broken down to nitrite, and then to nitrate, by bacterial action. Although the intermediate breakdown product, nitrite, is highly toxic. It is a transitory chemical that is usually quickly converted to nitrate. Like nitrogen, phosphorus is also an important nutrient for phytoplankton production in marine environments, although it is not normally limiting. However, it is important to monitor levels, especially in enclosed systems, where it can reach high levels and enhance phytoplankton growth. Assessment of chlorophyll 'a' levels in a water body can be correlated with algal productivity, and may help identify high levels of growth associated with increased nutrient availability. However, algal population densities fluctuate naturally due to a variety of factors other than nutrient availability (e.g. temperature, light availability and predation) so should be interpreted alongside aqueous nutrient data.

3.2.1 Methods

The exact sampling methodology has changed little over the years since the IoA has been collecting water quality data in Mulroy Bay. Analytical procedures have generally remained the same, although sampling positions in the water column appeared to have varied in the earlier surveys (i.e. surface, mid water and deep waters were sampled after 1990, while only surface and deep waters were sampled with any consistency before this date).

Samples for nitrate, nitrite, total ammonia, dissolved reactive phosphorus (DRP) and chlorophyll ‘a’ were collected from the surface, mid water and near the seabed during each annual survey, using a Van Dorn water sampler (only surface and deep-water data is available for earlier sample surveys). These were returned to the shore where they were filtered, the pH measured by meter, and then deep-frozen for transportation back to the Institute of Aquaculture for analysis. Temperature, salinity and dissolved oxygen were measured *in situ* at each station using submersible probes and meters. Transparency (in meters) at each sample station was measured using a Secchi Disc.

Total ammonia content (NH_3/NH_4^+) of the samples taken with the Van Dorn sampler was determined using the method described by Stickland and Parsons (1972). Nitrate (NO_3) and Nitrite (NO_2) were measured on a Technicon Autoanalyser, and dissolved reactive phosphorus (DRP) measured using the method described by Golterman *et al*. (1978). Chlorophyll ‘a’ was measured using a spectrophotometer after being extracted in neutralised acetone (Parsons *et al*., 1984).

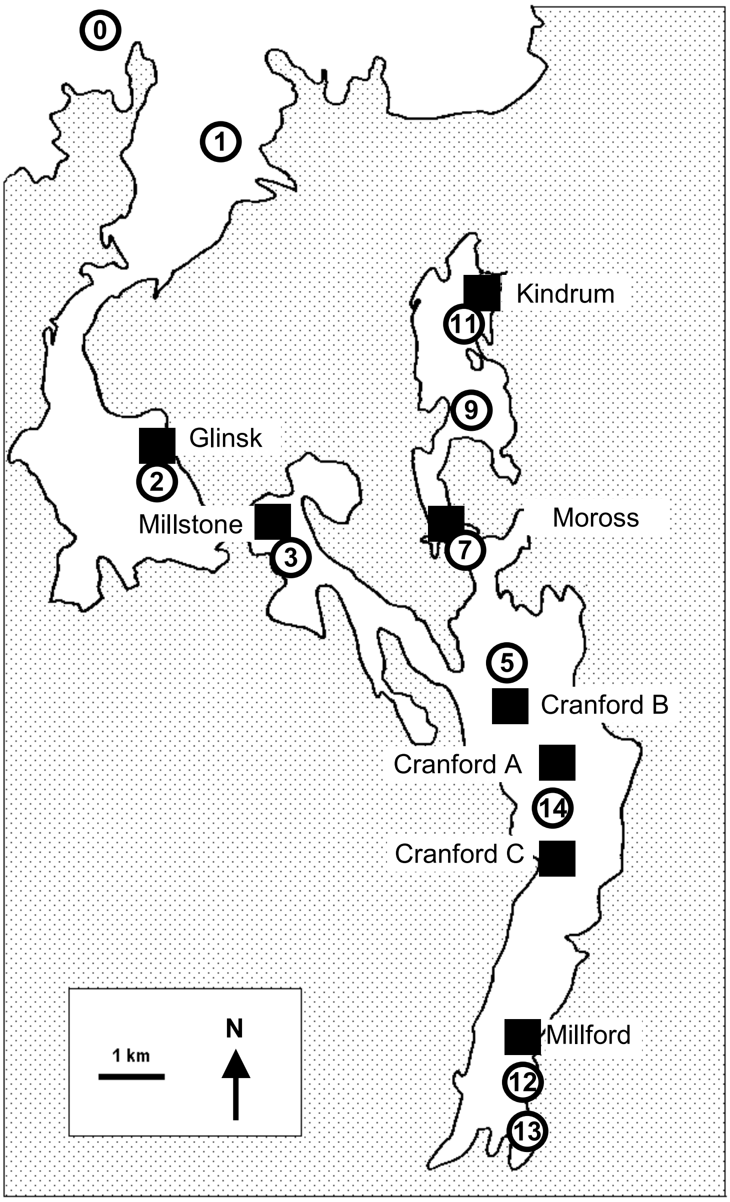

Figure 3.1: Water sampling stations (circles) used by the Institute of Aquaculture in Mulroy Bay between 1986 and 2000. Salmon cages sites are denoted by solid squares.

3.2.2 Results and Discussion

For comparative purposes, and to take in to account inconsistencies in earlier data sets collected by the Institute of Aquaculture, only the results of surface and deep water data collection are presented and commented on in the following section. However, full data sets for each water quality sampling station can be found in Appendix 1. In addition to data collected by the Institute of Aquaculture, total chlorophyll and nitrate concentrations recorded monthly by Hydro Seafood Fanad between 1997 and 2001 have been presented and commented upon.

Sampling Station	Water depth (m)	Longitude	Latitude
0	26	55^0 15.774' N	7^0 47.163' W
1	14	55^0 14.979' N	7^0 46.581' W
2	14	55^0 12.020' N	7^0 46.826' W
3	9	55^0 11.383' N	7^0 45.183' W
5	20	55^0 10.184' N	7^0 41.163' W
7	17	55^0 11.350' N	7^0 42.656' W
9	34	55^0 12.491' N	7^0 42.488' W
11	29	55^0 13.454' N	7^0 42.597' W
12	11	55^0 07.065' N	7^0 41.339' W
13	6	55^0 06.642' N,	7^0 41.903' W
14	17	55^0 09.273' N	7^0 41.519' W

Table 3.1: Longitude and latitude locations of water sampling stations in Mulroy Bay, July 2000.

Temperature

Water temperatures in the surface and deep waters at each sample station in Mulroy Bay between 1986 and 2000 are given in Figures 3.2 and 3.3 (no data was available for 1988). Trends in temperatures were found to be similar for both surface and deep waters, though more variation between stations was noted near to the seabed, likely due to the differences in water depth (see Table 3.1). As expected, water temperatures were found to be consistently lowest at the two outermost samples stations in Mulroy Bay, located near to its mouth (sites 1 and 2), while the innermost stations (sites 11, 12 and 13) were consistently warmer. Similarly, temperatures near the seabed were generally lower than those in the surface waters, as is usually the case. Evidence of some temperature stratification was observed in data collected throughout the water column in Northwater (stations 9 and 11), in the summers between 1996 and 2000. Stratification has also been observed in Northwater during other studies in Mulroy Bay during the summer months (C-Mar, 2000). This is likely to have been due to the deep, basin like nature of the area, leading to poor water flushing at depth at these sample locations. Occasional stratification was also observed at station 13 (South of Millford)

Temperatures at each sample station varied between sample years, likely as a result of different meteorological conditions in the months prior to the sampling periods. Warmest temperatures in the Bay occurred in July 1991, 1996 and 2000, while lowest temperatures were recorded in 1987, 1993 and 1998.

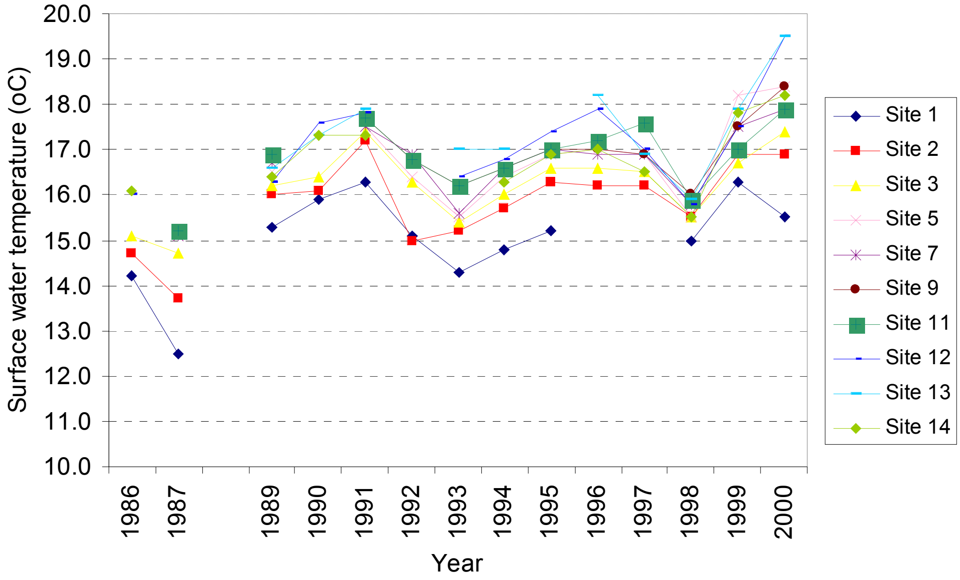

Figure 3.2: Surface water temperatures in Mulroy Bay between 1986 and 2000.

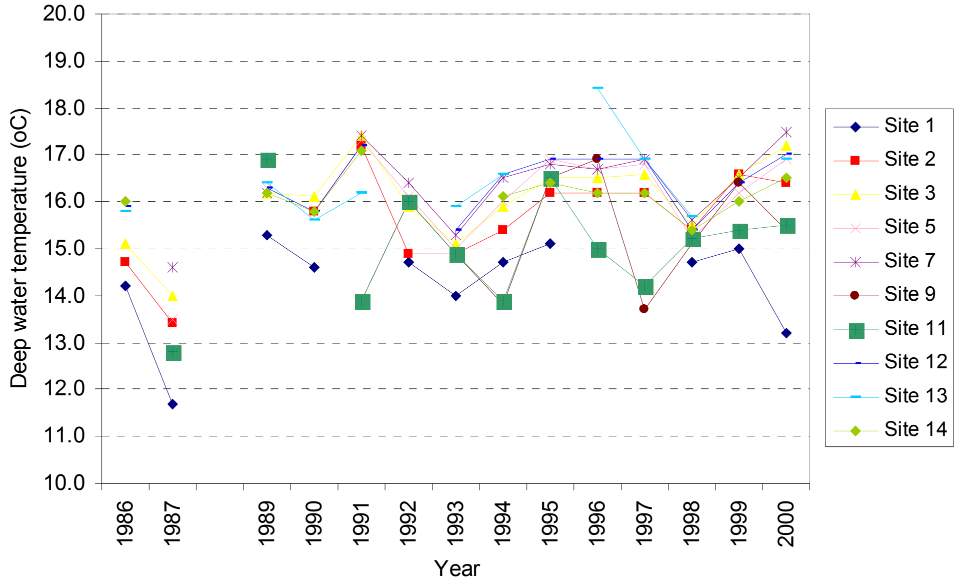

Figure 3.3: Deep water temperatures in Mulroy Bay between 1986 and 2000.

Dissolved oxygen

Dissolved oxygen concentrations at each sample station in July 1986 to July 2000 (with the exception of 1988), are presented in Figures 3.4 (surface waters) and 3.5 (deep waters). As with water temperatures, trends in D.O. were found to be similar at all stations throughout Mulroy Bay, with peaks and troughs in concentrations occurring in the same years (with occasional exceptions). As expected, D.O. concentrations were lower in deeper water than near the surface, and variability between stations was highest in deeper water, likely due to differing sample depths. Evidence of some stratification was observed in the Northwater area (stations 9 and 11), especially in 1991, 1995-1998 and 2000. This was thought to be due to the fact that the narrow, shallow channel at the mouth of the Northwater area reduced the amount of water exchange in its deeper waters, thus causing oxygen depletion near the seabed. The same phenomenon has been observed in fjordic systems, which have similar hydrodynamic and geomorphological features, (Skjoldal *et al.*, 1995). Stratification was also an occasional feature at station 13.

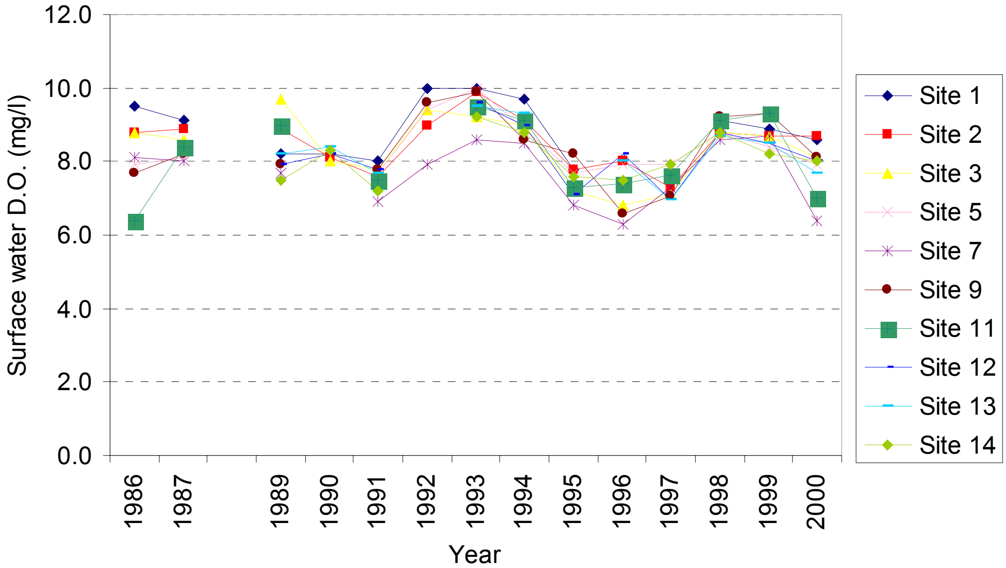

Figure 3.4: Surface water D.O. values in Mulroy Bay between 1986 and 2000.

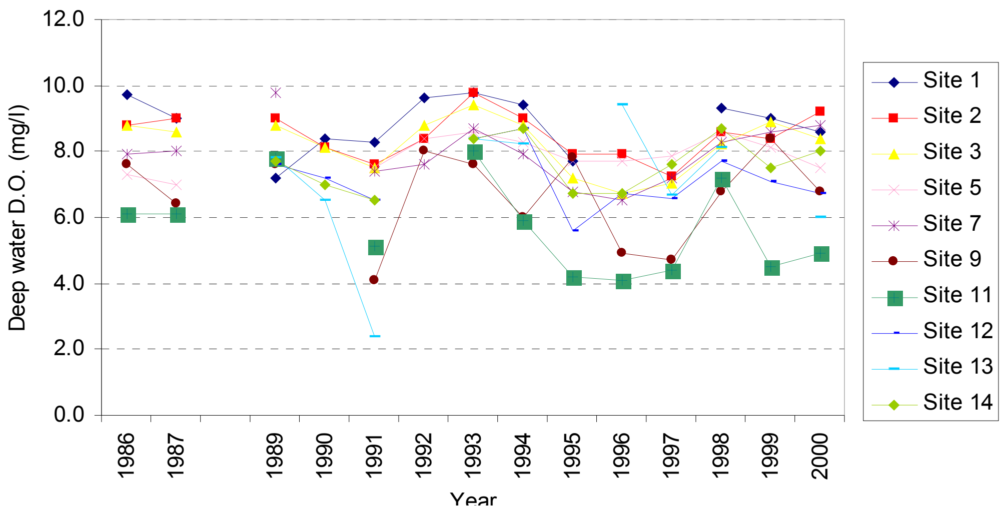

Figure 3.5: Deep water D.O. values in Mulroy Bay between 1986 and 2000.

As expected, D.O. concentrations were found to be negatively linked to trends in water temperature, with lowest D.O. levels occurring during years when water temperatures were highest (namely 1991, 1995-97 and 2000).

Data collected indicated that little overall variation had occurred in D.O. between 1986 and 2000, with levels varying annually at the surface between a minimum of 6.3 mg / l (station 7, 1996) and 10 mg / l (stations 1 and 3, 1993). Deep water D.O. varied between a minimum of 2.4 mg/l at station 13 in 1991, to a maximum of 9.8 mg/l at stations 1 and 2 in 1993.

pH

pH values at the surface and near the seabed at each sample station between 1986 and 2000 are shown in Figures 3.6 and 3.7. Generally, pH values were consistent between the surface and seabed at each sample station, with little overall variation between years. Highest surface pH values occurred at all stations in 1993 and 1994 (when values ranged from 8.33 to 8.53 at the surface, and from 8.31 to 8.52 at the seabed). A general trend of increasing pH values was observed between 1986 and 1993, when pH values rose from an average of 7.97 to 8.42. pH values then fell between 1994 and 2000, reaching their lowest point since recordings began in 2000 (pH average of 7.69). The reasons for this are unclear, though may be indicative of slight changes in water chemistry in the bay over time.

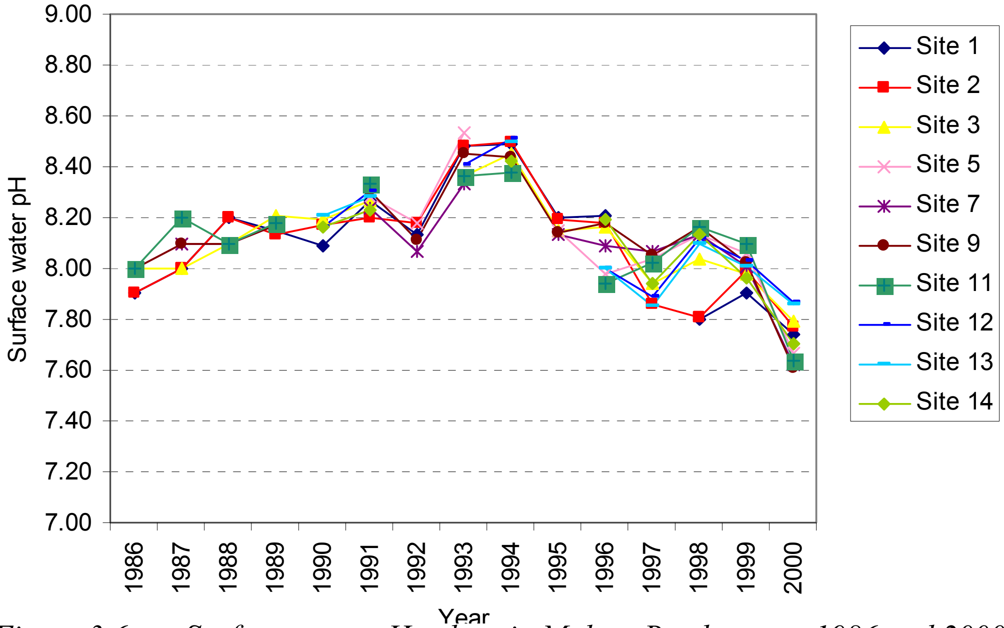

Figure 3.6: Surface water pH values in Mulroy Bay between 1986 and 2000.

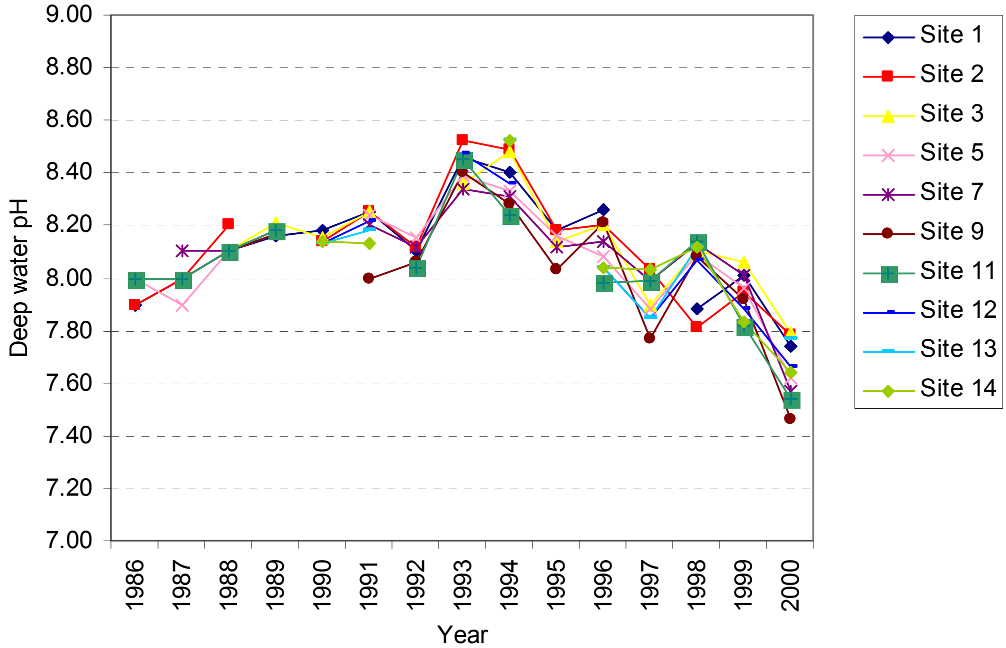

Figure 3.7: Deep water pH values in Mulroy Bay between 1986 and 2000.

Salinity

Salinity values at the surface and seabed in Mulroy Bay between 1986 and 2000 are given in Figures 3.8 and 3.9. Values were very similar at the surface and seabed at each sample station, with little variation occurring between sample years. Values generally ranged between 31 and 36 ppt, though a high value of 40 ppt was recorded at station 1 in 1989 (possibly due to meter error).
Within survey years, a trend of decreasing salinity was noted with increasing distance from the mouth of Mulroy Bay, likely due to increasing influence of freshwater run-off or inputs at the innermost parts of the bay. Salinity values at the outermost stations (1 and 2) were generally the highest in each survey year, while those at stations 11, 12 and 13 were lowest.

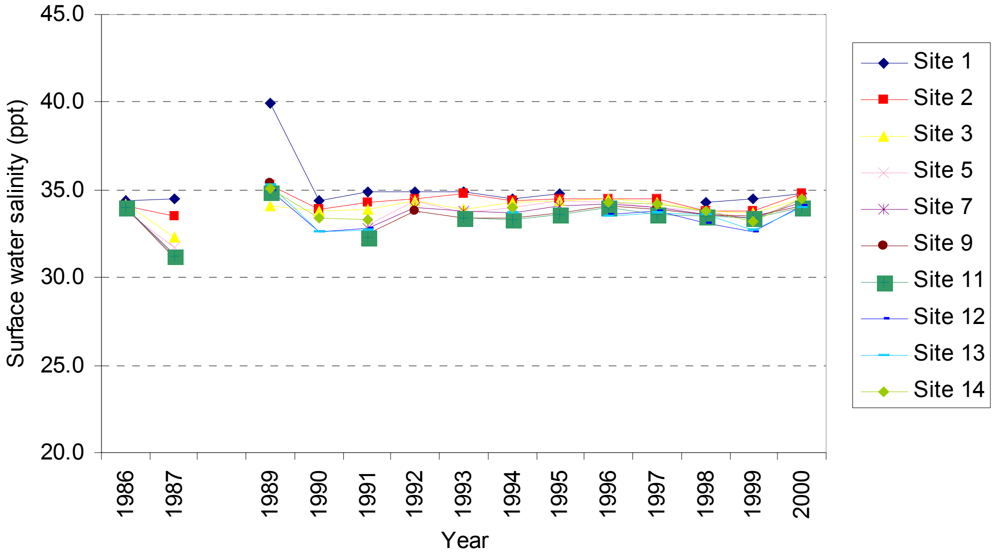

Figure 3.8: Surface water salinity values in Mulroy Bay between 1986 and 2000.

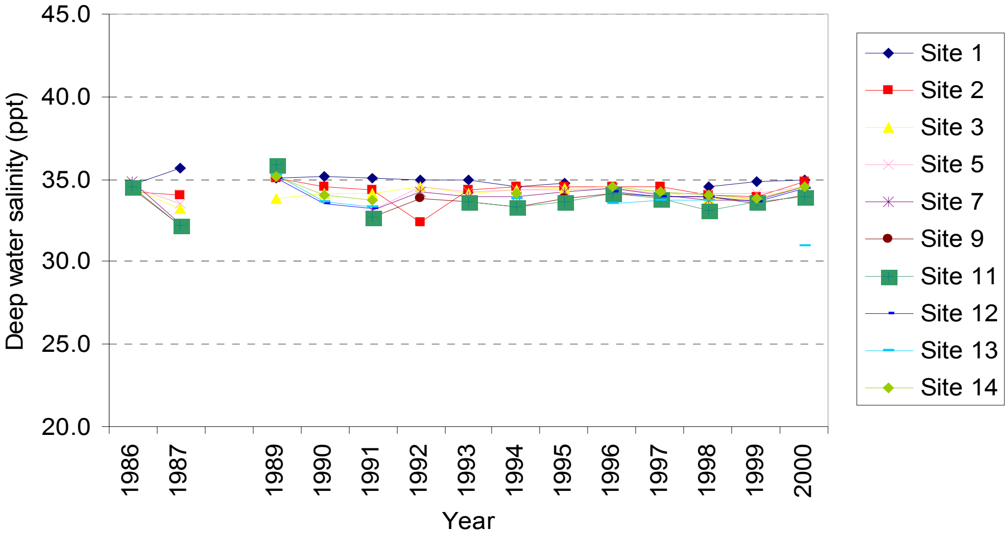

Figure 3.9: Deep water salinity values in Mulroy Bay between 1986 and 2000.

Nutrient levels

Ammonia concentrations in the surface and deep waters of Mulroy Bay have been monitored by the IoA annually between 1986 and 2000 (Figures 3.10 and 3.11). Concentrations were found to vary considerably between sample years, and sample stations within each sample year. Seabed and surface concentrations were also found to vary within sample stations, making overall trends difficult to predict. Ammonia concentrations at the surface were found to vary consistently between <0.1 and 30 µg l^{-1} in the surveys conducted between 1986 and 1997. However, higher concentrations were noted at stations 9 (Northwater) and 14 (Cranford) in 1998, with a peak of 109.6 µg l^{-1} occurring at the latter. This may have been due to a sampling event coinciding with post-feeding excretion by fish at the near-by Cranford cage sites. Despite this increase, concentrations of ammonia at stations 14 and 9 fell to pre-peak levels in 1999 and were lower still in 2000. Comparison of average ammonia values at the seabed and surface in 1986 and 2000 (8.8 and 7.3 µg l^{-1}) suggested no overall increase in concentrations had occurred in the water column between 1986 and 2000.

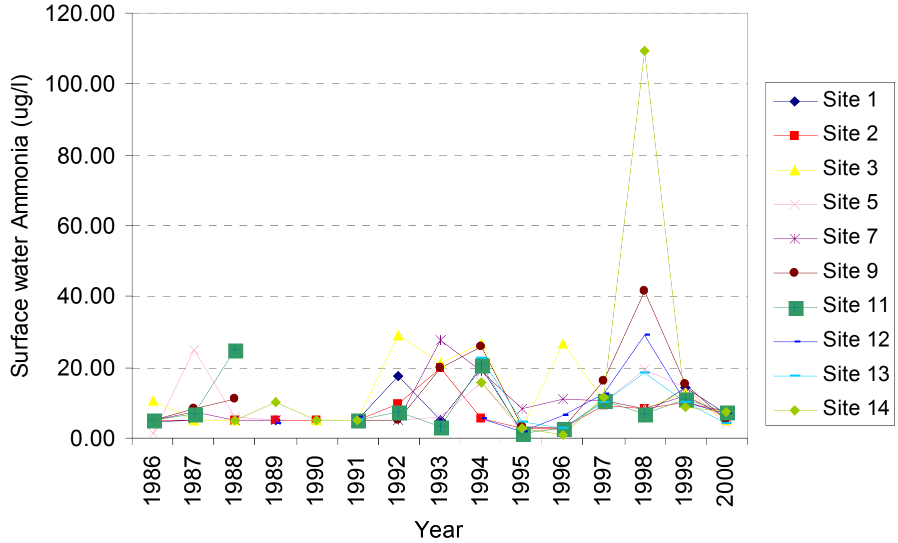

Figure 3.10: Surface water ammonia values in Mulroy Bay between 1986 and 2000.

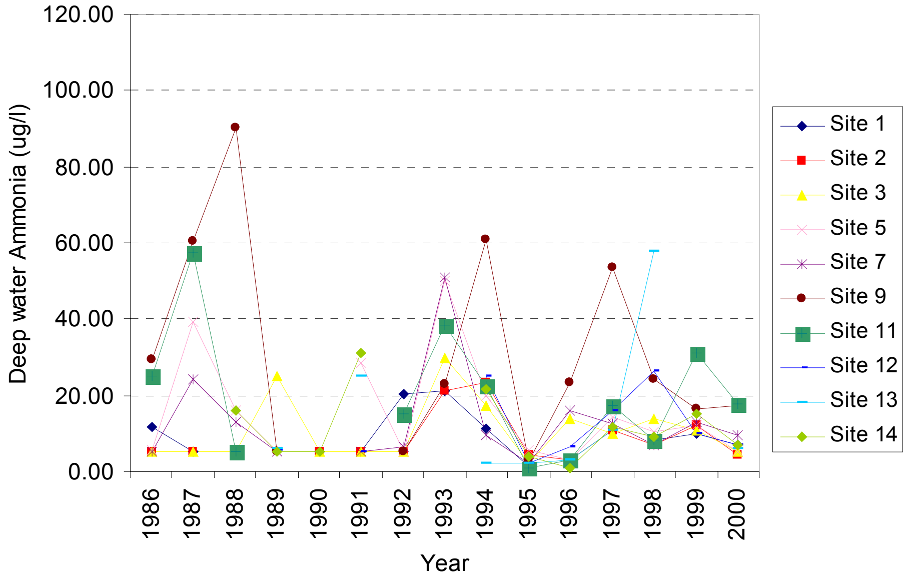

Figure 3.11: Deep water ammonia values in Mulroy Bay between 1986 and 2000.

Nitrate concentrations in the water column throughout Mulroy Bay were measured by the Institute of Aquaculture in annual surveys between 1986 and 2000, the results of which are presented in Figures 3.12 (surface waters) and 3.13 (deep waters). Trends were found to be very similar in both surface and deep waters, with large variations in concentrations being recorded between sampling stations and between sampling years from 1986 to 1993, and from 1998 to 2000.

Peaks in surface water nitrate concentrations occurred at station 7 in 1987 (44.8 $\mu g\ l^{-1}$), station 3 in 1989 (64 $\mu g\ l^{-1}$), and stations 7, 9, 13, 14, 11 and 12 in 1999 (highest concentration of 79 $\mu g\ l^{-1}$ at station 12). Deep water nitrate peaks occurred in 1989 (61 $\mu g\ l^{-1}$ at station 9), 1993-94 (50.1 $\mu g\ l^{-1}$ and 51.8 $\mu g\ l^{-1}$ station 3 respectively), 1997 (37.9 $\mu g\ l^{-1}$ at station 9) and as at the surface, stations 7-14 in 1999 (highest concentration of 98.8 $\mu g\ l^{-1}$ at station 7). The sporadic peaks observed in nitrate concentrations between 1986 and 1997 may have occurred due to occasional poor water flushing near to salmon cages within Mulroy Bay, or localised inputs from external sources (e.g. land run-off). However, the more widespread increases in nitrate levels observed at many stations in 1999 are likely to have coincided with high inputs of nitrate to the Broadwater and Northwater areas. The source of this nitrate peak is unknown, though could have been due to either a build up of waste products from aquaculture, or inputs from other sources such as local freshwater inflows or the local sewage treatment plant at Kerrykeel. More accurate predictions would require detailed knowledge of other activities / discharges in to Mulroy Bay at the time of sampling.

Although nitrate levels were elevated at most stations in the summer of 1999, levels were found to be very low in the summer of 2000, suggesting temporary peaks observed in the previous year. As with ammonia concentrations (Figures 3.12 and 3.13), nitrate levels showed no overall trends in terms of increasing or decreasing over the 14 years. Despite a large amount of variability between sample years and sample sites, nitrate concentrations were generally quite low in Mulroy Bay. Values presented by C-Mar (2000) showed nitrate levels tended to fluctuate throughout the year, and between sample years, with levels often being highest during the winter when phytoplankton activity was low, and uptake was decreased.

Nitrate concentrations recorded monthly by Hydro Seafood Fanad at each cage site in Mulroy Bay since 1997 are presented in Figure 3.14. The mean nitrate levels for the cage sites in Mulroy Bay are given in Figure 3.15. These figures both show that, as for the Institute of Aquaculture's data, nitrate concentrations between cage sites and over time can be highly variable. However, between 1997 and 1998, trends observed at each cage site were found to be very similar, with a peak in nitrate levels occurring throughout the Bay in the winter of 1998. Cages were fallowed over the autumn and early winter of 1999, after which nitrate levels were observed to have decreased compared to those recorded in the previous winter. Sporadic peaks were later observed at Moross in January 2000, and at Millstone in the following summer. Concentrations at Glinsk, however, remained very low over this period. Mean nitrate levels throughout the cage sites (Figure 3.15) show that despite large fluctuations in 1997 and 1998, concentrations were fairly consistent between 1999 and 2001, with values being within the ranges observed by the Institute of Aquaculture during their annual monitoring surveys.

Seasonal fluctuations in nitrate levels are common in aquatic systems, with concentrations usually being highest over the winter period, when algal biomass is low and uptake by phytoplankton is reduced, and lowest over the summer when algal productivity and nutrient uptake is increased due to higher light intensity and warmer water temperatures. Seasonal variation may therefore account for the peaks in nitrate concentrations observed at the cage sites in the winters of 1997 and 1998 (Figures 3.14 and 3.15). The data provided by Hydro Seafood Fanad suggests little in the way of a trend in nitrate levels at the cage sites between 1997 and 2001, other than that due to seasonality.

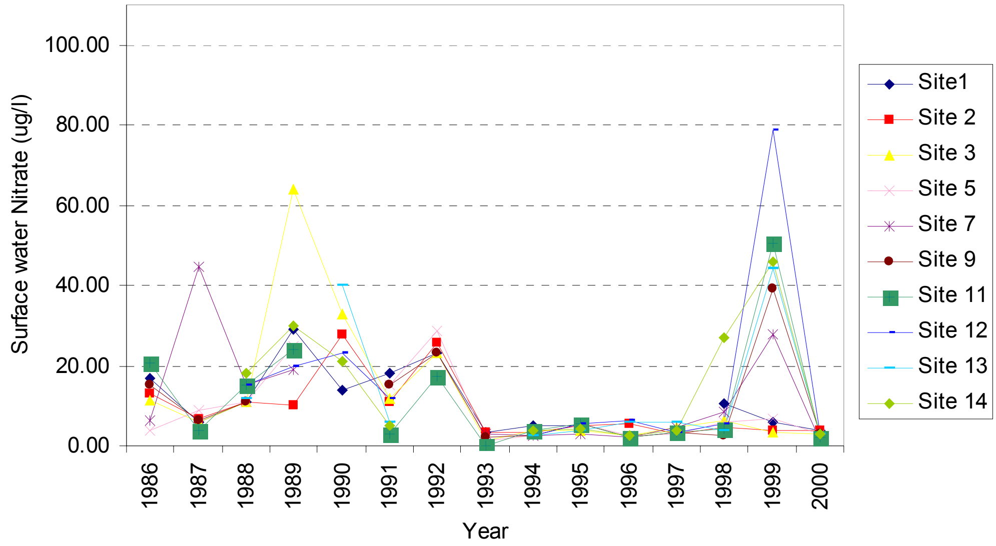

Figure 3.12: Surface water nitrate values in Mulroy Bay between 1986 and 2000.

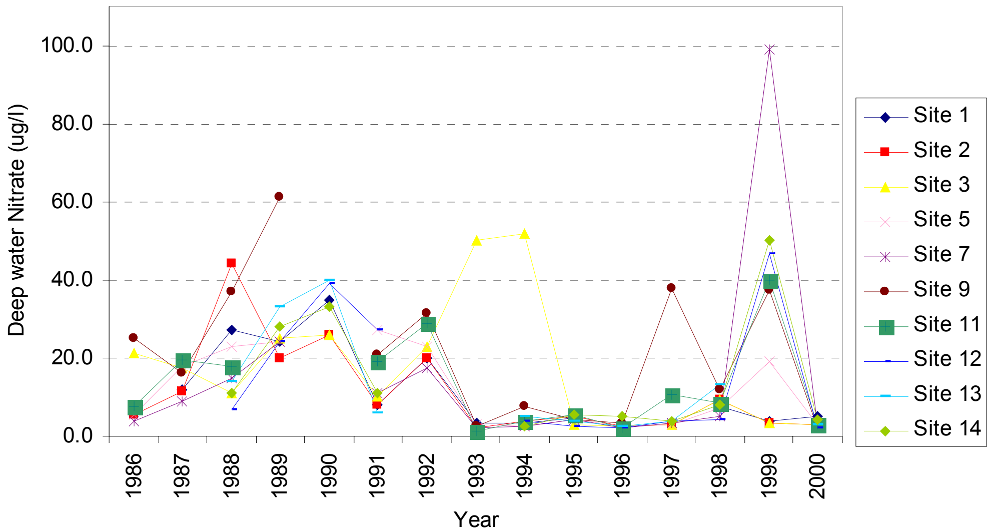

Figure 3.13: Deep water nitrate values in Mulroy Bay between 1986 and 2000.

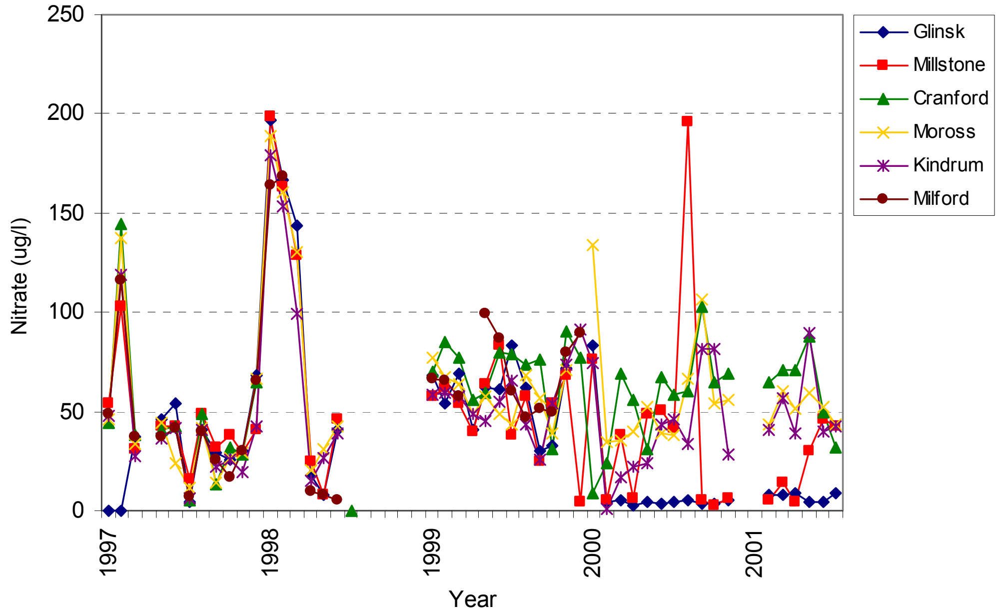

Figure 3.14: Surface water nitrate concentrations for each cage site in Mulroy Bay between 1997 and 2001, provided by Hydro Seafood Fanad.

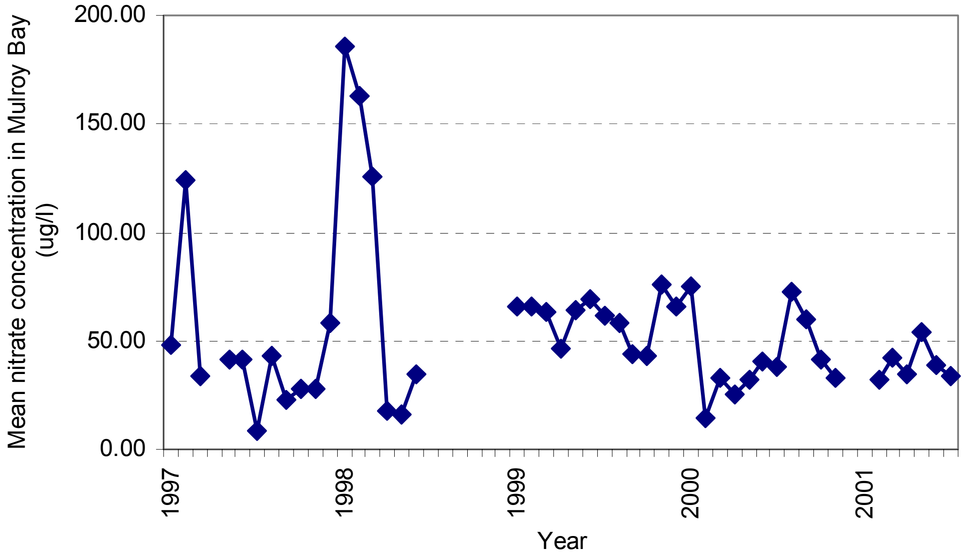

Figure 3.15: Mean surface water nitrate concentrations at the cage sites in Mulroy Bay between 1997 and 2001.

Nitrite concentrations measured by the Institute of Aquaculture at the surface and near the seabed at each sample station between July 1986 and July 2000 are given in Figures 3.16 and 3.17. Concentrations were generally low in surface waters throughout this period, with levels varying between <1.0 µg l^{-1} in 1995 (stations 7-12 and 14) and 11.3 µg l^{-1} in 1998 (station 14). Levels rose between 1997 and 1999 compared with previous years, but fell again to below 5 µg l^{-1} at all stations in 2000.

Seabed nitrite concentrations (Figure 3.17) were found to vary in line with those at the surface, though occasional peaks occurred at stations 5 (12.7 µg l^{-1}), 12 (11.3 µg l^{-1}), 14 (35.8 µg l^{-1}) and 9 (17.7 µg l^{-1}) in 1990, 1991, 1994 and 1997 respectively compared with levels at other sample stations in the same survey years. This may have been due to occasional poor water flushing near the seabed at these sample sites, allowing the temporary build-up of nitrogenous waste products to occur. In all cases, levels had returned to those found at other sample stations (generally below 10 µg l^{-1}) the year after the peak had occurred. As with ammonia and nitrate concentrations, no overall trend of increasing nitrite levels with time was apparent from the data collected, with the average nitrite value in 1986 (3.87 µg l^{-1}) being similar to that in 2000 (3.54 µg l^{-1}).

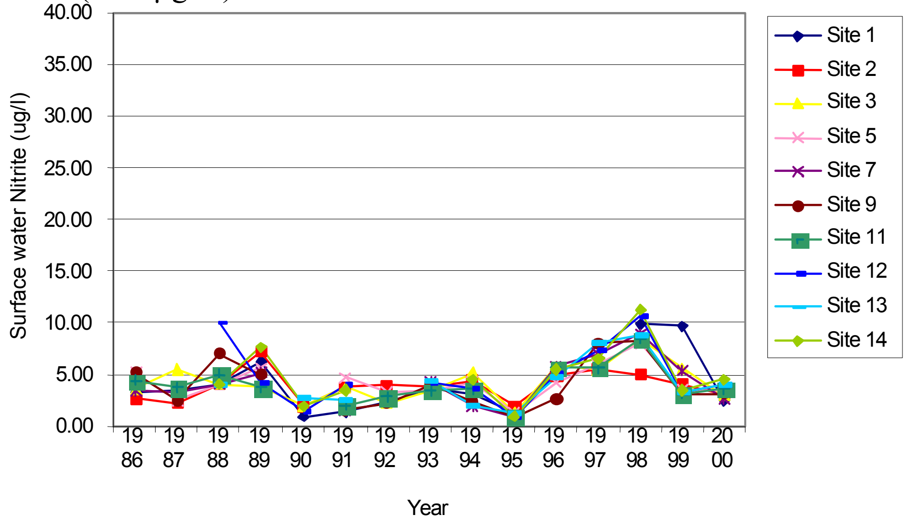

Figure 3.16: Surface water nitrite values in Mulroy Bay between 1986 and 2000.

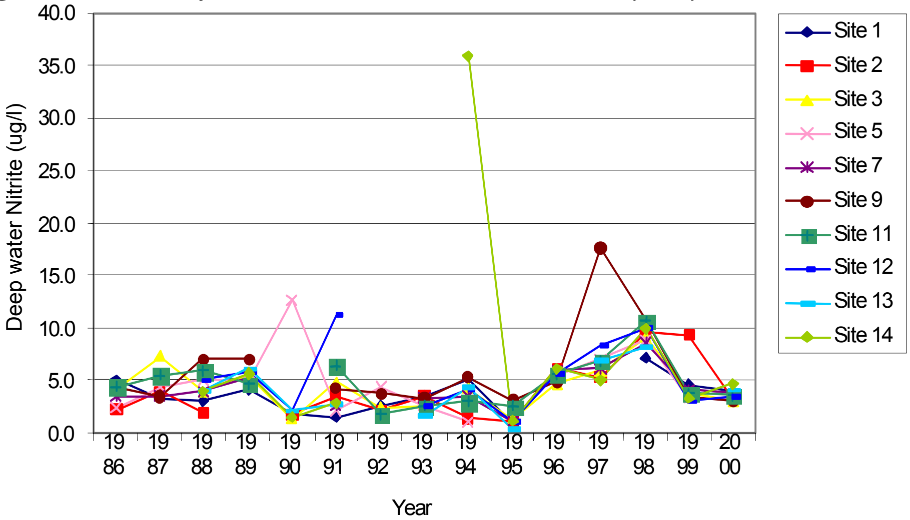

Figure 3.17: Deep water nitrite values in Mulroy Bay between 1986 and 2000.

Dissolved reactive phosphorus (DRP) concentrations at the surface and seabed in Mulroy Bay between 1986 and 2000 are presented in Figures 3.18 and 3.19 (no data was available for 1990). These varied considerably between both sample station and sample years, notably at the seabed. Surface concentrations were found to range mostly between 0 and 20 µg l^{-1} between 1986 and 2000, though tended to be higher during earlier sample years (1886 to 1989) than later years. However, in 2000, surface DRP concentrations rose at most stations to higher levels than in the previous 5 years. Peaks in surface DRP levels occurred at all stations within the enclosed areas of the bay (stations 5 to 14) in 1989 (maximum of 29.9 µg l^{-1} for station 13), with a peak (17.9 µg l^{-1}) also occurring at station 9 in 1987. Concentrations at station 9 (Northwater) were frequently among the highest recorded in any given sample year.

Seabed DRP concentrations were found to follow similar trends to those at the surface, though the levels recorded were slightly higher. This was particularly noticeable at station 9, where similar to the surface, concentrations were frequently observed to be the highest of those measured. Levels at station 9 peaked in 1988 (30 µg l^{-1}), 1994 (20.59 µg l^{-1}), 1997 (25.7 µg l^{-1}) and 2000 (23.5 µg l^{-1}). Peaks were also observed at stations 12 (21.7 µg l^{-1}) and 13 (24.6 µg l^{-1}) in 1989, as found in the surface waters. Levels at the three outermost stations (stations 1, 2 and 3) were low in all years between 1991 and 2000 at both the surface and seabed, likely due to higher water flows and flushing rates near to the mouth of the bay.

Overall, DRP levels rose and fell between sample years, showing little signs of an overall trend. Large fluctuations in phosphate concentrations were also noted in a report by C-Mar (2000) both throughout the year, and between different survey years. However, levels in 2000 were higher than in the previous few years, with averages for both surface and seabed concentrations being higher than those recorded in 1986 (averages of 6.47 µg l^{-1} (surface) and 6.58 µg l^{-1} (seabed) in 1986 compared with 10.1 µg l^{-1} (surface) and 13.3 µg l^{-1} (seabed) in 2000). Data collected in the following years will show whether this occurrence is evidence of an overall increase or a temporary fluctuation.

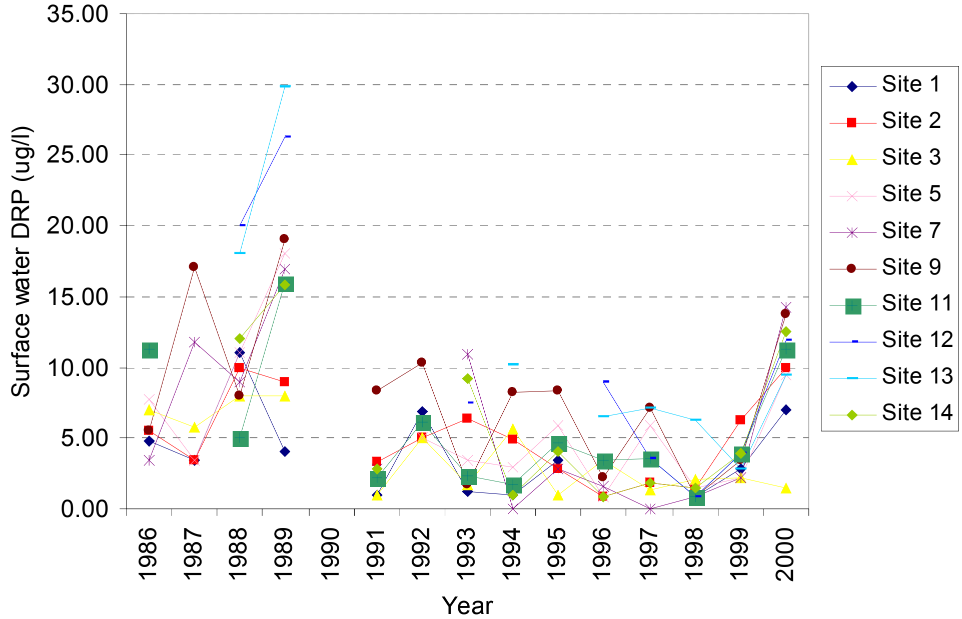

Figure 3.18: Surface water DRP values in Mulroy Bay between 1986 and 2000.

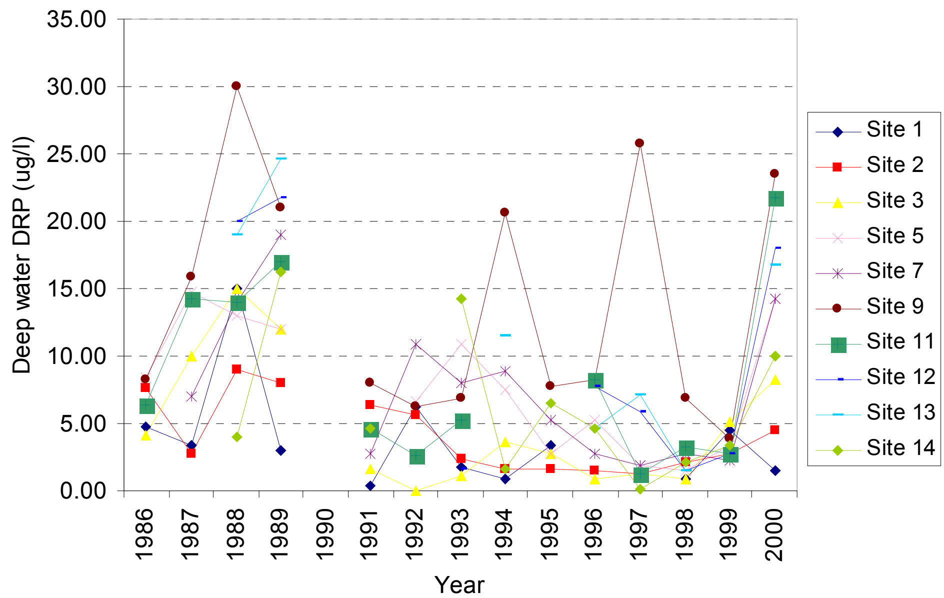

Figure 3.19: Deep water DRP values in Mulroy Bay between 1986 and 2000.

Chlorophyll 'a'

Chlorophyll 'a' concentrations measured at the surface and near to the seabed at each sample station in Mulroy Bay between 1986 and 2000 are given in Figures 3.20 and 3.21 respectively. Concentrations at the surface (Figure 3.18) were found to vary between sample stations and years, though were generally low in all cases. Seabed concentrations (Figure 3.19) were found to be more variable between stations and years, and this likely due to the differences in depths at which samples were obtained. A peak was observed in 1990 at station 13 (deep water) though this proved to be temporary, with levels falling in the following year. Similar overall trends were observed at the surface and seabed across the years, with a slight decrease in mean chlorophyll 'a' occurring between 1986 and 1988 (mean concentrations of 2.76 $\mu g\ l^{-1}$ in '86 and 0.81 $\mu g\ l^{-1}$ in '88), and levels rising slowly between 1988 and 1999 to peak at and average of 3.12 $\mu g\ l^{-1}$ in 1999 (individual peaks of 8.2 $\mu g\ l^{-1}$ and 6.5 $\mu g\ l^{-1}$ were observed in deep waters at stations 9 and 12 respectively in 1999). Increases were most apparent in Northwater and Broadwater areas during this period. However, overall concentrations fell in 2000, with an overall mean concentration of 1.68 $\mu g\ l^{-1}$ being observed.

Trends in chlorophyll 'a' concentrations did not closely follow trends in either aqueous nitrogenous nutrients or DRP concentrations, though it should be noted that the highest average chlorophyll 'a' concentrations occurred in 1999, coinciding with elevated levels of nitrate. Elevated chlorophyll 'a' concentrations were observed between 1994 and 1999, when nitrate concentrations were relatively low, suggesting other non-nutrient related factors are also likely to have had an important influence on phytoplankton growth. However, data of this nature should not be over interpreted, due to the fact that a once yearly sampling event (on which comments are based) may not encompass natural fluctuations in biomass that regularly occur in phytoplankton communities.

Data from 1994 to 1998 collected by C-Mar (2000) (supplied by Hydro Seafood Fanad) suggested that chlorophyll 'a' concentrations fluctuated throughout the year, though maximum concentrations appeared to decrease over this period. This has been attributed to increases in shellfish production in the bay, meaning that grazing rates by consumer organisms would have increased.

Total chlorophyll data supplied by Hydro Seafood Fanad in surface waters (3m depth) at each cage site in Mulroy Bay between 1997 and 2001 is plotted in Figures 3.22 and 3.23 (mean values). Total chlorophyll concentrations overall were found to be low for most of the year, with only occasional peaks in levels being observed. Figure 3.21 clearly shows that large variations in chlorophyll 'a' concentrations can occur between different areas of Mulroy Bay at a given sampling period, with variations also occurring over time. Highest mean chlorophyll concentrations over the period studied were found to occur at the start of each calendar year (Figure 3.22), in February and March. This is unusual in that peak chlorophyll levels in aquatic environments would often be expected to coincided with the onset of summer, due to increases in light and temperature. The increase in biomass at the start of each year observed in Mulroy Bay may partially be attributed to reduced grazing rate by resident invertebrate populations (such as mussels) during the period of lowest annual sea temperature, though this would need to be confirmed by further studies.

Long term changes in the phytoplankton community are difficult to predict from such a limited data set. However, as concluded from chlorophyll 'a' data collected by the Institute of Aquaculture between 1986 and 2000, there is little evidence to suggest that a decline in the phytoplankton biomass gradually occurred in Mulroy Bay between 1997 and 2001. Further studies would be required to investigate the grazing relationship between the large areas of shellfish culture within the Bay and the phytoplankton community. It should be noted however, that the amount of phytoplankton recorded in Mulroy Bay at any given time will be partially dependent not only on intrinsic features and processes, but also on water flushing rates / residence times, and the importation of phytoplankton from the adjacent sea areas.

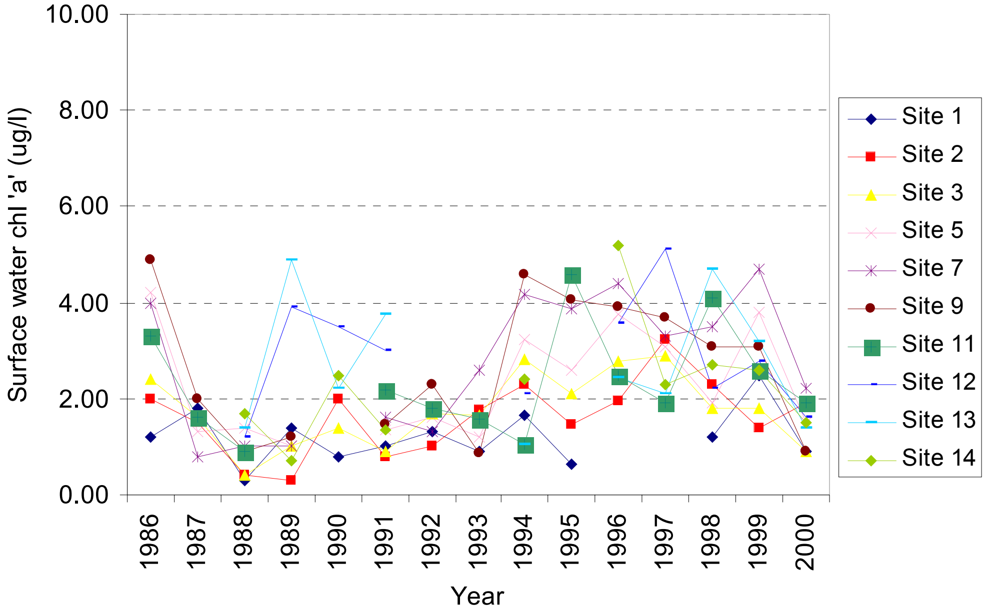

Figure 3.20: Surface water chl 'a' values in Mulroy Bay between 1986 and 2000.

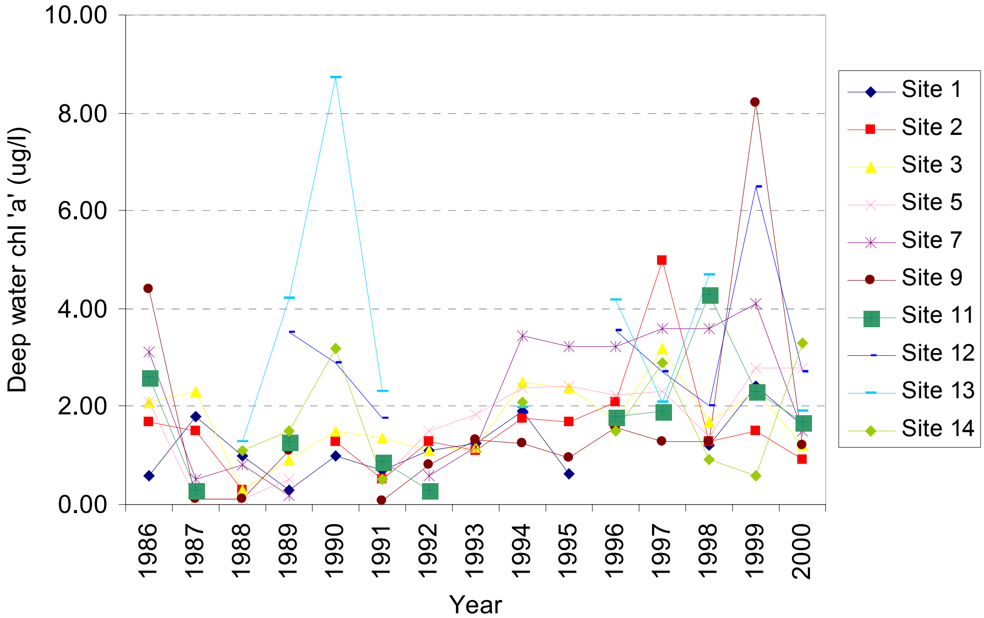

Figure 3.21: Deep water chl 'a' values in Mulroy Bay between 1986 and 2000.

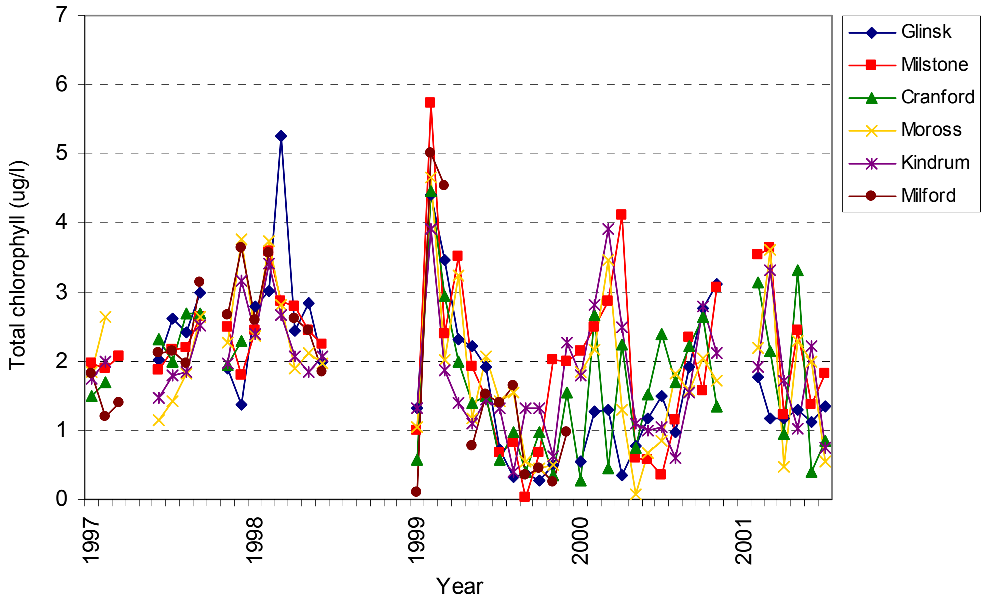

Figure 3.22: Total chlorophyll concentrations at each cage site in Mulroy Bay between 1997 and 2001.

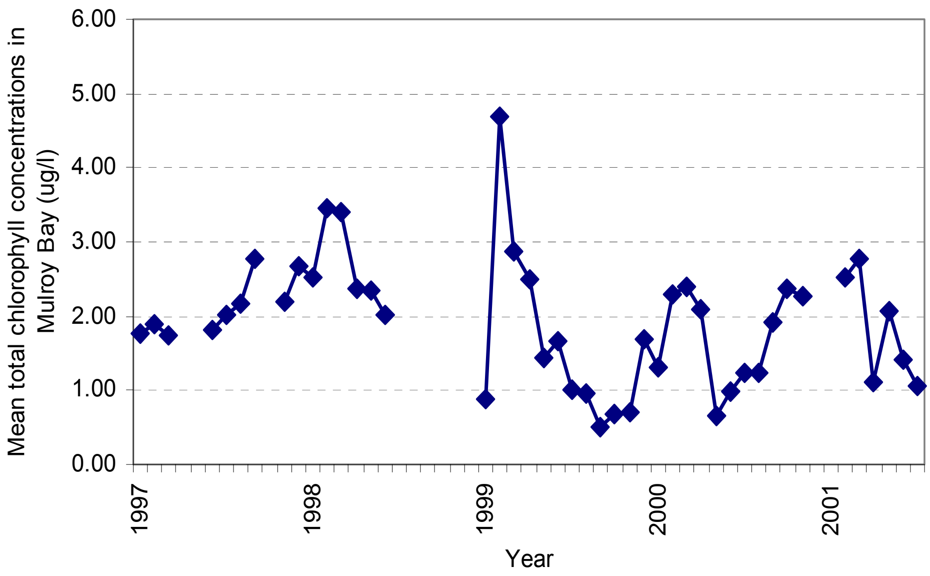

Figure 3.23: Mean total chlorophyll concentrations at each cage site in Mulroy Bay between 1997 and 2001.

Year	Area	NO_3 – N	NH_3/NH_4^+ - N	Chl 'a'
1991	Control	7.3 (6.0 – 8.0)	<5 (<5)	0.9 (0.7 - 1.0)
	Entrance	10.3 (7.0 –18.0)	<5 (<5)	0.9 (0.5 - 1.4)
	Northwater	12.0 (3.0 – 32.0)	16.4 (<5 - 92.2)	1.1 (0.1 - 2.2)
	Broadwater	14.6 (6.0 – 34.0)	18.0 (<5 - 85.0)	1.2 (0.5 - 1.6)
1992	Control	25.9 (23.0 – 28.8)	8.7 (<5 - 16.0)	2.9 (2.1 - 3.9)
	Entrance	22.4 (17.3 – 25.9)	11.4 (<5 - 29.1)	1.3 (1.0 - 2.1)
	Northwater	24.2 (17.3 – 48.8)	13.3 (<5 – 40.5)	1.4 (0.3 - 2.3)
	Broadwater	26.2 (23.0 – 34.5)	7.9 (<5 - 33.5)	1.7 (1.4 - 2.3)
1993	Control	4.1 (3.8 – 5.1)	25.2 (22.1 – 29.9)	1.6 (0.6 - 2.7)
	Entrance	8.4 (1.2 – 50.1)	21.8 (5.0 – 44.7)	1.3 (0.9 - 1.8)
	Northwater	2.0 (<1 - 3.2)	25.9 (8.1 – 50.9)	1.5 (0.9 - 1.8)
	Broadwater	2.2 (<1 - 3.5)	16.7 (5.0 – 50.1)	1.8 (0.8 - 3.3)
1994	Control	4.0 (3.4 – 5.1)	11.7 (11.4 – 12.3)	0.7 (0.6 - 0.8)
	Entrance	9.4 (3.4 – 51.8)	18.8 (5.4 – 26.8)	2.2 (1.5 - 2.8)
	Northwater	3.7 (1.3 – 9.1)	29.9 (<1 – 78.9)	2.4 (0.6 - 4.6)
	Broadwater	24.1 (2.5 – 266.5)	17.4 (2.0 – 28.5)	2.5 (1.0 - 3.8)
1995	Control	5.1 (4.9 – 5.2)	2.2 (1.9 - 2.6)	0.9 (0.7 - 1.3)
	Entrance	4.5 (2.8 – 5.1)	2.6 (1.3 - 4.5)	1.4 (0.4 - 2.4)
	Northwater	4.2 (2.7 – 5.6)	6.7 (0.7 - 23.0)	3.1 (0.7 - 4.6)
	Broadwater	4.3 (2.7 – 5.6)	2.9 (1.3 - 5.8)	2.5 (1.9 - 3.7)
1996	Control	-	-	-
	Entrance	3.1 (2.6 – 5.3)	9.1 (<1.0 – 26.7)	2.2 (1.5 - 2.8)
	Northwater	3.1 (2.1 – 4.4)	16.2 (<3.0 - 33.9)	2.6 (1.3 - 4.4)
	Broadwater	3.4 (2.2 – 6.4)	3.5 (<1.0 – 16.7)	3.6 (1.5 - 5.2)
1997	Control	-	-	-
	Entrance	3.4 (3.1 – 4.6)	10.4 (9.1 – 11.4)	3.4 (2.9 – 5.0)
	Northwater	9.1 (3.2 – 37.9)	17.7 (9.6 – 53.7)	2.4 (1.3 – 3.7)
	Broadwater	4.1 (1.9 – 5.8)	12.0 (10.0 – 15.9)	2.9 (1.6 – 5.1)
1998	Control	12.8 (7.7 – 20.1)	6.9 (5.8 – 8.1)	1.3 (1.2 – 1.4)
	Entrance	7.0 (4.7 – 9.3)	9.8 (4.6 – 17.3)	1.7 (1.3 – 2.3)
	Northwater	6.1 (2.5 – 12.0)	15.9 (6.9 – 41.5)	3.6 (1.3 – 4.6)
	Broadwater	10.3 (4.0 – 29.6)	28.3 (9.2 – 109.6)	~3.5 (<1.0 – 4.7)
1999	Control	-	-	-
	Entrance	3.8 (3.4 – 5.7)	9.8 (10.0 – 14.2)	2.1 (1.4 – 2.5)
	Northwater	45.4 (6.7 – 79.0)	11.3 (8.9 – 15.3)	3.7 (0.6 – 9.6)
	Broadwater	46.2 (27.8 – 98.8)	14.9 (10.0 – 31.1)	5.2 (2.3 – 8.7)
2000	Control	7.43 (6.1 - 8.7)	5.9 (5.0 - 6.8)	1.43 (0.9 - 1.9)
	Entrance	3.67 (2.9 - 5.2)	5.81 (4.2 - 7.6)	1.01 (<1.0 - 1.9)
	Northwater	2.49 (1.4 - 2.8)	9.33 (5.0 - 17.9)	2.08 (0.9 - 3.8)
	Broadwater	2.69 (2.1 - 11.6)	4.49 (4.2 - 8.5)	1.56 (1.4 - 3.8)

Table 3.2: Comparison of mean values and ranges of nutrient and chlorophyll 'a' concentrations ($\mu g\ l^{-1}$) for areas within Mulroy Bay between 1991 and 2000 (IoA data).

3.2.3 Overview and conclusions

Comparisons of overall nutrient and chlorophyll 'a' concentrations in different spatial areas of Mulroy Bay between 1991 and 2000 are given in Table 3.2. This shows both mean values, and maximum and minimum values for each sample period, compared to a control station (located just outside of the bay).

As found in the results for individual stations presented above, mean values of nitrate, ammonia and chlorophyll 'a' fluctuated between sample years in each area of Mulroy Bay. Nitrate concentrations were found to be highest in 1992 (all stations), 1994 (Broadwater) and 1999 (Broadwater and Northwater), tying in with peaks observed in Figures 3.12 and 3.13. However, values in 2000 were low and similar to those recorded previously, showing little or no evidence that overall levels had increased since data collection began. Similarly, ammonia concentrations were found to be elevated in 1993 (all areas), 1994 (Northwater) and 1998 (Broadwater), again tying in with elevated levels shown in Figures 3.10 and 3.11. Again, values fell in 2000 to low levels, suggesting no overall increase in concentrations. Chlorophyll 'a' concentrations were low throughout the monitoring history of the bay, with only slight increases being observed at the Control site in 1992 (outside the bay), in Northwater in 1995, and most areas in 1996-97. Though these results provide no indication of an overall increase in nutrient levels within Mulroy Bay over the time studied, they do highlight the fact that concentrations may become elevated from time to time in the inner parts of the bay, i.e. Broadwater and Northwater, likely due to reduced water flushing rates compared with outer parts of the bay.

Though nutrient inputs such as nitrogen are a causal factor in the production of phytoplankton and chlorophyll, their relationship is complex depending on other nutrients (phosphorus and silica), light and temperature. It is therefore difficult to assign values of nutrients which impact on chlorophyll levels. Marine coastal waters are considered to be hypernutrified if dissolved inorganic nitrogen levels reach 168 µg/L (SEPA, 2002).

Comparison with the nitrate concentrations for coastal waters defined by the Marine Institute (1999), the levels have never exceeded those considered as having low levels (i.e. < 200 µg NO_3-N/l). This suggests that during the summer months that nitrate levels are not at problematic levels within Mulroy Bay.

3.3 Sediment Quality: Chemistry and Macrofauna

3.3.1 Introduction

Physical and chemical analyses

Several physical and chemical analyses can be used to determine the quality of sediments, including redox potential, particle size and organic fraction (measured as a function of % carbon and nitrogen). The redox potential of sediment is a quantitative measure of oxygen demand. This is influenced by the rate of oxygen diffusion between the water column and sediment (a function of pore size and oxygen concentration in the overlying water) and rate of oxygen consumption by inorganic and organic processes in the sediment (e.g. chemical reactions and microbial activity). High inputs of organic matter from waste products to a system can severely reduce oxygen concentrations. This is due to increased amounts of microbial decompositional activity, leading to anoxic conditions and reduced redox potentials (Pearson and Rosenberg, 1978). Reduced oxygen availability can disturb benthic infaunal communities, while creating favourable conditions for anaerobic sulphur-oxidising bacteria. These organisms reduce sulphur to hydrogen sulphide that is released as a gas (a process termed out-gassing) and can be potentially damaging to fish.

Organic inputs to sediments can be measured directly by calculating their carbon content. Generally a carbon content of less than 5% indicates little or no organic enrichment, while values of 5-15% suggest a certain amount of enrichment. Values greater than 15% only generally occur in areas of serious organic loading. However, these values do not account for shell matter, containing calcium carbonate, that occurs naturally in sediments. This can increase percentage carbon values. It is therefore important to make observations of the physical characteristics of sediments to compare with the results of chemical analyses. Sediment nitrogen is often used as a more accurate indicator of sediment enrichment, due to the fact that this is mostly derived from external inputs, such as cage wastes.

Perhaps a more reliable indicator of nutrient inputs to sediment is a measure of organic nitrogen content. Nitrogen levels reflect the nutrient status of sediments and unlike carbon, are not influenced by the presence of shell matter.

A measure of the sediment carbon / nitrogen ratio is used to measure waste dispersion. Values of less than 11 generally indicate poor dispersion of waste food and faeces (Gowen & Bradbury, 1987) although this can again be influenced by shell matter. Tables giving sediment carbon, nitrogen and C/N ratio values for each site are given in Appendix 2.

Biological analyses

Changes in organic input to an area can lead to variations in chemical, physical and biological sediment characteristics, which in turn have direct or indirect impacts on benthic fauna. Fluctuations in organic load are considered to be among the principal factors influencing changes in benthic faunal communities. In marine sediments, polychaetes form the dominant fauna and can be used as indicators of change. Opportunistic species such as *Capitella* sp. and *Malacoceros fuliginosus* are known indicators of organic pollution and tend to dominate communities in organically enriched, oxygen depleted sediments (Pearson and Rosenberg, 1978). In areas where highly anoxic conditions exist, benthic fauna is often absent or severely reduced, and sediments are characterised by a high abundance of sulphur reducing bacteria. These conditions are often denoted by the formation of white mats of the sulphur oxidising bacterium, *Beggiatoa*.

A pollution gradient usually occurs with increased distance from a point source of organic waste input, such as a sea cage, along which sediment enrichment decreases (Pearson and Rosenberg, 1978). Generally, three zones of enrichment are recognised along such gradients: grossly / very impacted, impacted, and normal / unimpacted (Nichols, 1977; Dauer and Conner, 1980). Pearson and Rosenberg (1978) also identified a fourth zone called the transitory zone, where mild sediment enrichment could lead to an increase in species numbers and biomass above those encountered in unpolluted sediment. The extent of organic enrichment in the marine environment can therefore be assessed using changes in benthic invertebrate community composition. This can also be tied in with physical parameters of the sediment, such as particle size, in order to gain a more complete understanding of any differences in community structure that arise between areas.

3.3.1 Methods

Sediment samples were collected by both grab sampling and core sampling at each cage site during each survey year (cage sites are given as squares in Figure 3.1). Grab samples were taken from the centre of each cage block (termed ‘mid’) and at 0m, 25m and 100m from each end along a transect that was parallel to the prevailing currents. Core samples were taken from the centre of each cage block and at 25m from each end, again on a transect parallel to the prevailing current.

Control stations were located at least 500m from each cage site where water depth was similar to that at the cage. Both grab and core samples were taken at these sites. Where possible control stations were located in the same place as those used in previous years to help determine whether natural changes in conditions had occurred, and to maintain continuity between surveys.

Redox potential and visual assessment

One core sample was taken by diver at each of the sediment sampling stations using a 70mm perspex core. The redox potential of each core sample was measured at 1cm intervals, to a sediment depth of 4cm (where possible) using a Russell CEPTR II/300 redox probe attached to a Jenway millivolt meter. The probe was calibrated against Zobell's solution. A visual assessment of the physical sediment conditions, including colour, smell and texture, within each of the core samples was also made.

Carbon / nitrogen and macrofauna

Four grab samples (three for macrofauna and one for carbon / nitrogen analysis) were taken at each sediment sampling station using a van Veen grab. This samples a seabed area of $0.025m^2$. Approximately $10cm^3$ of sediment was collected from the grab allocated for carbon / nitrogen analysis. This was placed in to a universal container and frozen to for transport to the Institute of Aquaculture, Stirling, for analysis. Samples were oven dried on arrival in the laboratory, and carbon and nitrogen contents analysed using a Perkin Elmer 2400 Series ii CHNS/O analyser.

The sediment from the three grabs taken for macrofaunal analyses were preserved in 10% formosaline solution and stored in labelled, double-wrapped plastic bags for transport back to Stirling where they were sieved through a 1mm mesh. Macrofauna were then removed from the remaining sediment using fine forceps and an illuminated magnifying lens, and placed in 70% alcohol to await identification and enumeration. Macrofaunal samples were identified using a stereo dissecting microscope with a range of 10 to 40 x magnification with a light source of adequate power to clearly see features for both incident and transmitted light. If further magnification was required (such as observation of the structure of polychaete chaetae) a compound microscope with range of 100 to 1000 x (using oil immersion) magnification was be used. Specimens were identified to the lowest taxonomic level possible (usually species) and enumerated to allow univariate and multivariate analyses to be performed. Identification of macrofauna in recent years has been consistent with the National Marine Biology Accredited Quality Control scheme, to which the Institute of Aquaculture belongs.

3.3.2 Results

The results of physical / chemical and macrofaunal analyses for each cage site in Mulroy Bay collected over the past decade by the Institute of Aquaculture are given in the following sections. Redox levels at a depth of -2cm in sediment cores are presented graphically, along with % dry weights of carbon and nitrogen in the sediments. Macrofaunal data has been summarised in the form of ranked species lists and diversity graphs. Raw data can be found in Appendices. Due to the fact that many of the cage blocks in Mulroy Bay are moved from time to time, sample stations other than the control may not be consistent between surveys. However, this will be commented upon in the text where appropriate. Results from each cage block will be dealt with on an individual basis, followed by a general summary of the findings.

Moross 1

Salmon cages have been in place at Moross since the late 1980s, for which monitoring data is available from 1988 onwards. Initially one cage was located at Moross, though up to three sites have been present since monitoring began. In recent times, Moross 1 and Moross 2 have been present on a regular basis, and will be discussed here. Since the locations of early cage sites in relation to modern ones are unclear, only data for which exact sample locations is known will be discussed.

Data of known origin is available from 1991 for Moross 1. Sediment at the cage site has traditionally been found to be composed of medium - soft, fine sand and mud, of a brown or black colour. Waste accumulation from overlying salmon cages has been frequently observed on the sediment surface, notably at mid cage and the immediate vicinity of the site. As a result of this, redox potentials (-2cm) (Figure 3.24 (a)) have consistently been low at mid cage (with the exception of 1997 due to changes in cage position).

Carbon and nitrogen contents in the sediments at Moross 1 are given in Figures 3.24 (b) and (c). Despite sediment being oxygen depleted at mid cage, and waste being observed to accumulate on the sediment surface, carbon and nitrogen levels directly under the cages were often lower than levels in the surrounding sediments. This may have been partly due to effective bioprocessing by high numbers of opportunistic polychaetes in the enriched sediments. In general, % carbon values in sediments at Moross fluctuated inconsistently between sample years, with no overall trends being discernible. The highest carbon level (6.67%) was observed at 25m N in 1997, while the lowest (2.37%) was observed at mid cage in 1999. Values generally fluctuated around 5% carbon, suggesting sediments throughout the site, including the control, were mildly enriched. This may have been partly exacerbated by slow seabed currents in the area, allowing wastes and detritus to readily settle and accumulate.

Percentage nitrogen levels at Moross again fluctuated in an unpredictable manner, with no distinct trends being observed. As for carbon, nitrogen levels at mid cage were often among the lowest recorded in each year, though levels to either side of the cage block mostly indicative of mild enrichment. The lowest nitrogen level in the sediment was observed at 25m N in 1996 (0.14%), while the highest level was observed at 100m S in 1997 (1.73%).

Macrofauna data for the Moross area dates back to 1988. Data collected during the period of 1988 to 1993 shows that sediments under the Moross cage sites have traditionally been dominated by indicator species, notably *Capitella* spp. and *Malacocerus fuliginosus*. Areas of high impact (as denoted by the presence of opportunistic species) were found mostly to have been confined to the sediments directly below and around the cage sites, with less tolerant species such as *Pseudopolydora paucibranchia* dominating mildly enriched sediments beyond the cage edges. Diversity was found to increase in all years with increasing distance from the cages, with a return to background community composition being noted between 25m and 40m from the cages in most years (IoA, unpub. data).

Macrofauna data collected in recent years at Moross 1 is summarised in the form of a table showing the top five ranked abundant species at each sample station between 1996 (Appendix 3 Table 1), and in the form of a graph of diversity (Shannon-Weiner function) at each sample station in Figure 3.24 (d).

The macrofauna at Moross 1 (1996 – 1999) was found to be typical of fine, slightly enriched sediments, with species such as *Prionospio fallax*, *Lumbrineris latreilli, Mediomastus fragilis* and Nematoda spp. being common throughout the area and at the control site in each sample year. Spionid polychaetes were among the commonest groups found in the sediments, with a variety of species being recorded.

As in earlier surveys during the late 1980s and early 1990s, enrichment indicator species, such as *Capitella* spp. and *Malacoceros fuliginosus* were found to be abundant, and in many cases dominated (accounting for over 50% of individuals) the macrofaunal communities in sediments at 0m S, mid cage, and up to 25m N of the Moross 1 cage site over the last five years. (Indicator species were absent from sediments at mid cage and 0m S in 1997, but present up to 100m N, likely due to a change in cage position prior to that survey). Enrichment indicator species dominated the macrofauna at 25m S in 1996, though this was not observed in later surveys.

Species compositions at 25m S, 100m S and 100m N of the cage site generally resembled that at the control station, suggesting that the highest enrichment effects from cage waste accumulation were confined to the immediate vicinity of the cage, and up to 25m N. (Again, results in 1997 showed impacts extended to 100m N, likely due to the change in cage position since the previous survey, so that sampling sites were located over other old sites). However, species assemblages overall, including those at the controls, reflected the fact that background sediment enrichment was a feature of the Moross area.

Species diversity values at Moross 1 (see Figure 3.24(d)) reflected the observed dominance of enrichment indicator species between 0m S and 25m N of the cage block. Diversity values were found to be low at these sample stations, again suggesting that impacts from cage waste accumulation had influenced macrofaunal community composition, with one or two species being present in high numbers, and other species being scarce.

In summary, although no clear tends were observed in sediment carbon and nitrogen, redox and macrofaunal data showed cage waste accumulation had some negative effects on sediment condition both under and in the immediate vicinity of the Moross 1 cage block. Results indicated that the primary impact zone (where sediments were strongly dominated by enrichment indicator species) at Moross 1 did not generally stretch beyond 25m from the cage edge, thus suggesting most waste accumulation was confined to a small area of seabed under and around the site. However, evidence of some background enrichment was found throughout the site, denoted by the abundance of species such as *Mediomastus fragilis* in sediments beyond the cage edges and at the control. There was no evidence to suggest that impacts had increased over the past five years, or that the impact zone had extended beyond that observed in earlier sampling years.

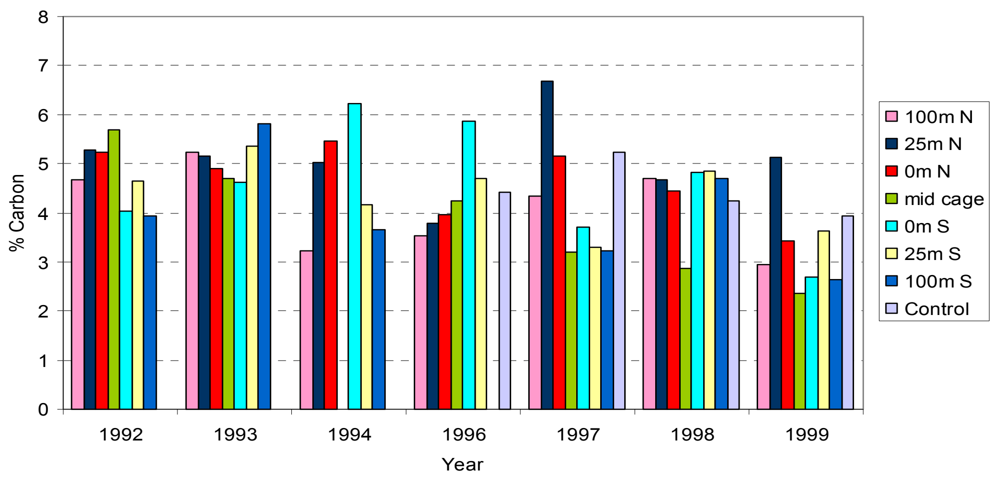

Figure b: % Carbon levels at Moross 1 between 1992 and 1999.

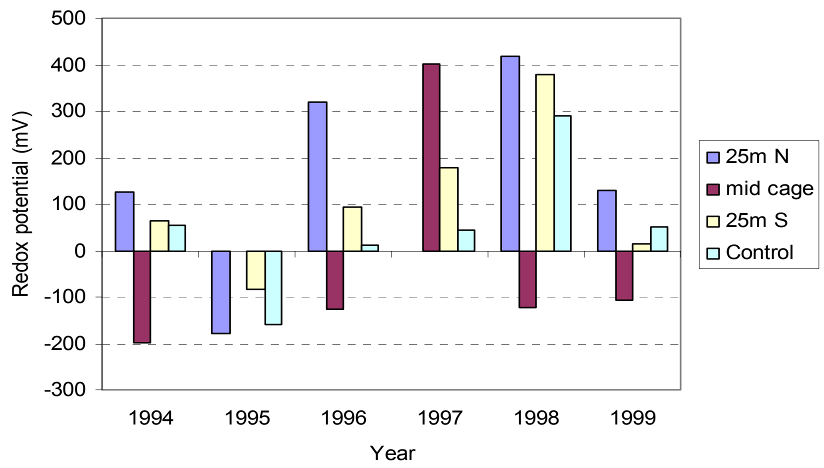

Figure a: Redox potentials at Moross 1 between 1994 and 1999.

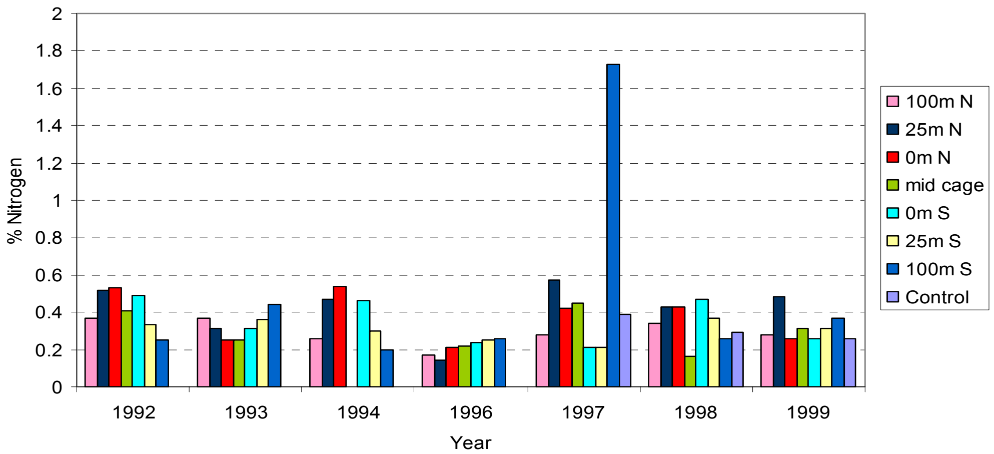

Figure c: % Nitrogen levels at Moross 1 between 1992 and 1999.

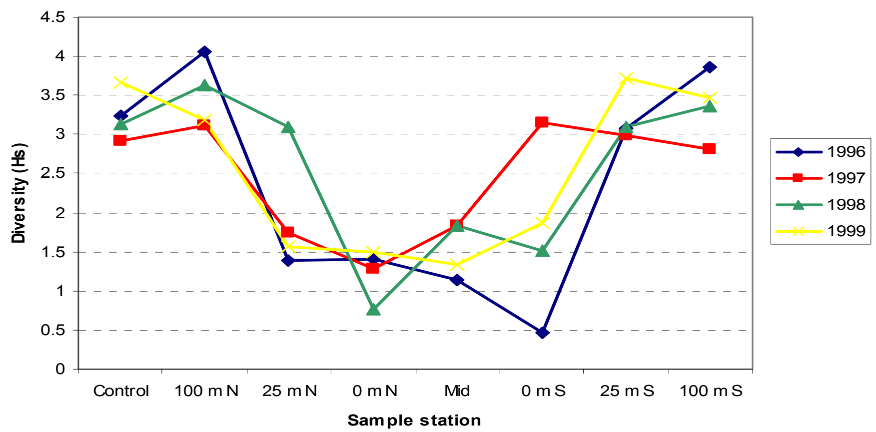

Figure d: Species diversity (Hs) levels at Moross 1 between 1996 and 1999.

Figure 3.24: Chemical and biological properties of sediments at Moross 1.

Moross 2

Salmon cages have been sporadically present at Moross 2 over the past decade. In the past five years, cages have been stocked three times: in 1996, 1999 and 2000. Results of sediment sampling and analyses during these years are given in Figure 3.25. As at Moross 1, the sediment at Moross 2 has traditionally been found to be composed of medium / soft, fine sand and mud, with signs of waste accumulation often being observed at mid cage and occasionally on the surrounding sediments. Redox values (shown in Figure 3.25(a) at -2cm) have generally reflected this, with mid cage values being lower than at the control site in recent years. However, values at 25m N have also been observed to be low (1999 and 2000) suggesting sediment deoxygenation has not always been confined to the immediate cage site boundary.

Carbon and nitrogen levels (as percentage dry weight) in sediments at Moross 2 are given in Figures 3.25 (b) and (c). In general, percentage carbon levels observed at Moross 2 over the years ranged between 2.37% (mid cage in 1999) and 7.08% (0m S in 2000). Mean percentage carbon values, however, were highest in 1993 (5.11%). These values suggested that percentage carbon content of the sediment was mostly low, with levels only exceeding 5% at sporadic intervals. Noticeably, percentage carbon levels at the control station were often higher than at mid cage, and in 2000, were among the highest recorded in that year. No clear trends have been displayed in percentage carbon data over the years, though carbon levels at the mid cage sample site appeared to be influenced by fallowing events, indicated by a fall in levels between 1993 and 1996, and between 1996 and 1999. Levels increased slightly at mid cage in 2000, likely as a result of consecutive stocking in 1999 and 2000. Percentage carbon levels at the other sample sites were often higher than at mid cage (as observed at Moross 1), suggesting that carbon levels in the background sediment were elevated.

Percentage nitrogen levels at Moross 2 (Figure 3.25 (c)) generally followed similar trends to those for percentage carbon within each sample year. Values at each sample site ranged from 0.14% at 25m N in 1996, to 0.52% at 25m S in 2000, with the highest mean value for all sites over the years being observed in 2000 (0.36%). These values were suggestive of mild nutrient inputs from waste feed. Lowest mean values were observed in 1996, likely due to some sediment recovery since previous fish stocks were removed in 1993.

As with carbon, mid cage nitrogen values throughout the Moross 2 cage site followed no obvious trends, with mid cage values often being lower than those at other sample sites. However, nitrogen levels in control sediments were lower than in sediments closer to the cages in 1999 and 2000, suggesting some waste accumulation may have occurred in these years. As described for the Moross 1 cage block, macrofaunal data for the Moross area dates back to 1988. Data collected during the period of 1988 to 1993 shows that sediments immediately under the Moross cage sites have traditionally been dominated by opportunistic species, notably *Capitella* spp. and *Malacoceros fuliginosus*. These species are tolerant of low sediment oxygen resulting from high nutrient loading. Areas of high impact (as denoted by the presence of opportunistic species) were found mostly to have been confined to the sediments directly below and around the cage sites, with less tolerant species such as *Pseudopolydora paucibranchia* dominating mildly enriched sediments beyond the cage edges. Macrofaunal diversity was found to increase in all years with increasing distance from the cages, with a return to background community composition generally being noted between 25m and 40m from the cages (IoA, unpub. data).

Results of recent macrofaunal analyses for Moross 2 are seen in Appendix 3 Table 2 and Figure 3.25 (d). As at Moross 1, the species assemblages observed at Moross 2 were typical of the fine nature of the sediment found in the vicinity of the cage site, again with species such as *Lumbrineris latreilli, Mediomastus fragilis,* Nematoda spp., *Prionospio fallax*, and other spionid polychaetes being commonly encountered.

In the late 1980s and early 1990s, enrichment indicators species, i.e. *Capitella* spp. and *Malacoceros fuliginosus* were found to be abundant in sediments close to the cage site, and dominated sediments at mid cage, 0m N and 25m N (and occasionally 0m S) in each sample year. This suggested that cage waste accumulation had had an impact on sediment quality and macrofaunal communities near to the cages. The fact that impacts were more widespread to the north may have been indicative of prevailing currents.

Macrofauna at 100m N was found to be very similar to the control site, suggesting sediment conditions approached those that could be considered background at that distance from the cage site. However, it should be noted that enrichment tolerant species such as *Mediomastus fragilis* were abundant at both locations, suggesting that nutrient levels were elevated in some areas of the background sediment. Macrofauna at 25m S and 100m S was not indicative of high levels of nutrient enrichment, and although was slightly different to that at the control (possibly due to differences in particle size), could be classed as representative of background conditions.

Diversity (Shannon Weiner) in sediments at Moross 2 varied little between survey years (Figure 3.25(d)), with trends being similar. As indicated from macrofaunal community compositions (Appendix 3 Table 2), diversity was found to be consistently low between 25mN and 0m S of the cage site, as a result of numerical dominance exerted by enrichment indicator species. Diversity scores were similar to those at the control at 100m N, 25m S and 100m S (except at 25m S in 1999 due to high abundances of *Lumbrineris latrelli* and *Jasmineira caudata*) suggesting sediments approached background conditions in those locations.

In summary, results of sediment analyses at Moross 2 over the years showed no overall trends of worsening or improving sediment conditions. Carbon and nitrogen levels were found to fluctuate between sample years in a fairly inconsistent manner, though the mean nitrogen level recorded in 2000 was higher than any recorded previously. This may have been due to that fact that long fallow periods existed prior to earlier survey years, but fish stocks had been in place over two consecutive years by July 2000, meaning waste accumulation would have been higher than previously recorded. Macrofaunal data showed consistent evidence of nutrient enrichment in sediments under and near to the cages, with a zone of highest impact stretching from 0m S to 25m N. However, the macrofaunal communities in sediments beyond these sample sites, and in background conditions at the control, were also found to reflect elevated nutrient levels, with high abundances of *Mediomastus fragilis* being noted at 100m N and at the control site. This suggests that low levels of nutrient enrichment may have been a feature of the sediments throughout the site, due to reasons other than fish farm activities.

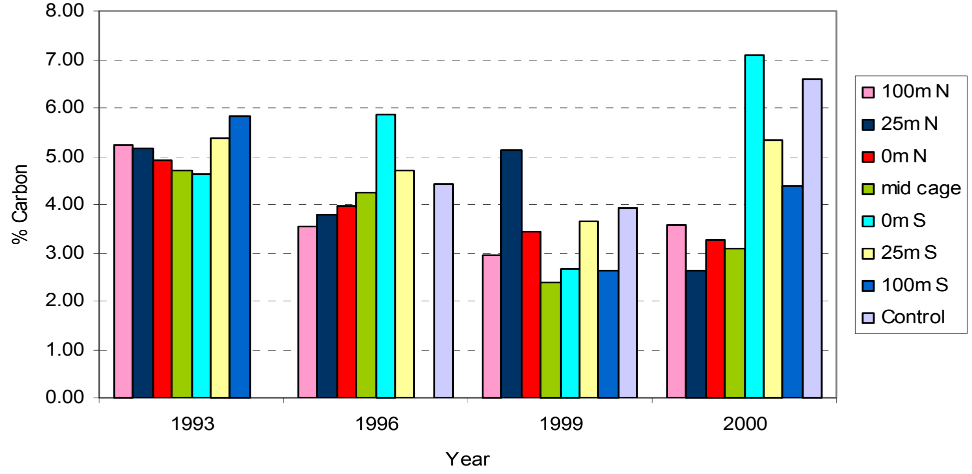

Figure b: % Carbon levels at Moross 2 between 1993 and 2000.

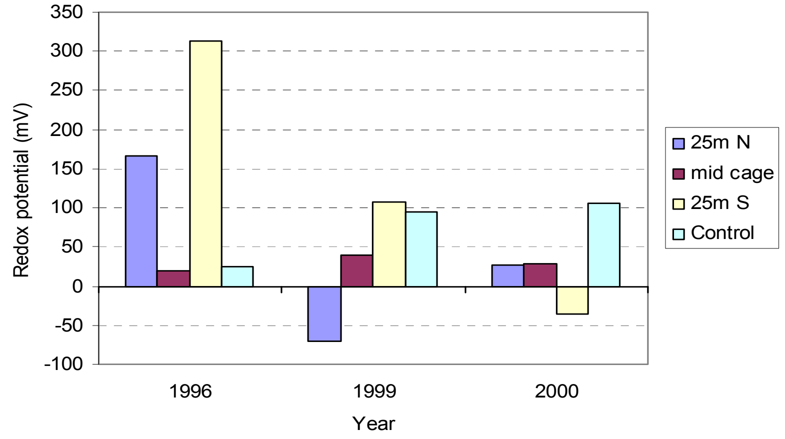

Figure a: Redox potentials at Moross 2 in 1996, 1999 and 2000.

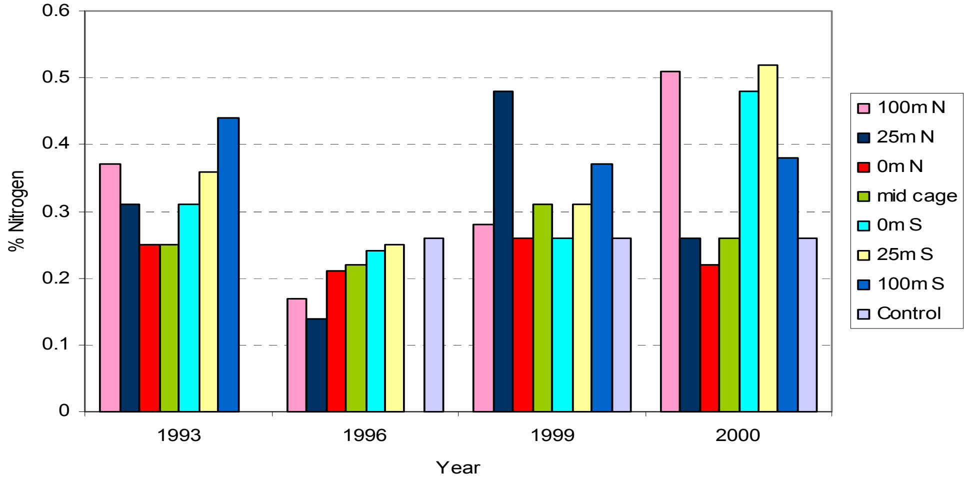

Figure c: % Nitrogen levels at Moross 2 between 1993 and 2000.

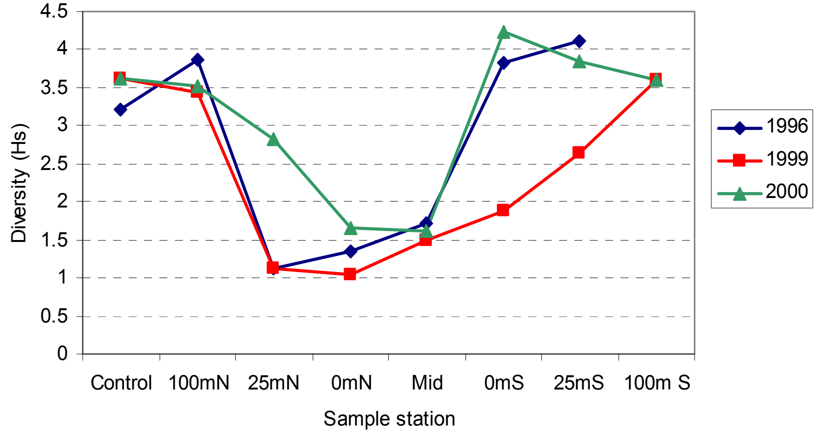

Figure d: Species diversity (Hs) levels at Moross 2 in 1996, 1999 and 2000.

Figure 3.25: Chemical and biological properties of sediments at Moross 2.

Cranford A

Cages have been present at Cranford since the mid 1980's, though these have moved and expanded several times over the years. Currently, three cage sites are present in the Cranford area, namely sites A, B and C. Some of these sites have been fallowed at various times since monitoring work began, so that only data for the last few years, when cages have been consistently stocked has been examined in detail. The Cranford sites are currently located near to an expanding area of mussel production lines, located several hundred meters to the south of the southern most salmon cage site (Cranford C). Sediment redox potentials, percentage carbon and nitrogen, and macrofaunal diversity at Cranford A are presented in Figures 3.26 (a) - (d). This cage block was moved prior to both the 1997 and 2000 surveys.

Cranford A is located in the middle of the three Cranford cage blocks. Sediment at the site is known to be of a very fine, soft nature, and waste feed accumulation has often been found to occur under the cages themselves. Redox values (-2cm) have been consistently low at mid cage (Figure 3.26 (a)), except in 1997 when the block was moved to a new location. This saw an increase in the mid cage redox level, though levels fell again in 1998. Redox levels at sample sites to either side of the cage block, and at the control site, decreased between 1998 and 2000, suggesting the conditions of sediments beyond the cage edge at Cranford A may have deteriorated slightly with time. This may not only have been due to continued fish production at Cranford, but also due to increases in other activities in the vicinity. Lowest overall redox levels were obtained in 1995, though levels improved in 1996.

Percentage carbon and nitrogen levels (as % dry weight) in sediments collected at Cranford A are given in Figures 3.26 (b) and (c). Values of both parameters have remained fairly consistent at all sample sites throughout the years, and have been similar to those found in sediments at the control site. Both carbon and nitrogen values have consistently been indicative of some sediment enrichment in sediments at all sample station in all survey years, with mean carbon values ranging from 7.41% in 1994 to 9.50% in 1995 (due to a peak at mid cage), and nitrogen values ranging from 0.45% in 2000 to 0.79% in 1995 (again largely due to a peak at mid cage). Values at the control sites were similar to those found at sample sites near to the cages, suggesting background enrichment levels in sediments throughout the Cranford area were elevated. Despite carbon and nitrogen values recorded at Cranford A being generally high, no trends of increasing enrichment were observed throughout the years, and, apart from sporadic peaks, levels remained consistent between surveys.

Macrofaunal data dating back to 1988 is available for the Cranford cage sites, though exact cage positions and sample locations from this period (1988 - 1995) are unknown. Descriptions of macrofaunal communities dating back to this time shows that sediments under and around the Cranford cage areas have traditionally been nutrient enriched, with species such as *Capitella* spp. and *Malacocerus fuliginosus* dominating sediments under the cages. Macrofauna from sediments out-with the immediate cage block areas have also traditionally been indicative of mild enrichment, with species such as *Pseudopolydora paucibranchia, Lumbrineris latreilli, Capitomastus minimus, Protodorvillea kefersteinia* and oligochaetes being commonly encountered, often in high abundances, at up to 80m from the production sites. Data from this period also suggests that highest effects have traditionally been confined to the sediments beneath and adjacent to the cages.

The diversity of macrofaunal species from 1996 onwards at Cranford A is presented in Figures 3.26 (d) (Shannon - Weiner diversity scores). Abundance of the commonly found taxa are given as ranked species (Appendix 3 Table 3). As at Moross 2, the species assemblages found throughout the Cranford A site during this time period were mostly representative of the fine, slightly enriched sediments known to be a feature of the area, with species such as *Scalibregma inflatum*, *Prionospio fallax*, *Lumbrineris latreilli, Mediomastus fragilis* and *Jasmineira caudata* being commonly encountered. As in earlier surveys, nutrient enrichment indicator species, namely *Capitella* spp. and *Malacoceros fuliginosus* were found in sediments near to and under the cage, notably at mid cage where they tended to be the most abundant species. Indicator species were also present among the most abundant taxa at 0m N and 0m S of the cage in most survey years, and were also encountered at 25m N in 1998. This may have been partly due to a change in cage position in the previous year, meaning sample sites were located over old areas of sediment enrichment.

The species assemblages found after 25m N and 25m S of the cage site resembled those at the control, suggesting that impacts were generally confined to the sediments in the immediate vicinity of the cage. This was reflected in diversity scores for sample stations throughout the years (Figure 3.26 (d)), with diversity being consistently low at mid cage, 0m S and 0m N and similar to the control at all other sample stations. The exceptions to this were at 25m N in 1996, when diversity was low due to a high abundance of the spionid polychaete, *Polydora caeca,* and in 2000, where diversity was similar to the control at all sample sites. This is likely to have been due to the change in cage position since the previous survey, meaning that although *Capitella spp*. were present in sediments under and around the cages, their levels were not high enough to exert overall dominance.

To summarise, the data from Cranford A showed that sediments were mildly enriched across the whole survey area, including the control, likely due to combined effects of on-going fish production, other aquculture activities in the area, and the fine nature of the sediments. Carbon and nitrogen values were consistent throughout the years, showing no signs of significant increase or decrease. The low oxygen levels found in sediments at mid cage, combined with the elevated nutrient status, meant sediments were typically dominated by enrichment indicator species, with diversity levels being low. However, no obvious trends of increasing or decreasing nutrient enrichment were observed from either carbon, nitrogen or macrofaunal results. Redox values were observed to fluctuate throughout the last five years, though decreased slightly towards 2000, possibly suggesting sediment oxygen levels were slowly decreasing as a result of prolonged enrichment effects.

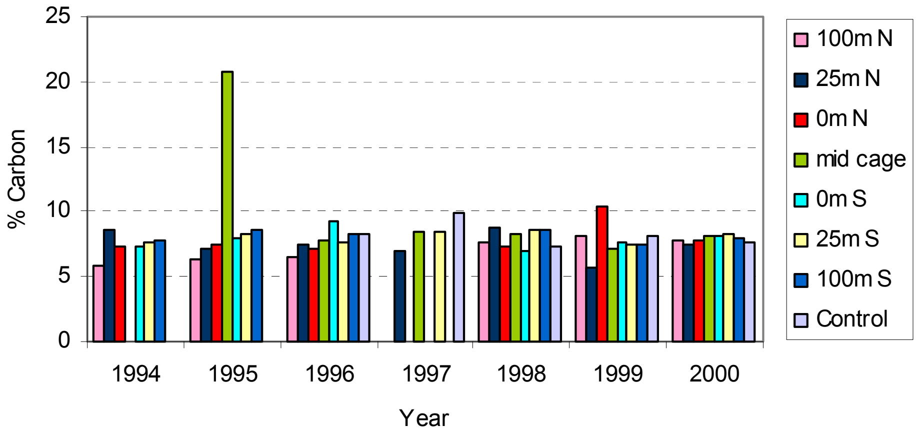

Figure b: % Carbon levels at Cranford A between 1994 and 2000.

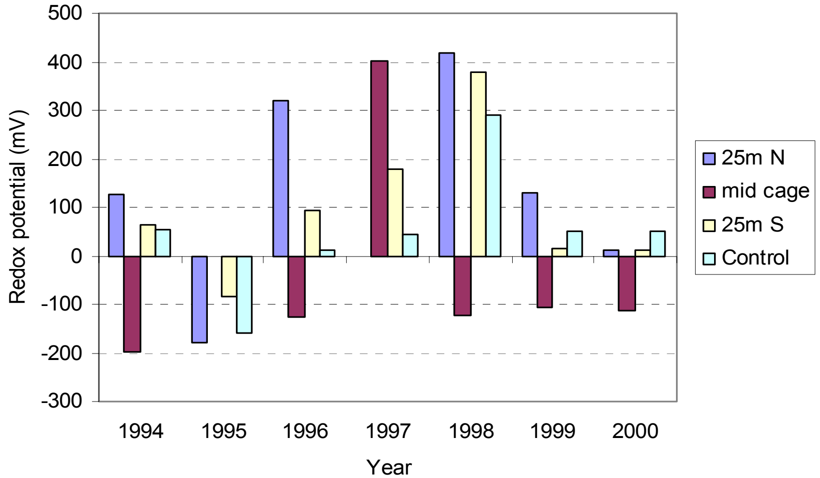

Figure a: Redox potentials at Cranford A between 1994 and 2000.

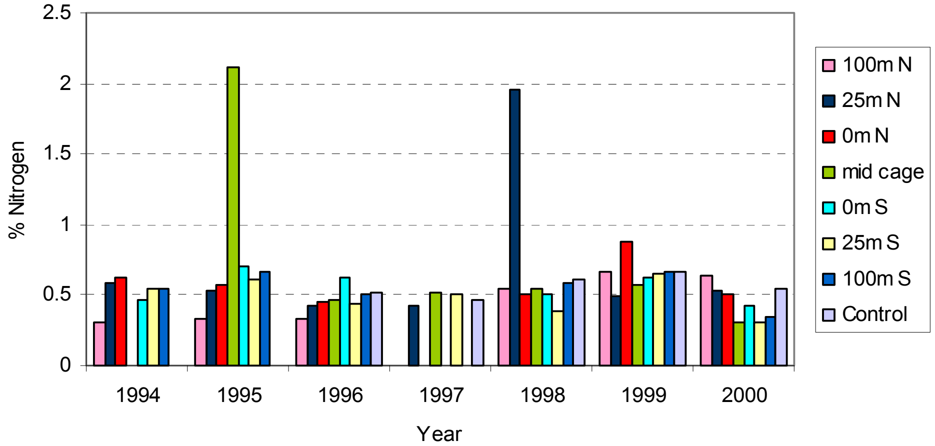

Figure c: % Nitrogen levels at Cranford A between 1994 and 2000.

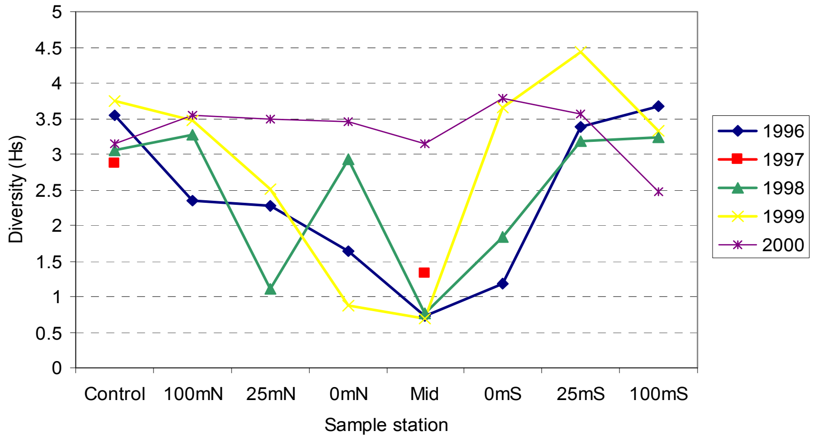

Figure d: Species diversity (Hs) at Cranford A between 1996 and 2000.

Figure 3.26: Chemical and biological properties of sediments at Cranford A.

Cranford B

Cranford B cage site has been present in the Cranford area of Mulroy Bay for several years, though as at Cranford A, it has been moved from its original position twice in the past 5 years (once in 1995-97 and once in 1999 to a position north of Cranford A). Consequently, this will have influenced the continuity between sampling sites and results in different survey years, so that long-term trends specifically in one location may not be clear. Results of sediment and macrofaunal analyses at Cranford B are given in Figures 3.27 (a) - (d) and in Appendix 3 Table 4.

The sediment at Cranford B has traditionally been found to be very similar to that at Cranford A (as would be expected for neighbouring sites), being of a very fine, soft nature. In recent years, waste feed has been observed to accumulate in sediments at both mid cage and 25m N, with the sediment coloration being grey / black, suggesting some oxygen depletion. However, fine sediments are often slightly oxygen depleted below the surface due to the small interstitial spaces that restrict water exchange and oxygen replenishment with the overlying water column. This was reflected in redox measurements (-2cm) recorded at the site in 2000, when levels, particularly at mid cage and 25m N, were low.

Trends in sediment carbon and nitrogen levels were found to be similar between years 1997 and 2000, with levels of both parameters being highest in 1997 and lowest in 2000. Mean percentage carbon levels ranged from 4.53% in 2000 to 7.88% in 1997, and levels at the control site were often higher than those nearer to the cages. As at Cranford A, this reflected the high background nutrient levels in the sediment throughout the Cranford area. Mid cage carbon levels were the highest of those found near to the cages in 1997, 1999 and 2000, though in all cases, values were less than or equal to those found in the control sediment. In general the % carbon results suggested sediments at Cranford B were mildly enriched, though the fact that levels decrease with time to their lowest in 2000, possibly indicated that overall, nutrient accumulation had reduced slightly in recent years.

As for carbon, nitrogen values varied throughout the past four years, with the highest mean value being obtained in 1997 (0.43%), and the lowest in 2000 (0.11%). Again, control nitrogen levels were often higher than those found in sediments near to the cages, reflecting the high background nutrient levels in sediments throughout the whole Cranford area. Percentage nitrogen values appeared to decrease to their lowest in 2000, again, possibly indicating reduced inputs of waste feed to the sediments in recent years.

As at Cranford A, macrofaunal data from Cranford B generally reflected the fine slightly nutrient enriched nature of sediments at the cage site, with species such as *Mediomastus fragilis, Prionospio fallax, Pseudopolydora paucibranchia* and other spionid polychaetes being commonly encountered. Many of these species have been commonly recorded since monitoring began in the late 1980s (see 3.3.2.3 for a description of macrofaunal assemblages at Cranford during the late 1980s /early 1990). Species assemblages at 100m S, 25m S and 100m N were similar to those at the control site. However, enrichment indicator species such as *Capitella* spp. and *Malacoceros fuliginosus* were abundant nearer to the cages. *Capitella* spp. dominated the macrofauna at 0m S in 1997 and 1998, though were not among the top five species in 1999 and 2000, with *Spio decorata* and *Mediomastus fragilis* (also known as an enrichment tolerant species) taking over as the most numerous taxa. This may have been due to changes in cage position and / or changes in production levels at the site. Indicator species (mainly *Capitella* spp. and / or *Malacoceros fuliginosus*) tended to dominate the macrofaunal community at mid cage, 0m N and 25m N between 1998 and 2000, though abundances of these were reduced in 2000, possibly reflecting a decrease in nutrient inputs to the sediment.

Diversity (Shannon Weiner) values for macrofauna at Cranford B are given in Figure 3.27 (d). Values fluctuated between years, though tended to be lowest at mid cage and 0m from the cage site, reflecting the high abundance of indicator species in those locations. The lowest diversity score was obtained at 0m S in 1997, due to the strong dominance of *Capitella* spp. at that time. Diversity values recorded at 25m S and 100m S were generally similar to those at the control in most years, though values were depressed for a distance of up to 100m from the cage site to the north in 1997 and 1998. This was not due to dominance by indicator species, but to the abundance of spionid polychaetes often associated with sediment of the nature found at the cage site.

In summary, sediments below the cages at Cranford B were found to be nutrient enriched in most years, though carbon and nitrogen levels, and abundance of enrichment indicator species fell in 2000, possibly resulting from reduced nutrient inputs. This may have been associated with a change in cage position, and /or changes in production intensity at the site. Carbon and nitrogen levels were consistently lower than those in the control sediment in recent years, reflecting the high overall background enrichment of sediments in the Cranford area.

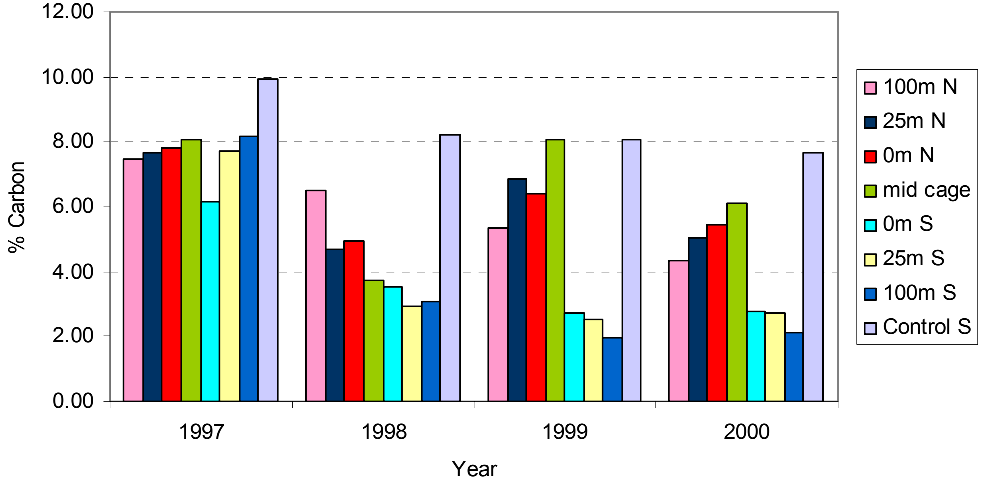

Figure b: % Carbon levels at Cranford B between 1997 and 2000.

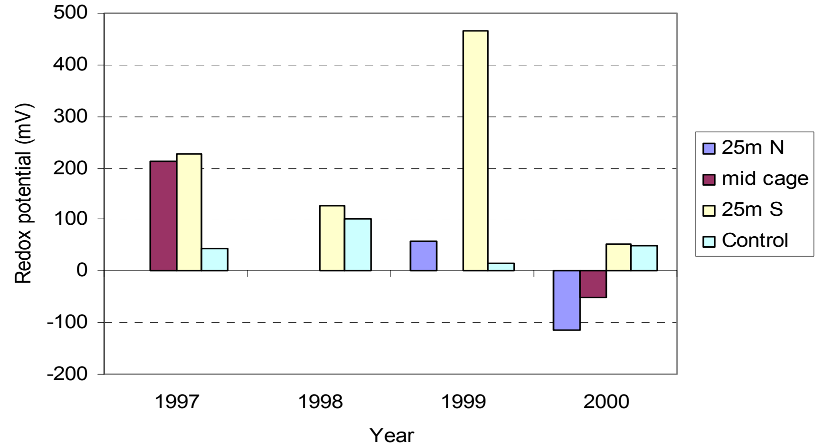

Figure a: Redox potentials at Cranford B between 1997 and 2000.

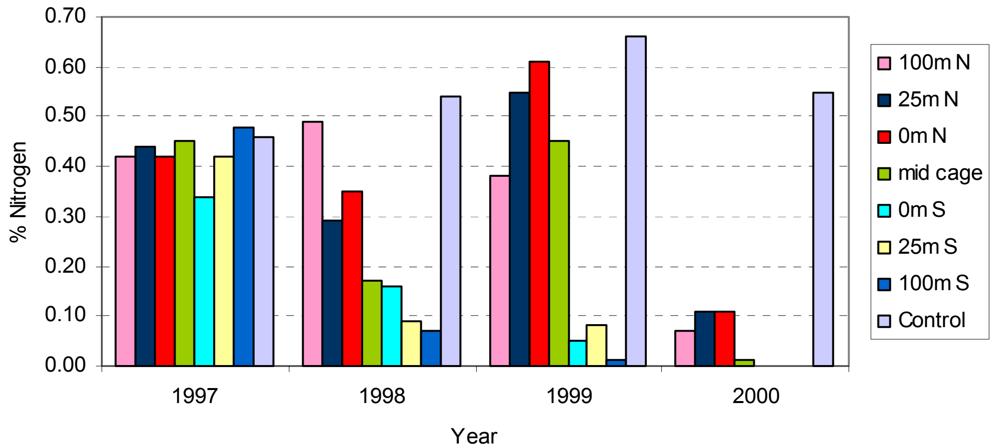

Figure c: % Nitrogen levels at Cranford B between 1997 and 2000.

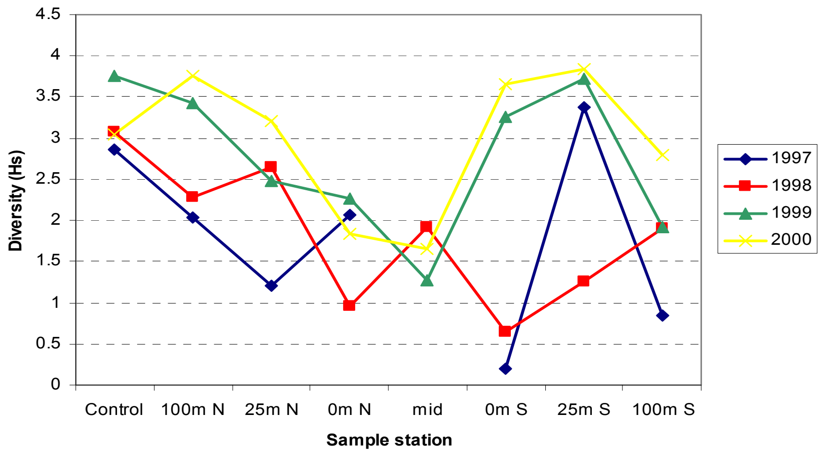

Figure d: Species diversity (Hs) at Cranford B between 1997 and 2000.

Figure 3.27: Chemical and biological properties of sediments at Cranford B.

Cranford C

Sediment chemistry and macrofaunal data for the cage site at Cranford C are given in Figure 3.28 and in Appendix 3 Table 5. Cages were absent in 1998. As at the other Cranford sites, sediment at Cranford C is known to be of a fine, soft nature, and has often appeared to be slightly oxygen depleted, with waste feed and hydrogen sulphide detection being noted periodically at mid cage. Redox values (-2cm) at the site (Figure 3.28 (a)) were found to fluctuate between sample years, though were lowest in 2000, notably at mid cage and 25m N.

Carbon levels in the sediments measured at Cranford C are given in Figure 3.28 (b). Though levels fluctuated slightly between years, values at all sample stations were generally indicative of some nutrient enrichment, with levels ranging from an mean of 6.92 % in 1999 to 8.69 % in 1994. Control site carbon levels were found to be similar to those observed at sample stations under and around the cage site, again showing that background sediments were nutrient enriched.

The mean percentage nitrogen levels recorded annually in sediments at Cranford C, as for carbon, were highest in 1994 (0.85%) and lowest in 2000 (0.25%). Levels tended to show a slight decrease over time, possibly suggesting that waste feed inputs to sediments (for which nitrogen levels would be elevated compared with faeces and background sediment) had reduced slightly. Again, nitrogen levels at the control site were higher than those nearer to the cages in recent years, possibly reflecting nutrient inputs or accumulation from other sources in the Cranford area.

The top five ranked abundant species found in sediments at Cranford C between 1996 and 2000 are shown in Appendix 3.7. Many of these species have been commonly recorded since monitoring began in the late 1980s (see 3.3.2.3 for a description of macrofaunal assemblages at Cranford during the late 1980s /early 1990). As at the other Cranford sites, the species encountered generally reflected the fine, slightly enriched nature of the sediment with species such as *Scalibregma inflatum, Lumbrineris latreilli* and *Mediomastus fragilis* being numerous throughout the survey area. The indicator species *Capitella* spp. dominated macrofauna at mid cage in every survey year, and at both 0m N and 0m S in 1997 and 1999, suggesting highest nutrient loading and therefore waste accumulation, was confined to the immediate vicinity of the cage site. However, in 2000, the species was absent at the latter two sample stations, possibly indicating some sediment recovery. This would agree with findings of % nitrogen analyses, though would not agree with decreased redox values observed in 2000.

Macrofaunal diversity (Shannon-Weiner) scores for sample stations at Cranford C are given in Figure 3.28 (d). Values were consistently low at mid cage, reflecting the dominance of *Capitella* spp. in sediments directly beneath the cage site. Diversity was also low at 0m S and 0m N in years when *Capitella* spp. were dominant at those locations, though values were higher in 1996 and 2000, when indicator species did not exert numerical dominance over the assemblage. Values at more distant sample stations (i.e. beyond 25m from the cage edge) were mostly similar to those at the control, suggesting that impacts were largely confined to the mid cage and cage edge sediments.

In general, sediment conditions at Cranford C were similar to those at the other Cranford cage sites, with highest impacts on macrofauna being confined to sediments directly below or adjacent to the cages. Despite redox values being lowest in 2000, nitrogen levels had fallen since the previous year, and macrofaunal community structure had changed slightly, suggesting that nutrient inputs to the sediment may have been reduced.

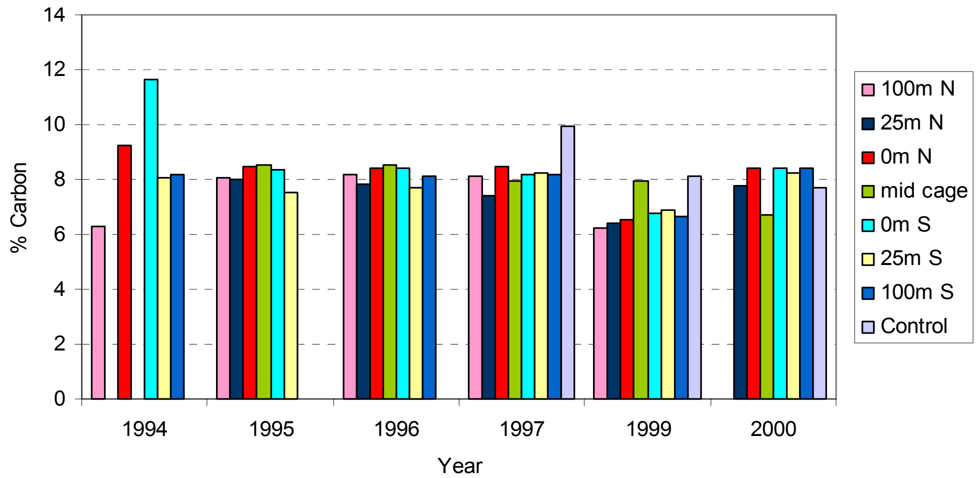

Figure b: % Carbon levels at Cranford C between 1994 and 2000.

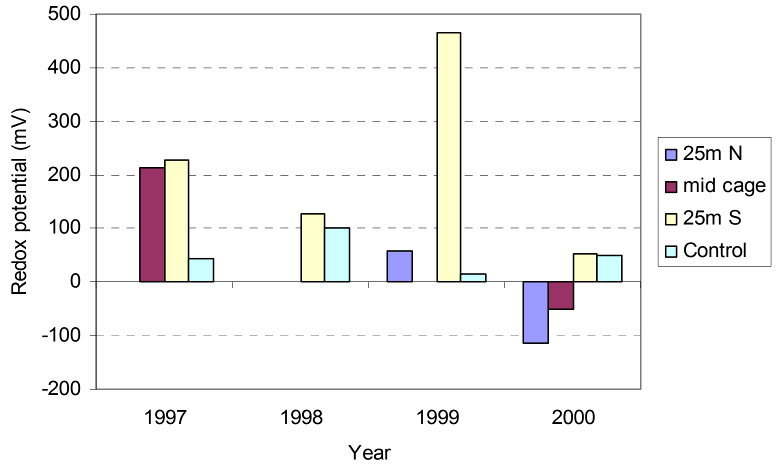

Figure a: Redox potentials at Cranford C between 1997 and

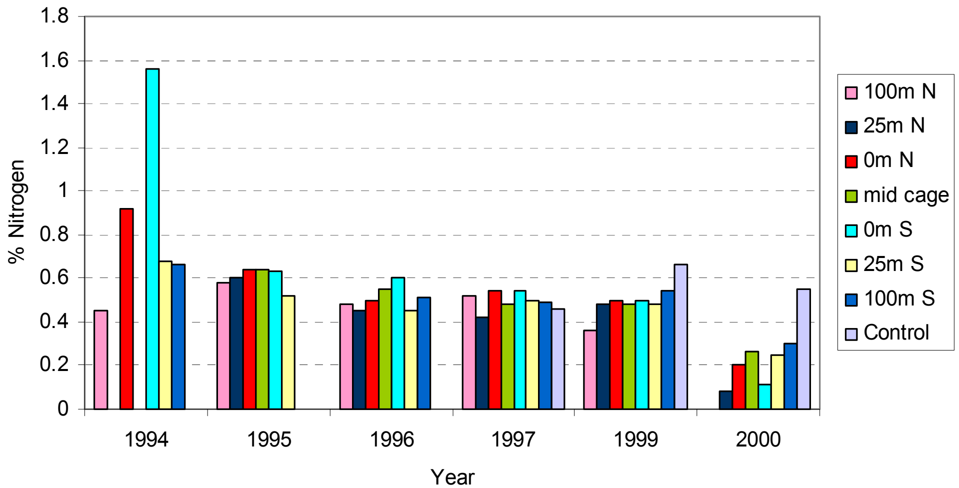

Figure c: % Nitrogen levels at Cranford C between 1994 and 2000.

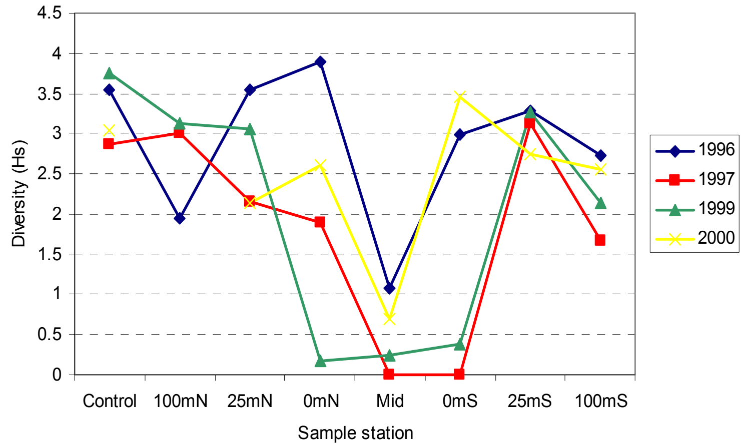

Figure d: Species diversity (Hs) at Cranford C between 1996 and 2000.

Figure 3.28: Chemical and biological properties of sediments at Cranford C.

Millstone

Cages at the Millstone production site are located in the Narrows part of Mulroy Bay, where tidal flows are often quite strong, allowing for good waste dispersion potential (see Chapter 4). Cages have been consistently present and stocked for many years, though have periodically been moved to slightly new locations, as in 1999 / 2000. Sediment chemistry and macrofaunal data are presented in Figures 3.29 (a) to (d) and in Appendix 3 Table 6, and are discussed below.

Sediment at Millstone is known to be composed of firm, medium to coarse sand, with maerl often being observed on the surface. Near to the cages, waste feed has periodically been observed on the sediment surface, with the resident maerl occasionally dying off due to accumulation of cage wastes. Signs of some oxygen depletion and hydrogen sulphide have occasionally been detected in mid cage sediments, reflected by redox values recorded throughout the site (Figure 3.29 (a)). Lowest redox values (-2cm) occurred in the 1999 survey, though values had improved by 2000, likely due to the change in cage position. Redox values prior to this had been highest in 1997, falling at mid cage in 1998.

Carbon levels in sediments at Millstone are presented in Figure 3.2(b). These were not indicative of high levels of waste accumulation, with mean values ranging from 4.60% in 1998 to 3.08% in 2000. Peaks above 5% carbon were occasionally observed at stations between 25m N and 25m S of the cage site, with mid cage values often being among the highest recorded in each survey year. Control sediment carbon levels were generally lower than those recorded at the cage site, except in 2000, where a decrease in carbon levels was observed at most stations to the North (possibly due to a changed cage location), while an increase had occurred at the control.

Nitrogen levels at Millstone (Figure 3.29 (c)) were found to be consistently highest (or among the highest) at the mid cage station in each survey year, indicating this was a site of some nutrient accumulation from cage wastes. Mid cage values ranged from 0.21 % in 1997 and 2000, to 0.62 % in 1995, while mean % nitrogen values ranged from 0.16% in 1996 to 0.22 % in 1997. Overall, a trend of decreasing nitrogen level was observed in mid cage sediments over the years. However, a peak in nitrogen levels was observed at 0m S in 2000, possibly due to the recent change in cage location, placing 0m S over a previous mid cage position.

Macrofaunal data from Millstone collected between 1996 and 2000 is presented in the form of ranked top five species in each year (Appendix 3 Table 6), and diversity scores in Figure 3.26 (d). (No data was available prior to this period). The species assemblages found throughout the Millstone area reflected the medium to coarse sediment, with spionid polychaetes, amphipods, and other crustaceans being abundant. Oligochaeta spp., such as *Tubificoides benedenii* were present at several sample sites, including the control, suggesting that either some low levels of enrichment or a freshwater influence was present in the background environment.

Nutrient enrichment indicator species, such as *Capitella* spp. and *Malacoceros fuliginosus* were present in mid cage sediments in all survey years since 1996, suggesting this was a site of cage waste accumulation. Indicator species were also present at 0m S and up to 25m N in most years, with *Capitella* spp. also being relatively abundant at 100m N in 1997. However, the abundance of indicator species was generally lower than those found at other cage sites in Mulroy Bay, suggesting that although macrofaunal communities in sediments were affected by cage waste accumulation, the degree of impact was less.

Macrofaunal diversity (Shannon Weiner) scores for sample stations at Millstone are given in Figure 3.29 (d). Diversity was generally lowest between mid cage and 25m N of the site, possibly due to the dispersal of wastes in a northerly direction by prevailing current flows. Results reflected the tendency for opportunistic species to dominate sediments near to the cages, with diversity levels increasing to those that were similar to the control at 25m S and 100m N.

To summarise the results of sediment analyses at Millstone over the years, sediment was generally found to have low carbon and nitrogen values, possibly due to the coarse, mobile nature of the sediment in the area. Despite this, mid cage carbon and nitrogen levels values were often among the highest recorded at Millstone each year. A trend of decreasing nitrogen levels was observed in recent surveys, possibly due to reduced inputs of waste feed at the cage site. Signs of waste accumulation and oxygen depletion were often encountered at mid cage and in sediments to the north, which was reflected by low macrofaunal diversity and the presence of opportunistic species in these areas. Waste accumulation, though lower than at some other cages sites, tended to occur in sediments under and to the north of the cages, likely due to prevailing current flows.

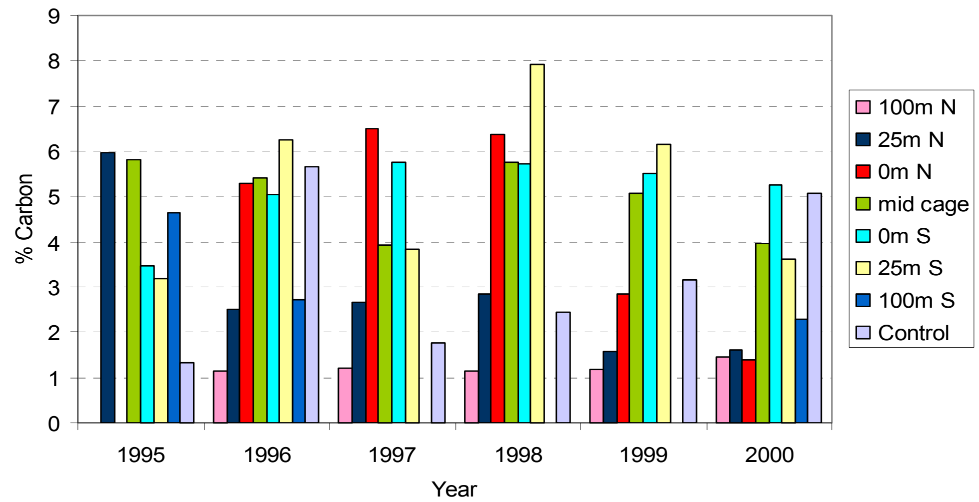

Figure b: % Carbon levels at Millstone between 1995 and 2000.

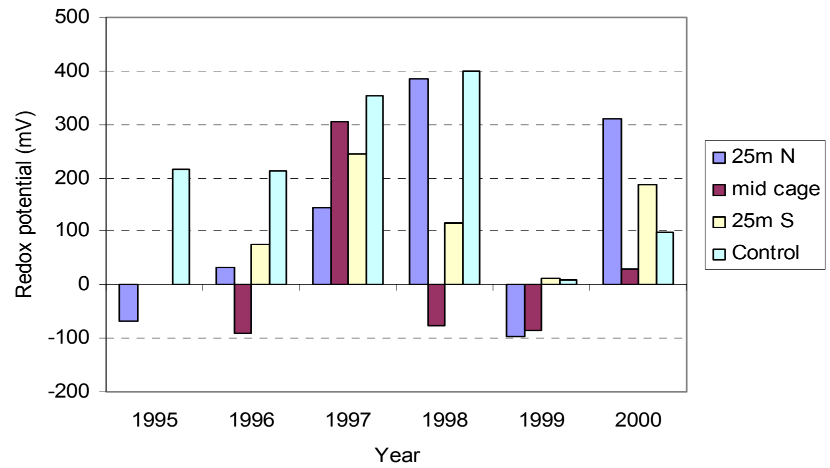

Figure a: Redox potentials at Millstone between 1995 and 2000.

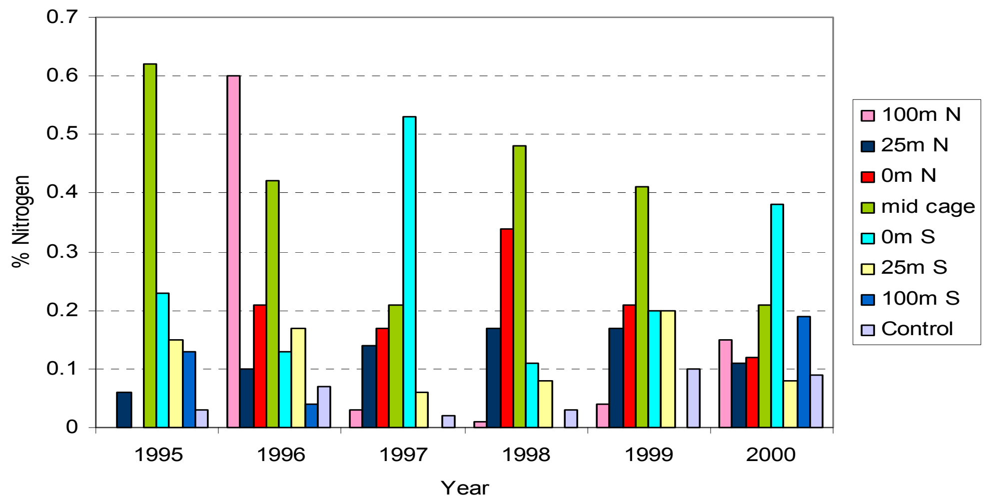

Figure c: % Nitrogen levels at Millstone between 1995 and 2000.

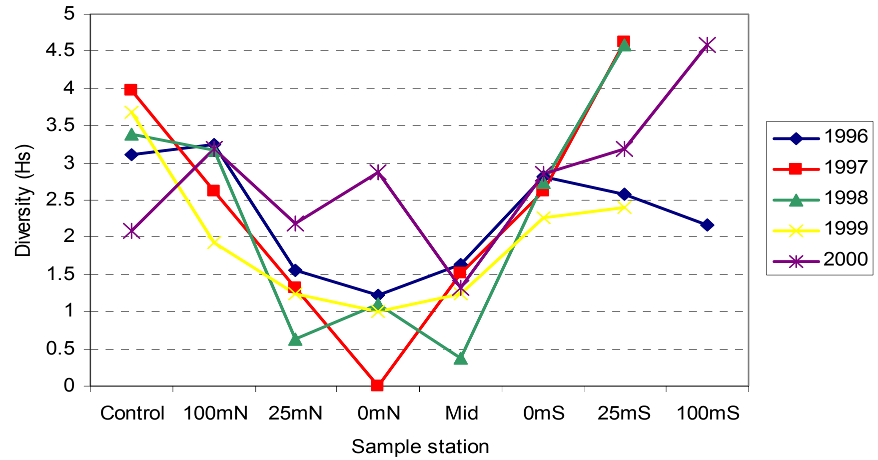

Figure d: Species diversity (Hs) at Millstone between 1996 and 2000.

Figure 3.29: Chemical and biological properties of sediments at Millstone.

Kindrum

Cages have been present at Kindrum since salmon production began in Mulroy Bay in the mid 1980's. In recent years, intensive fish production at the site has been ceased, and is now only used to hold broodstock fish on a temporary basis. The cages at Kindrum are situated close to the shore in an enclosed bay in the Northwater part of Mulroy Bay, where waters are fairly deep. Sediment chemistry and macrofaunal data collected at the cage site are presented in Figures 3.30 (a) to (d) and in Appendix 3 Table 7. Traditionally, data collection has only possible to the south of the site, due to the proximity of the cages to the shore on the northern side.

Sediment at Kindrum has been traditionally composed of very soft, fine mud and silt, with low seabed current flows being a feature of the area. Large amounts of wastes have been observed to accumulate under the cages from time to time, and sediment deoxygenation has, at times, been severe, with hydrogen sulphide production being consistently recorded under, and to the south of the site. Redox values (-2cm) at Kindrum cage site are given in Figure 3.30 (a). These have been consistently low at mid cage (except in 1995) and up to 25m S, showing that oxygen depletion in the sediments has been a widespread phenomenon over the years. Control sediments have generally been more oxygenated than those near to the cages over the years, though levels were observed to fall in recent surveys.
Sediment carbon levels at Kindrum are given in Figure 3.30 (b). These have generally been indicative of moderate nutrient enrichment in the sediments collected every year, with mean values ranging from 7.47 % carbon in 1999 to 10.24 % carbon in 1995. Within each survey year, mid cage carbon levels have generally been the highest; these decreased from their maximum (14.22 %) in 1995 to a minimum of 8.94 % in 1998 before increasing slightly to 9.80 % in 2000. A peak in % carbon was also noted at 25m S in 1995, though this was not observed in following years. Percentage Carbon levels at the control were also found to be slightly elevated (> 5 %), and were indicative of some nutrient enrichment of the background sediments in the area. This may have been partly due to low current flows, the basin-like morphometry of the seabed, and inputs from both fish farm and other activities in the vicinity.

Nitrogen levels at Kindrum were similar to those for carbon, in that highest levels in each survey year were recorded in mid cage sediments (Figure 3.30 (c)). Levels throughout the cage site in each year were elevated, and were again suggestive of nutrient inputs from cage waste accumulation. Mean % nitrogen values ranged from a minimum of 0.6 % in 2000 to a maximum of 0.88 % in 1999, while mid cage values ranged from a minimum of 0.60 % in 2000 to a maximum of 1.55% in 1995. Nitrogen levels in sediments at the control site, though generally slightly lower than those close to the cage site, were found to be elevated above those expected in unenriched sediments, again suggesting that the whole area was prone to nutrient accumulation and enrichment.

Macrofaunal data has been collected at Kindrum since 1988, and has often reflected the high levels of nutrient accumulation observed in sediments under the cages. Macrofauna collected directly beneath the production site has traditionally been poor, with opportunistic species such as *Capitella* spp. and *Malacoceros fuliginosus* dominating fauna. At times (e.g. 1990-1992, 1999 and 2000) opportunists (indeed all macrofauna) have been scarce or absent in this area, due to the highly anoxic nature of the sediment.

Detailed macrofaunal data at Kindrum from 1996 onwards is presented in Appendix 3 Table 7 and in Figure 3.30 (d). Species assemblages at the control site and 100m S have usually been typical of the fine, slightly enriched sediment found in the Kindrum area, with species such as Nemertea spp., *Mediomastus fragilis, Prionospio fallax* and *Scalibregma inflatum* being abundant. As in earlier surveys, opportunistic species, associated with areas of nutrient enriched sediments (e.g. *Capitella* spp., *Mediomastus fragilis* and *Malacoceros fuliginosus*) were found consistently at mid cage (where nutrient enrichment was highest according to chemical analyses), and at 0m S and 25m S between 1996 and 1999. No macrofaunal organisms were found in sediments collected at mid cage and 0m S in 1999, likely due to very poor sediment quality in that year (as indicated by highest mean nitrogen levels in 1999). Species assemblages showed a slight sign of recovery at 0m S and 25m S in 2000, with a change from a few opportunists to *Mediomastus fragilis* dominated communities. (This species has also been associated with highly enriched sediments (Pearson and Rosenberg, 1987)). However, only 3 individuals in total were found at mid cage, showing that sediment conditions remained poor directly below the cages.

Species diversity (Shannon-Weiner) values for macrofaunal communities found at Kindrum are presented in Figure 3.30 (d). Diversity at all stations, including the control, was relatively low in all years, with no values above 3 being recorded. As might be expected, lowest diversity values were recorded at mid cage and up to 25m S in all years. No diversity values could be calculated for 0m S and mid cage sediments in 1999 due to a total absence of individuals in the samples collected. Diversity increased with distance away from the cages, to reach levels similar to those at the control by 100m S.

In summary, the data collected from sediment samples at Kindrum over the years has shown quality to be consistently low, both in terms of low oxygen and high carbon / nitrogen levels. This has been reflected in the macrofaunal community, with opportunistic species dominating sediments throughout the cage site, though often in low numbers due to very poor conditions. Macrofauna were found to be absent from samples collected near to the cages in 1999, suggesting this was a time of very low sediment quality. Sediment quality has shown little or no sign of improvement over time, with occasional deterioration occurring in recent years. Sediment at the control site near to Kindrum also showed signs of moderate nutrient enrichment, likely due in to the combined effects of low seabed current flows, fish farm waste dispersion and accumulation of nutrients and detritus from a variety of other sources (such a land run-off).

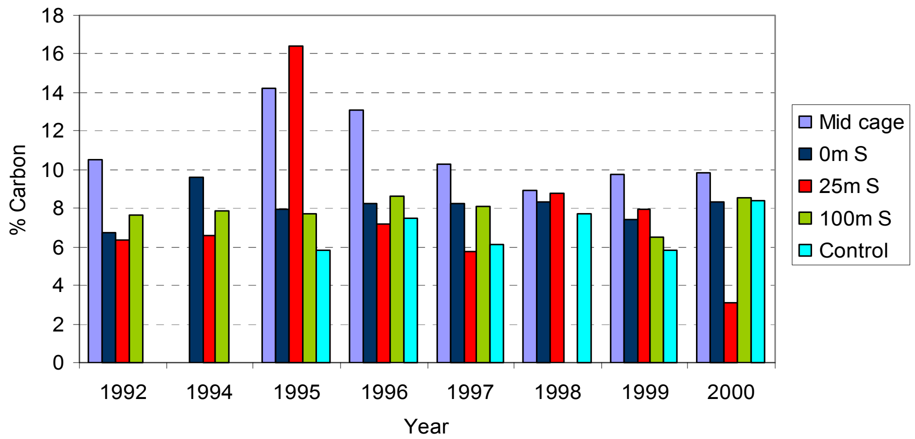

Figure a: % Carbon levels at Kindrum between 1992 and 2000.

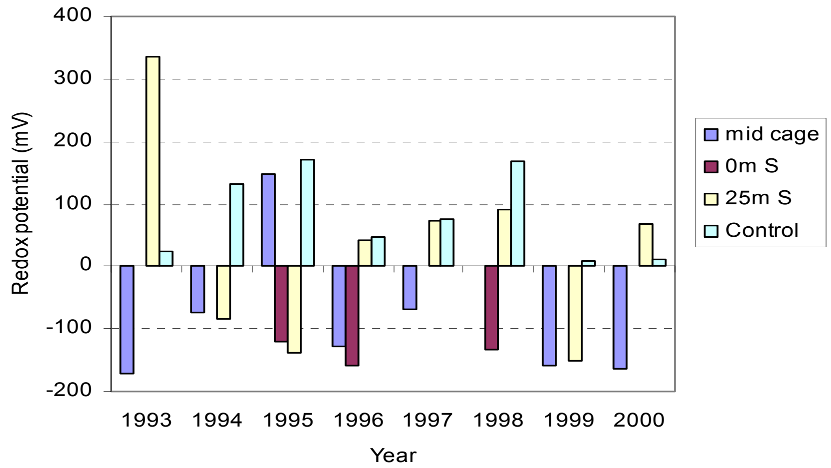

Figure b: Redox potentials at Kindrum between 1993 and 2000.

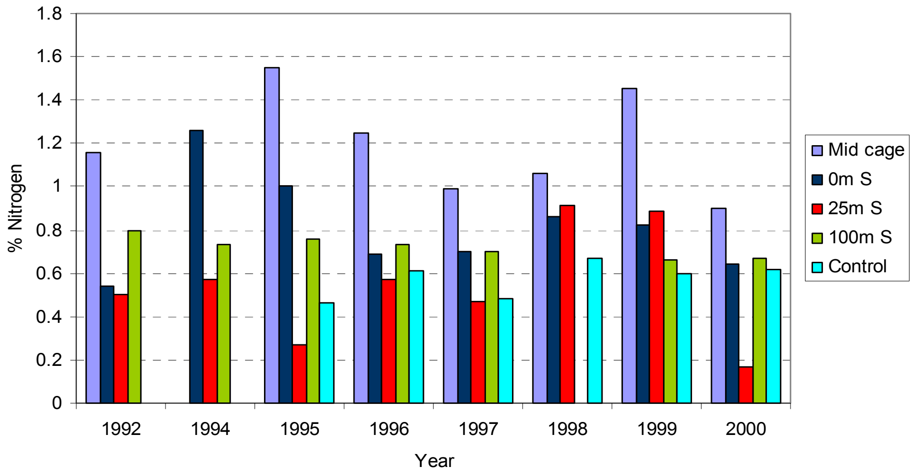

Figure c: % Nitrogen levels at Kindrum between 1992 and 2000.

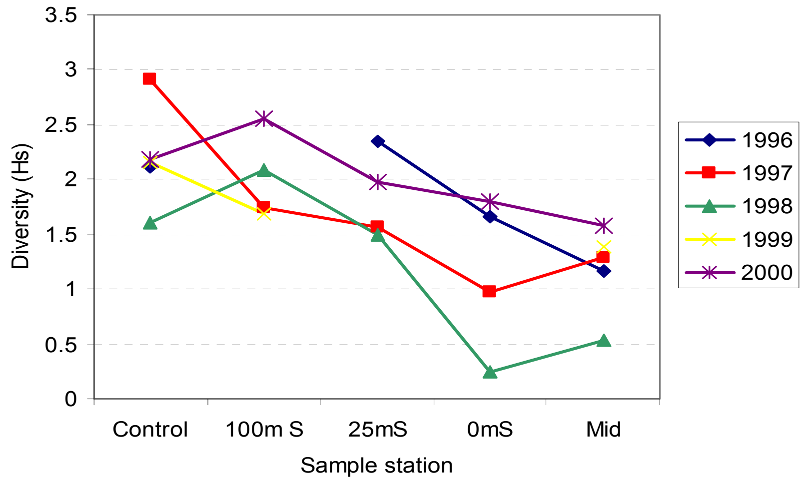

Figure d: Species diversity (Hs) at Kindrum between 1996 and 2000.

Figure 3.30: Chemical and biological properties of sediments at Kindrum.

Glinsk

The salmon production site at Glinsk is located towards the outer reaches of Mulroy Bay, in an area where current flows are often high. As a result, sediment at Glinsk has been found to be of a mostly coarse nature, with patches of both live and dead maerl being observed on the sediment. As with the other fish production sites in Mulroy Bay, cages have been moved at Glinsk from time to time. For example, new cages were added to the northern end of the site in 1995, then moved 50m NE in 1996. A further move occurred prior to the survey undertaken in 2000. As a result sample station locations described in the following section will not necessarily be consistent with those in previous years, meaning trends may be difficult to discern.

Redox values at Glinsk are given in Figure 3.31 (a). These show that sediments have traditionally been fairly well oxygenated at all sample stations except at mid cage, with the exception of 1999, when levels at 25m N were also low. Redox levels in the control sediments were often lower than those recorded near to the cages, except in 1999 and 2000, when redox levels at sample stations were slightly lower at most stations than in previous years, suggesting a slight reduction in sediment quality.

Carbon levels in sediments at Glinsk are shown in Figure 3.31 (b). Carbon levels tended to increase from north to south along the direction of prevailing current flow in all survey years. Mean carbon levels ranged from 4.9 % in 1996 to 6.0 % in 1998, though maximum values within each year remained consistent at between 7.0 % to 8.5 %. Carbon levels at mid cage and to the south were found to be >5 % in each year, though this was not thought to be due to high levels of cage waste accumulation. Rather, the elevated carbon levels were likely to have been due to the presence of large amounts of shell matter and maerl in the sediments (shell is composed largely of calcium carbonate and may therefore increase percentage carbon levels). No overall trends of increasing or decreasing levels over the years were observed in the data presented.

Nitrogen values at Glinsk were generally very low at sample stations in all survey years, as shown in Figure 3.31 (c). The exception to this was at mid cage in 1998, when levels reached a peak of 0.44%. Aside from this nitrogen values remained below 0.15%, with mean values in each year ranging from 0.04% in 1999 and 2000, to 0.15% in 1998. These results suggested that accumulation of waste fish feed and faeces in the sediments at Glinsk was low. As with carbon, no trends of increasing nitrogen levels were apparent from the data collected, suggesting waste inputs had not increased in recent years.

Macrofaunal data collected at Glinsk is presented in the form of a table of the top five ranked abundant species (Appendix 3 Table 8) and a plot of diversity (Shannon-Weiner) scores in Figure 3.31 (d).

The ranked species abundance data in Appendix 10 showed that assemblages observed at Glinsk between 1996 and 2000 were similar to those observed at the nearby Millstone cage site, and reflected the coarse sandy nature of the sediment (bivalve molluscs, spionid polychaetes, *Scoloplos* armiger, *Nepthys* spp. and Maldanids (such as *Myriochele spp.*) were commonly observed throughout the area). Despite indications from sediment chemistry data that nutrient levels in the sediment were low, opportunistic species such as *Capitella* spp. and *Malacoceros fuliginosus* were consistently present as the most abundant taxa (along with *Tubificoides benedenii*) at the mid cage site in each survey year. These species were often noted to be present in sediments up to 25m distance from the cage site between 1997 and 2000, suggesting some waste accumulation had taken place. However, abundances of these species were low compared with other cage sites, suggesting waste accumulation levels were also comparatively low. The low nitrogen levels observed may reflect the fact that the sediment flora and fauna were efficiently bio-processing any nutrient wastes that had been deposited on the sea bed. Communities at 100m N and S of the cages were similar to those at the control, suggesting waste deposition was confined to an area up to 25m from the cages.

Species diversity values for macrofaunal communities at Glinsk are presented in Figure 3.31 (d). Diversity was consistently found to be lowest at mid cage in 1996, 1998 and 2000 (no data was available for 1999), though was also low at 0m N in 1997. The low diversity levels recorded at these sites were found to reflect the abundance of opportunistic species in sediments directly below and adjacent to the cage site. Diversity at the control site was found to fluctuate greatly between years, from 1.58 in 1996 to 3.94 in 1997, with the former value being due to the fact that only one individual of three species were found at the site in 1996. This variability reflects the inconsistent nature of sediment macrofaunal communities in the Glinsk area, as may often be the case with coarse, mobile sediments.

In summary, sediment at Glinsk was found to have low levels of nutrient enrichment compared with other cage sites within the bay. Results of sediment chemistry gave no evidence of high levels of waste feed or faeces accumulation, with levels remaining fairly consistent between survey years. Despite this, some waste accumulation was believed to have occurred in sediments near to the cages, as reflected by the frequent occurrence of opportunistic species in macrofaunal surveys. Any impacts of this nature were restricted to the sediments immediately under the cages, and up to 25m S of the site.

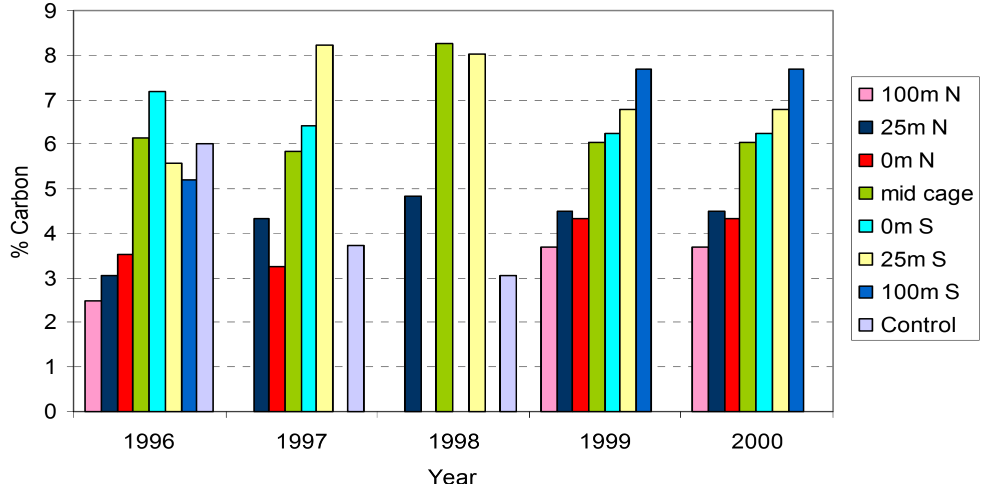

Figure b: % Carbon levels at Glinsk between 1996 and 2000.

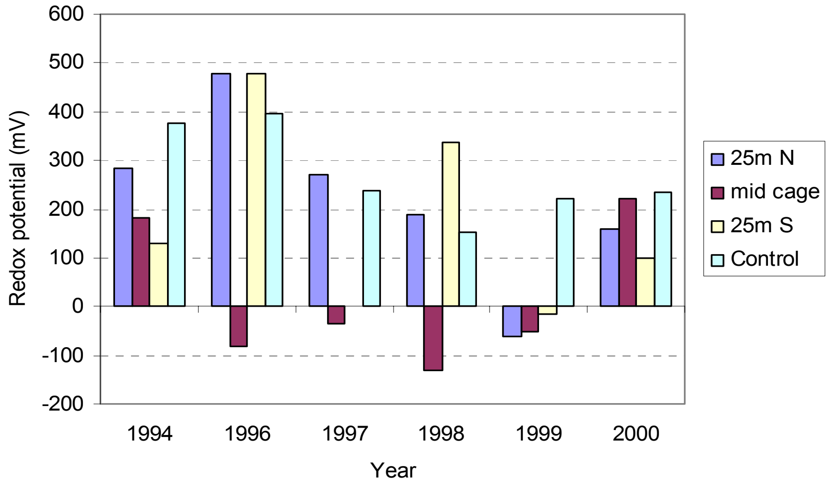

Figure a: Redox potentials at Glinsk between 1994 and 2000.

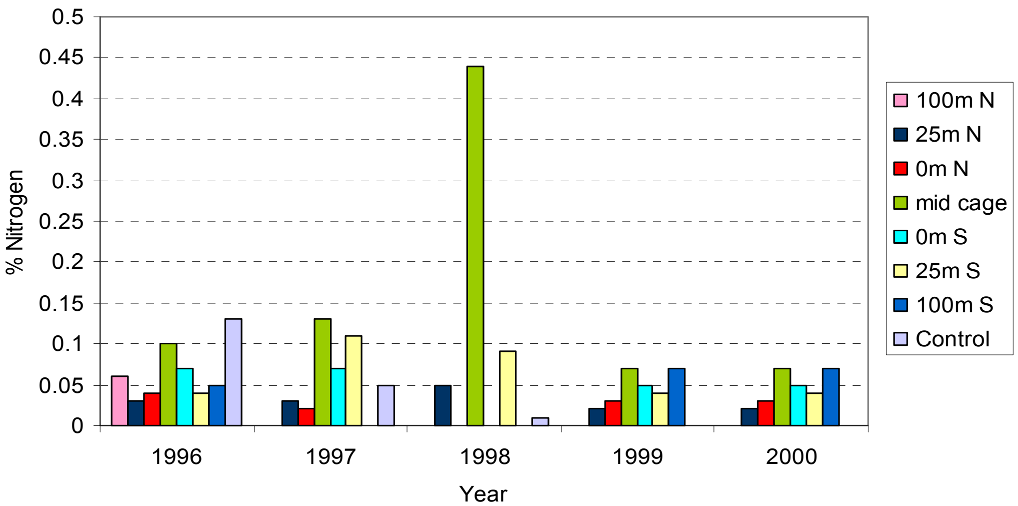

Figure c: % Nitrogen levels at Glinsk between 1996 and 2000.

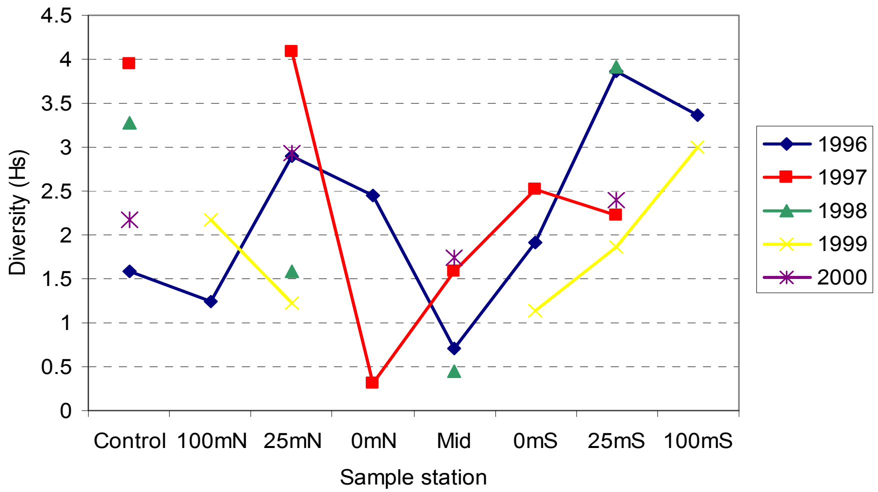

Figure d: Species diversity (Hs) at Glinsk between 1996 and 2000.

Figure 3.31: Chemical and biological properties of sediments at Glinsk.

Millford

Cages at Millford have been sporadically stocked over the past few years, with no fish being present in 1999 - 2000. Sediment at the site has been observed to be of a soft fine nature, with high levels of oxygen depletion being noted at the mid cage site in recent years (Figure 3.32 (a)). Waste feed has frequently been found in sediments under the cages, both during periods of production, and after the cages were fallowed (e.g. in 2000), suggesting breakdown and processing of wastes at the site may be slow (possibly due to sediment anaerobism).

Carbon and Nitrogen levels in sediments at Millford (shown in Figures 3.32 (b) and (c)) have been very high in all survey years between 1994 and 2000, with mean carbon levels in each year ranging from 7.82 % in 1999 to 9.97 % in 2000, and mean nitrogen levels ranging from 0.77% in 1998 to 0.94% in 1995. Values of this kind indicate that sediments at Millford have undergone relatively high levels of nutrient loading throughout the last decade, with little recent sign of improvement, despite cages being empty for the last two years. Peaks in carbon levels occurred at 25m S in 1999 and 2000 (11.33 and 11.68% respectively) despite cages being empty, and coincided with peaks in nitrogen at the same sample location (1.57 % and 1.53% respectively). This suggested that not only were nutrient waste levels remaining high in sediments at Millford due to slow waste processing, but that nutrients may be entering the sediment from external sources. Large amounts of plant debris such as leaf litter have been observed on the sediment surface in recent surveys, suggesting that organic matter from the surrounding land may have been adding to the nutrient load at Millford, possibly bought in by a nearby freshwater source. Carbon and nitrogen levels in control sediments have also traditionally been elevated, though have usually been lower than those recorded near to the cages.

Macrofaunal data has been collected at Millford since 1988. Results of surveys conducted in the late 1980s and early 1990s showed that sediments at the site have traditionally been characterised by some nutrient enrichment, with opportunistic species such as *Capitella* spp. dominating samples taken directly beneath the cages. In 1991, no macrofauna were recorded below the cage block as a result of high sediment anoxia. An improvement was noted in later years, with a return to opportunistic dominance. Sediments around the periphery of the cage site have been characterised by occasional opportunists, as well as species typical of fine, mildly enriched sediments, such as *Pseudopolydora paucibranchia* (often recorded in very high numbers in zones of secondary enrichment).

Macrofaunal data collected from Millford sediments in recent years is presented in the form of ranked species abundances between 1997 and 2000 in Appendix 3 Table 9, and as a plot of species diversity (Shannon-Weiner) in Figure 3.32 (d). As in earlier surveys, macrofaunal species compositions throughout Millford reflected the fine sediment types and surface detritus that were found in the area, with amphipods being highly abundant, notably at the Control site, and at 25 -100m away from the cages. Enrichment indicator species were consistently present at 0m S, 0m N and occasionally at 25m N / S and mid cage. However, contrary to sediment chemistry data, numbers were not indicative of severe enrichment. In 1999 and 2000, a change from Nematoda spp. and *Capitella* spp. as the most abundant taxa was observed at mid cage, with *Mediomastus fragilis* taking over. (This species is common in fine, mildly enriched sediments, though is also known to be associated with high levels of nutrient enrichment (Pearson and Rosenberg, 1987)). Although abundance of all species were low in these years, data would suggest a slight recent improvement in mid cage sediment quality.

Species diversity data at Millford (Figure 3.32 (d)) showed varying trends between sample years, with values being lowest at mid cage and 0m N in 1998 and 1997, at 0m S in 2000, and at 25m S in 1999 (tying in with peaks in nitrogen and carbon at that station in 1999 as shown in Figures 3.32 (b-c)). Diversity levels in all years were similar to those at the control in sediments at, or further than, 25m from the cage edge, suggesting that the highest impacts from nutrient enrichment on macrofauna were confined to the immediate vicinity of the cage site.

In conclusion, recent sediment chemistry data from Millford suggested conditions were poor, with high nitrogen and carbon loads being consistently observed, and redox levels being low at mid cage. No improvement in conditions was observed in 1999 and 2000, despite cages being fallow during these survey years. Despite this, macrofaunal data suggested impacts were not of a severe nature, with diversity scores returning to control sediment levels after 25m from the cage edges. Although opportunistic species were regularly encountered throughout the Millford survey area, abundances were fairly low, and non-indicator species were present alongside. Recent macrofaunal data suggested impacts may have decreased at mid cage since fish stocks were removed in 1998 / 99.

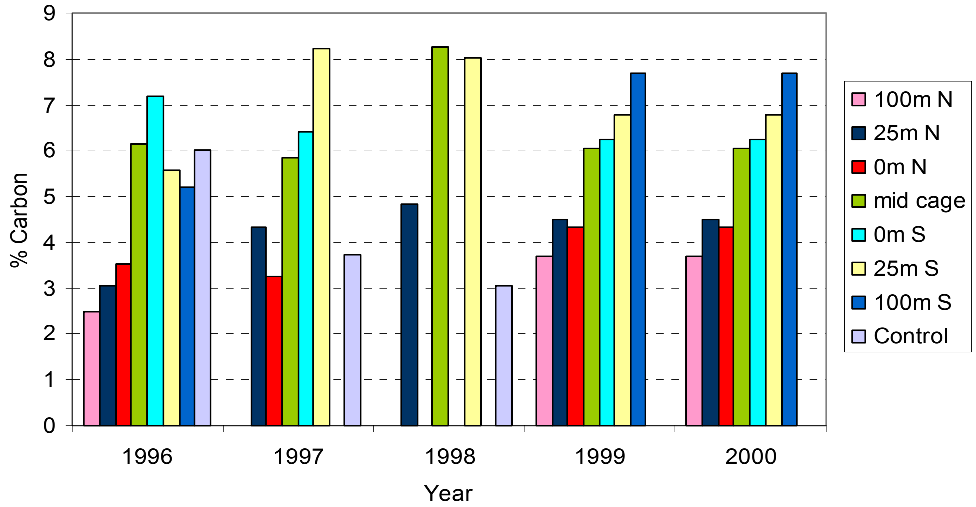

Figure a: % Carbon levels at Millford between 1996 and 2000.

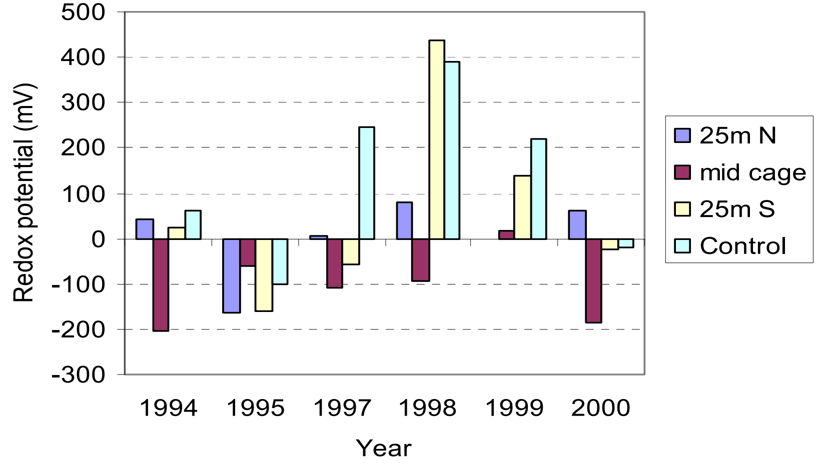

Figure b: Redox potentials at Millford between 1994 and 2000.

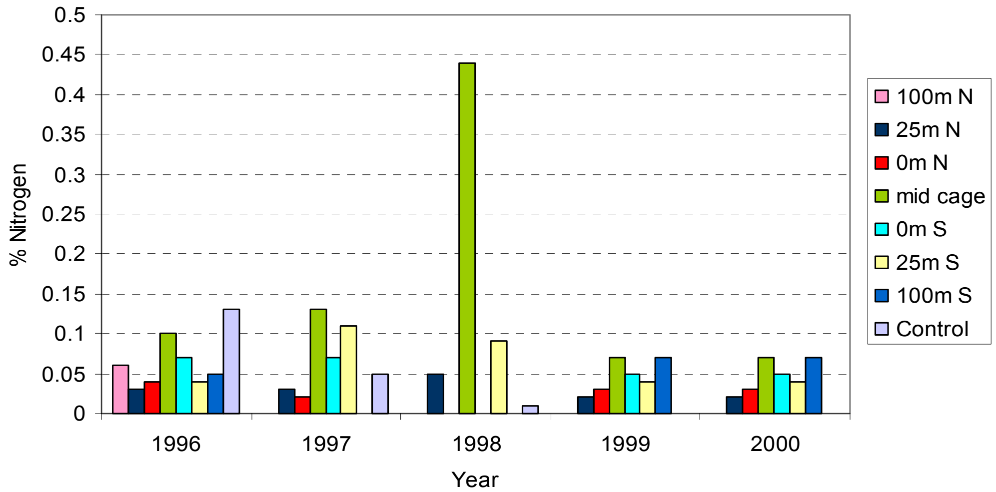

Figure c: % Nitrogen levels at Millford between 1996 and 2000.

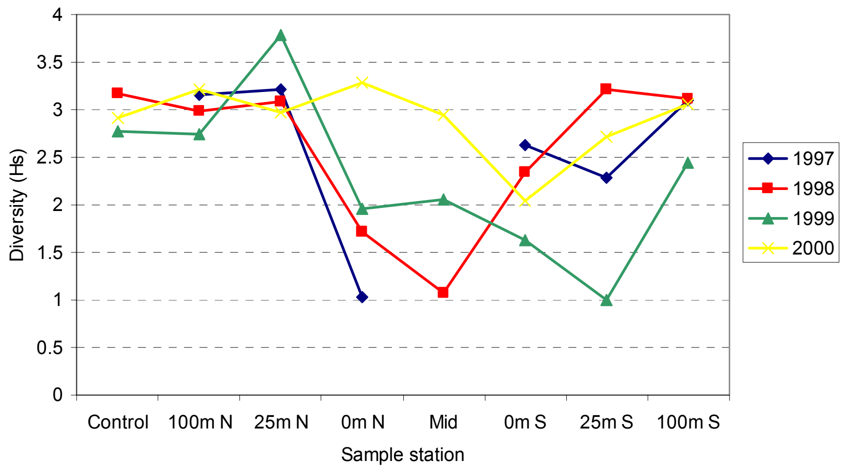

Figure d: Species diversity (Hs) at Millford between 1997 and 2000.

Figure 3.32: Chemical and biological properties of sediments at Millford.

3.3.3 Impact Zones predicted by Macrofauna

The presence or absence of certain sediment macrofaunal species, termed indicator species, is often used to determine areas of impact from nutrient enrichment e.g. from sewage outfalls or fish cages (Pearson and Rosenberg, 1987). Typical indicator species include members of the Capitellidae family (e.g. *Capitella* spp., *Mediomastus fragilis*), certain spionid polychaetes, such as *Malacoceros fuliginosus* and oligochaetes such as *Tubificoides benedenii* (Pearson and Rosenberg, 1987, etc.). Such information can be used to compliment physical and chemical sediment data, and outputs from waste dispersion models (see Chapter 4) in order to aid the interpretation and extent of environmental impacts from point waste sources such as fish cages.

Impact zones, as predicted by the resident macrofauna, at each cage site in 1999 / 2000 (according to most recent survey data) are illustrated in Table 3.12. This uses the presence (+) or absence (-) of macrofaunal opportunistic / indicator species, such as those known to be associated with nutrient accumulation (Pearson and Rosenberg, 1987) to identify areas of enrichment in the sediments around each of the cage sites. The table shows that impacts were confined to the sediments immediately below and adjacent to the cages at Cranford A and Cranford C. Although more widespread effects were noted elsewhere, e.g. opportunistic species were abundant in sediments up to 25m N at Cranford B. In particular, widespread enrichment was noted at Kindrum, Millford and Moross. Zones of high impact were confined to a distance of between 0m and 25m from the cages at these sites, though evidence of some sediment enrichment was found to occur beyond this, including sediments at the control sites, denoted by the abundance of the Capitellidae species, *Mediomastus fragilis*. In all cases, the presence of enrichment tolerant macrofaunal assemblages at the control sites indicated that high background nutrient levels were features of the sediments at Millford, Kindrum and Moross, likely due to low current flows, their deep basin morphometries, and likely tendencies to act as sinks for nutrients from a variety of aquatic and terrestrial sources.

Table 3.12 Impact zones for sediments at each cage site, as indicated by those areas colonised by opportunistic macrofaunal species, in 1999 / 2000.

Site	Year	Control	100m S	25m S	0m S	mid cage	0m N	25m N	100m N
Moross 1	1999	**+ mf 3**	**+ mf 2**	+	+	+	+	+	**+ mf 2**
Moross 2	2000	**+ mf 1**	**-**	**-**	**-**	+	+	+	**+ mf 1**
Cranford A	2000	**-**	**-**	**-**	+	+	**+ mf 1**	**-**	**-**
Cranford B	2000	**-**	**-**	**-**	**+ mf 2**	+	+	+	**-**
Cranford C	2000	**-**	**+ mf 3**	**-**	**-**	+	**-**	**-**	**n.d.**
Millstone	2000	**-**	**-**	**-**	**-**	+	+	+	**-**
Kindrum	2000	**+ mf 2**	**+ mf 1**	**+ mf 1**	**+ mf 1**	+	**n.d**	**n.d.**	**n.d.**
Glinsk	2000	**-**	**n.d.**	+	**n.d.**	+	**n.d.**	**-**	**n.d.**
Millford	2000	**+ mf 2**	**+ tb 2**	**+ cc 3 / tb 4**	**+ mf1**	**+ mf 1 / cc 2**	**+ mf 2**	**+ mf 2**	**+ mf 4**

Key:
+ = the opportunistic species *Capitella* spp. or *Malacoceros fuliginosus* either dominated, or were among the top five most abundant macrofaunal species
+ = other opportunists (*Mediomastus fragilis, Tubificoides bendenii*) either dominated, or were among the top five most abundant macrofaunal species
Where a number appears next to an abbreviation (e.g. mf 3), this refers to the ranked abundance of the species (in this case, *Mediomastus fragilis* was ranked as the third most abundant species).
n.d. = No Data

Species Abbreviations:
mf = *Mediomastus fragilis*
tb = *Tubificoides bendenii*
cc = *Capitella* sp.
Where no abbreviation is used, the dominant macrofaunal species was *Capitella* spp. or *Malacoceros fuliginosus*

3.3.4 Conclusions

The above summaries of sediment chemistry and macrofaunal data at cage sites in Mulroy Bay have shown that sediment quality varied between sites, largely as a function of sediment types and associated current flows. Cage sites located in areas of higher tidal flows (i.e. Millstone and Glinsk) were found to have coarse sediments and lowest sediment nutrient loads (though waste dispersion was sometimes found to occur over a larger area), while cages located in areas of comparatively low tidal flow (i.e. Cranford sites, Millford, Moross and Kindrum) generally had fine, more enriched sediments. Impact zones at all sites, however, were mostly confined to an area of up to 25m from the cage edges, with occasional exceptions being observed following movements of cages to new locations (this was often due to positioning of new sample sites over old cage locations).

Sediment quality over the years for which data is available was consistently observed to be lowest at Kindrum (in terms of both macrofauna and chemistry), where macrofauna was found to be absent or of a strongly opportunistic dominated nature both under, and to the south of the cages. This was partly due, however, to local hydrography, meaning that water exchange near the seabed in the Kindrum area was poor. Control sediments near to Kindrum were found to be nutrient enriched, and were of a similar status to sediments adjacent to the Cranford cage sites (in terms of chemistry and macrofaunal species composition). This indicated that either enrichment was a natural feature of the basin in which the Kindrum cage site is located, where the deep sediments may be naturally anaerobic and thus slow to bioprocess nutrients, or that there has been a consistent build up over time of particulate wastes from fish farming activities causing a cumulative impact throughout the upper bay. This is consistent with the residual water currents described in section 2. Sediments at Millford were also observed to be highly nutrient enriched (as shown in chemistry data), despite fish stocks being absent in recent surveys, suggesting decomposition of nutrients and sediment recovery rates at the site would be low. Again, the Millford site is characterised by low seabed current flows, and may be a site of detritus accumulation from local vegetation and / or stream inputs.

Despite cage waste and nutrient enrichment being a feature of sediments at each salmon production site, the regular repositioning of cages throughout Mulroy Bay was shown (as expected) to have a positive effect on sediment conditions, while also allowing previously impacted sediments to recover. This, along with regular rotation and fallowing of sites, should be encouraged in order to allow environmental recovery to occur between production periods.

4. AQUACULTURE CAPACITY MODELS

The capacity of a system to sustain a function may be calculated in a variety of ways and is dependent on many factors, and as such can be an inexact measure. However, despite its limitations an estimate of capacity can be used as a tool for management of resources, providing it is applied carefully and with expert knowledge.

4.1 The concepts of carrying capacity within the marine environment

The term environmental capacity used here defines the amount of aquaculture production that can be sustained by an environment, within certain defined criteria. Irrespective of environment or method of aquaculture, all capacity models must consider the following:

- What determines the productivity of the environment
- What the farmed organisms consume/produce in terms of food/wastes
- How the environment responds to waste loadings
- How much change is permissible

Marine aquaculture is dependent on resources from the environment such as provision of feed, broodstock or juveniles, space and supply of good quality water. These resources are known as “environmental goods”. Conversely, cage aquaculture is also dependent on the environment to dilute and bioprocess dissolved and solid wastes produced. These are “environmental services” (see Figure 4.1). The ability of the environment to sustain these goods and services without undue detriment is referred to as the environmental capacity. This overall process is highly complex with many variables, and thus difficult to calculate and implement as an overall management tool.

A more common method of assessing environmental capacity for environmental management is to investigate a more discrete locality, such as an enclosed system or embayment. Here the capacity can be assessed using individual or multiple factors which are required to sustain aquaculture, such as good water quality, and therefore depending on its nature will set limits on levels production.

Two methods of estimating the capacity of Mulroy Bay for aquaculture have been used during this study. The first is the calculation of the availability of food for estimating the sustainability of mussel and shellfish farming within the bay. The second is the estimation of the oxygen demand of the system from physico-chemical processes arising from natural and anthropogenic nutrient inputs.

The investigation of food supply for the shellfish has been accomplished previously for Mulroy Bay (C-Mar, 2000). However, this only allowed an estimation of sustainable levels of mussels, scallops and oysters in the bay, but not fish production in farms. Food supply to shellfish is reported in section 4.3.1.

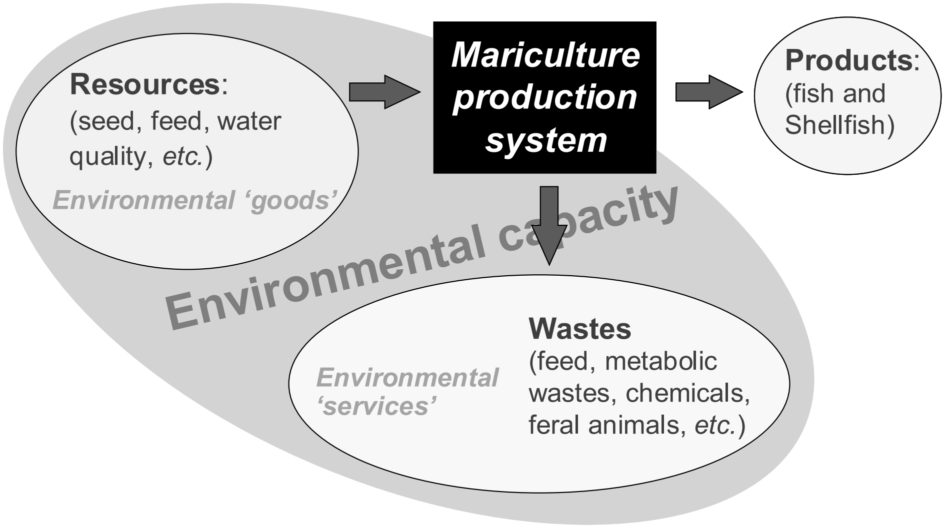

Figure 4.1 The concepts of Environmental Goods and Services, and Environmental Capacity as related to Mariculture.

Estimation of the oxygen budget and demand for the Bay is dependent on the demand from two compartments – water and sediments:

Oxygen demand by the water column is largely dependent on the microflora which contributes to an oxygen equilibrium by photosynthesis (net provider of oxygen during the day) and respiration (net user of oxygen during the night). Estimation can be a complex process and controlled by physical factors, such as oxygen exchange with the atmosphere (dependent on surface area of the lough, exchange due to wind action etc), productivity of the water column (dependent on nutrient input and utilisation by plankton and bacteria). The nutrients can come from many sources around the lough – farm run-off, fish farms, and sewage. Some of these may be measured, others modelled but others must be included as an assumption due to lack of information.

Oxygen demand by the sediments is often considered less important than for the water column. However, sediments are the primary sink for particulate nutrients within marine systems and will therefore be subject to considerable bacterial breakdown. As this breakdown increases, due to higher nutrient inputs, bacterial oxygen demand will become greater and eventually sediment may be anoxic, with a complete equilibrium shift from oxygenation to reduction processes. The later also requires an oxygen demand to allow the equilibrium to shift back. A key to estimating the sediment oxygen demand is amount of particulate nutrients entering the system. This may be measured or modelled. Measurement is difficult in a large system as there is great variability due to natural and anthropogenic causes. Information on inputs of the sources have to be obtained. One of the main inputs of particulate nutrients is from the fish cages within Mulroy Bay. This can be calculated based on the amount of food used in the production process and the faecal production. However, the nutrient waste from the fish farms is dispersed in the currents, with the bulk settling beneath the cages. The gradient of settlement is variable from the cages and thus the sediment oxygen demand may be different depending on the dispersion factor of the fish farms site.

4.2 Models of the distribution of particulate waste material from fish cages

The loading area of sediment around a fish cage is an important factor in determining the impacts of waste accumulation, and is dependent on the quantity of waste released, water depth, current velocity and direction, and the settling rate of waste particles. Where wastes are readily dispersed and low levels of loading occur over a wide area, or where inputs do not exceed the carrying capacity of the sediment, adverse effects will be slight. In some cases, low levels of extra nutrients provide additional food for sediment infauna, leading to increased abundance and diversity. On the other hand, if high loading occurs in a small area then impacts are likely to be higher, but less widespread.

The predicted dispersion of this material can be determined by using a waste dispersion model that is currently under development at the Institute of Aquaculture, Stirling University. This uses the equation of Gowan *et al*. (1988) for estimating horizontal dispersal of uneaten feed and faeces:

$$D = \frac{dV}{v} \qquad \text{.................... (1)}$$

where: D = horizontal distance dispersed (m)
d = water depth (m)
V = current velocity (ms^{-1})
v = settling velocity of wastes.

This equation assumes that:

4 there is no re-suspension or redistribution of settled wastes,
5 current speed and direction are uniform with depth, and
6 the depth under the cage is constant.

Additionally, the models do not allow for removal of waste by uplift systems in use at many of the fish cages in Mulroy Bay.

Two versions of the model exist; one based on spreadsheet calculations and operated within a Microsoft Excel™ workbook, the second undertakes the same basic calculations within a GIS environment using the program IDRISI™. The latter may calculate the dispersal of material in actual directions using recorded current data than arc sectors as described above.

The spreadsheet model estimates dispersal from individual cages within a block in 10m intervals for 200m along eight compass axes (North, North East, East, South East, South, South West, West, North West). Water flow data collected over at least half a lunar cycle is used for estimating the amount of waste dispersed along each compass direction. The amount of waste falling into each sector defined by the arc subtended at the dispersal point with 10 m length is calculated. The amount of particulate material entering the sections is corrected by assuming that currents will distribute the waste normally around the sectors, thus the loading of sediment at each distance from the cage is weighted by relative probability (equation 2):

$$Z = \frac{I}{\sigma\sqrt{2\pi}} e^{1/2((Y-u)/\sigma)^2} \qquad \text{………… (2)}$$

where: Z = relative probability
σ = sample deviation
Y = distance
u = sample mean

The outputs from the model are both numerical and graphical, the latter being the most descriptive way to present the results. The method of plotting the output varies between the form of the model used.

The spreadsheet model
The output from the spreadsheet model is given as grid table, with squares being of 10 m resolution. The grid can be plotted as contour using the graph functions of the spreadsheet or more effectively by importing the grid file into the spatial interpretation program SURFER^{TM}.

The GIS based model
The results are presented using the contour plotting functions of IDRISI (Perez *et al*, 2001). These use the interpolation functions to smooth the data trends to give even plots of the input of organic carbon into sediments. The GIS version of the model uses filtering functions of GIS to allow for re-suspension and re-distribution of settled material, based on empirical data (Chen *et al.*, 2000), to be taken into account. Validation experiments have shown that this model has a percentage error of 7 to 40% for Atlantic salmon culture.

Spatial dispersal models which calculate sediment loading for organic carbon were calculated for each site within Broadwater, Northwater and the Narrows. Hydrographic and bathymetric data collected and represented in Chapter 2 was used in conjunction of production data provided by Marine Harvest Ireland Ltd for salmon production at each site in 2001. No hydrographic data was available for Milford so modelling could not be done.
The model outputs, plotted as contour diagram within SURFER^{TM} are give in Figures 4.2 to 4.9.

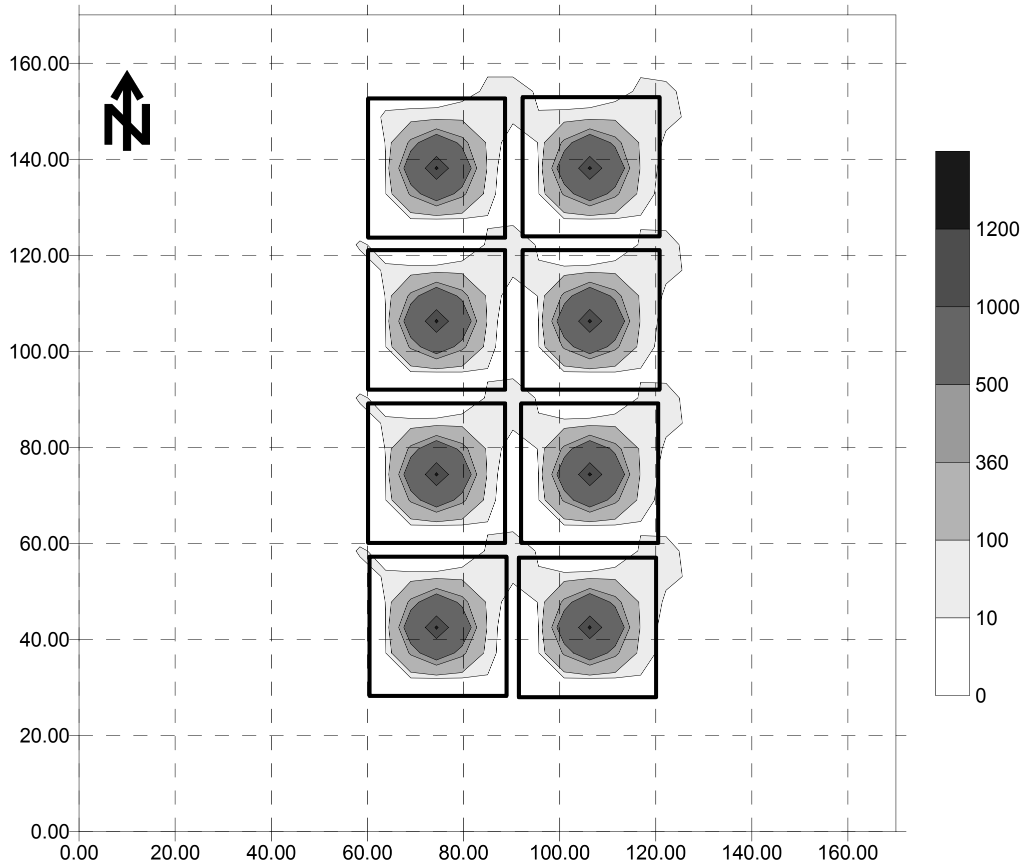

Figure 4.2 *Model of the spatial distribution of organic carbon to the sediments ($gC/m^2/yr$) for the Kindrum fish farm.*

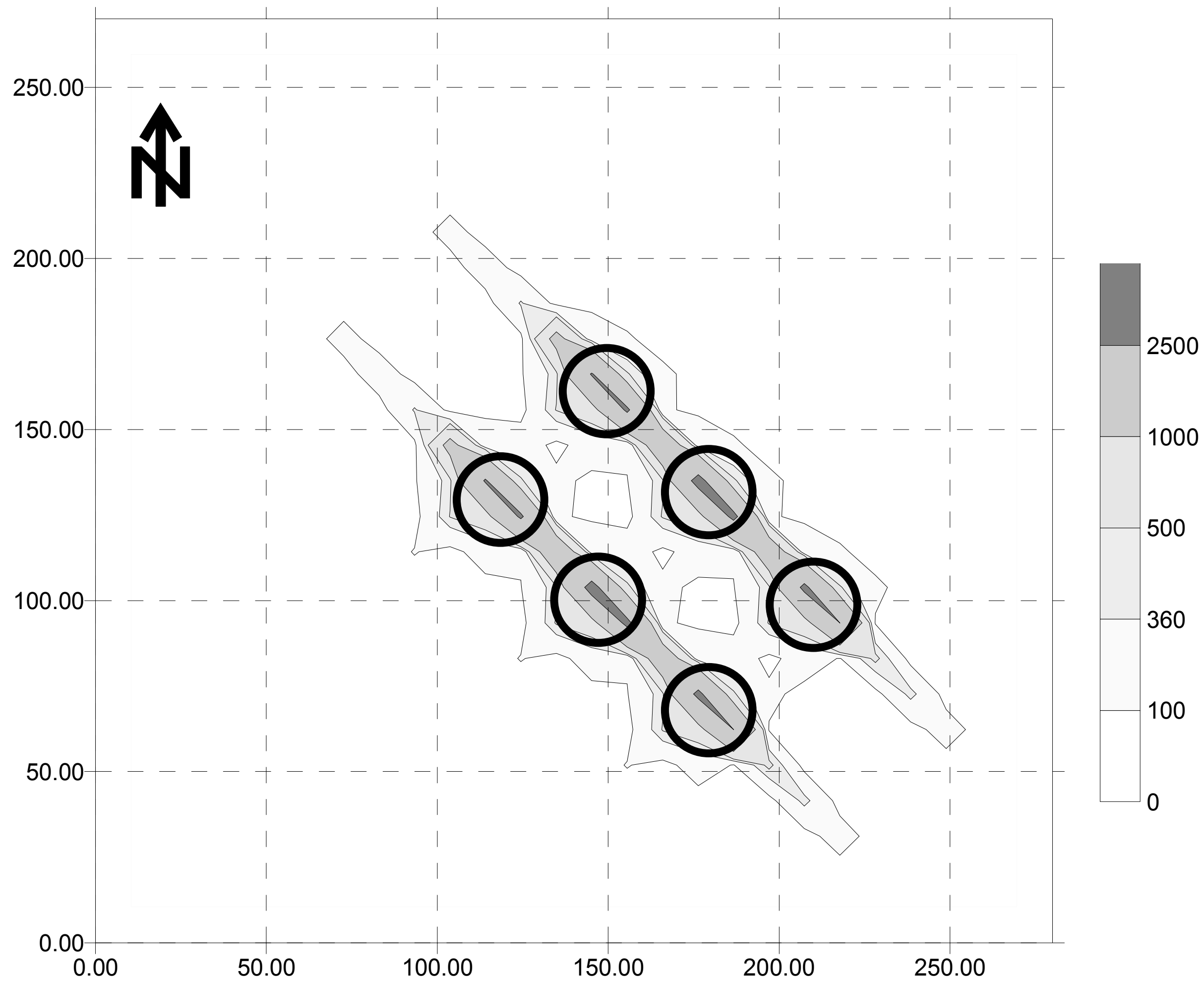

Figure 4.3 *Model of the spatial distribution of organic carbon to the sediments ($gC/m^2/yr$) for the Moross 1 fish farm. Axis units are in metres.*

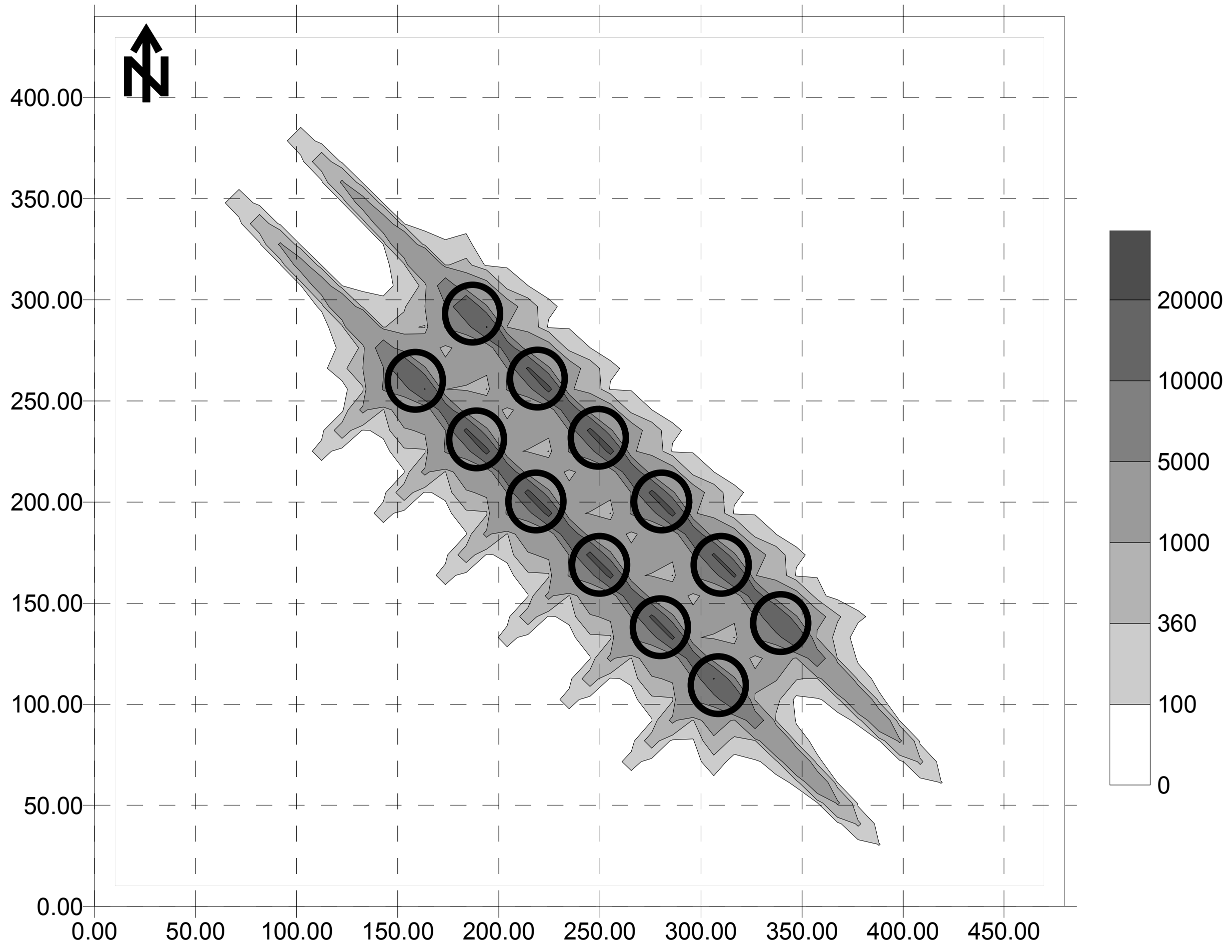

Figure 4.4 Model of the spatial distribution of organic carbon to the sediments ($gC/m^2/yr$) for the Moross 2 fish farm. Axis units are in metres.

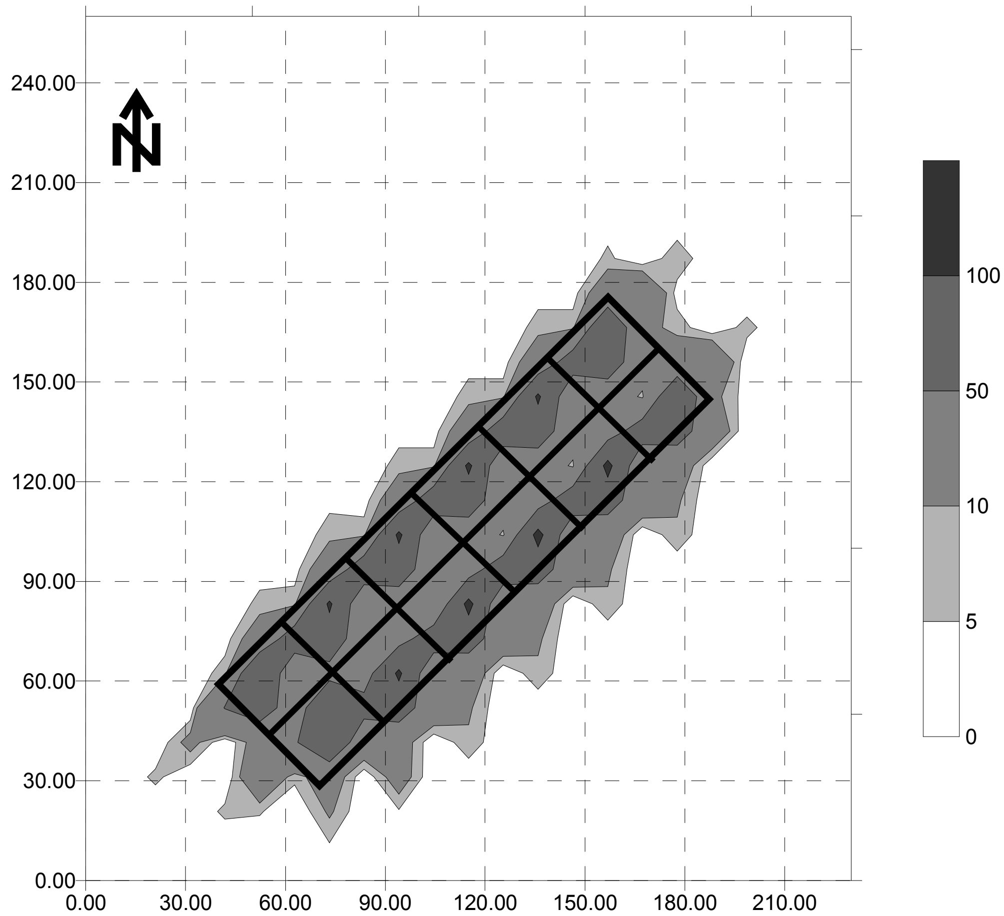

Figure 4.5 *Model of the spatial distribution of organic carbon to the sediments ($gC/m^2/yr$) for the Cranford B fish farm. Axis units are in metres.*

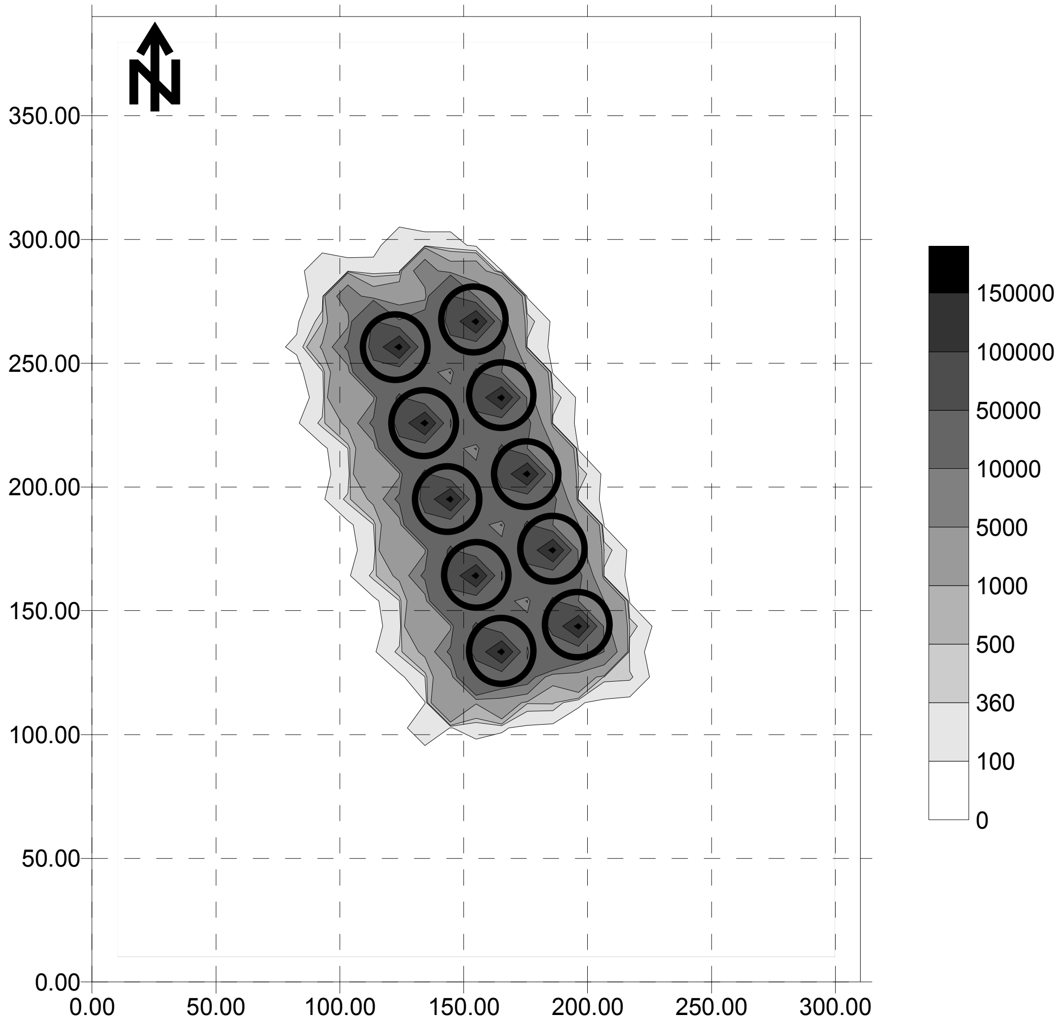

Figure 4.6 Model of the spatial distribution of organic carbon to the sediments ($gC/m^2/yr$) for the Cranford A fish farm. Axis units are in metres.

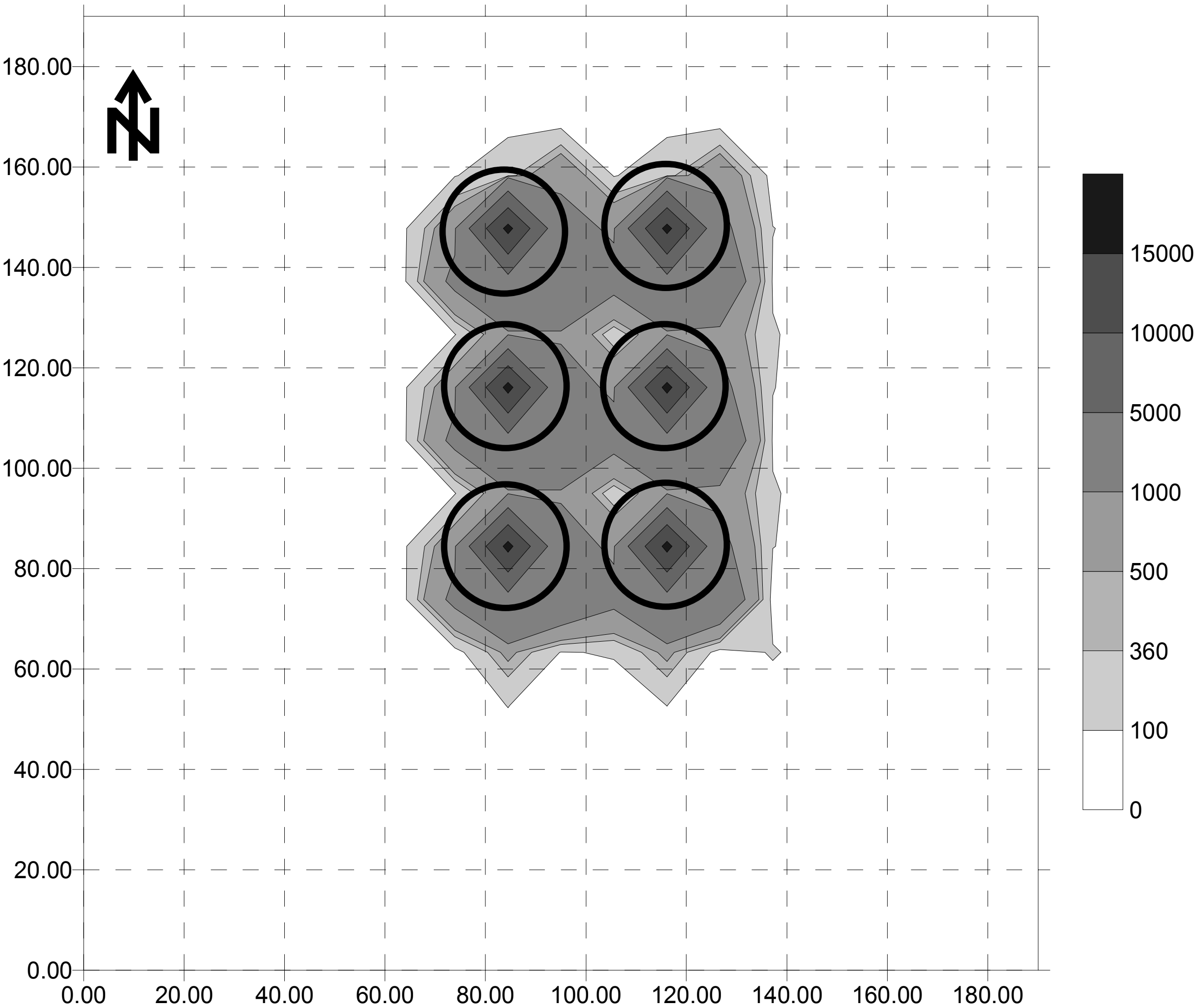

Figure 4.7 *Model of the spatial distribution of organic carbon to the sediments ($gC/m^2/yr$) for the Cranford C fish farm. Axis units are in metres.*

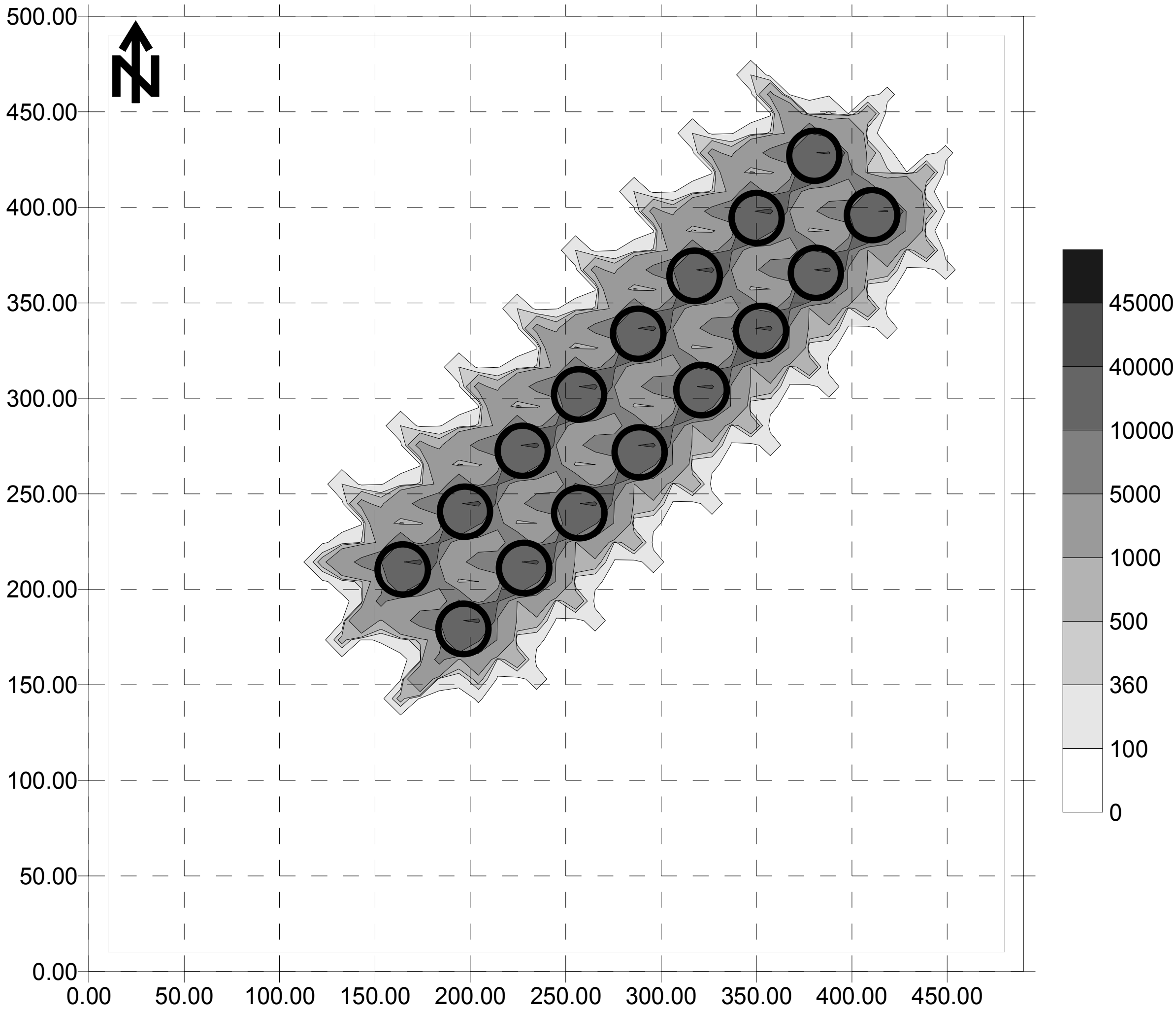

Figure 4.8 *Model of the spatial distribution of organic carbon to the sediments ($gC/m^2/yr$) for the Millstone fish farm. Axis units are in metres.*

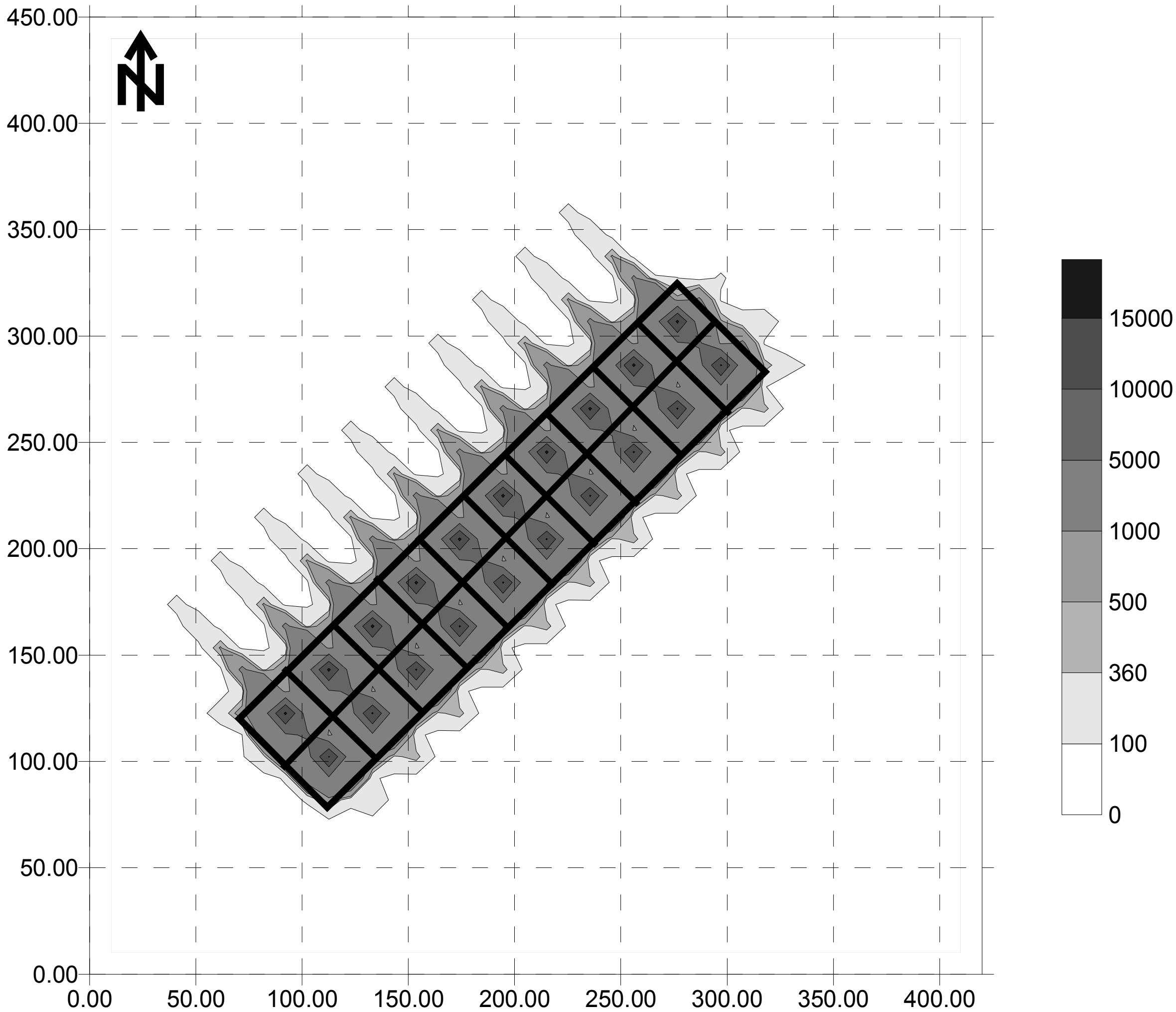

Figure 4.9 Model of the spatial distribution of organic carbon to the sediments (gC/m²/yr) for the Glinsk fish farm. Axis units are in metres.

These models indicate the spread of the particulate waste from the cages in terms of distance and amount. This version of the model tends to overestimate build-up of material beneath and adjacent to the cages, which will provide a "worse case" situation. The build up of organic material within the sediments will increase the bacterial activity thus increasing the oxygen demand, leading to changes in the benthic community structure. The increase in the sediment oxygen demand will, in turn, provide additional stress on the oxygen demand of the system and thus use up some of the oxygen capacity. Evidence from Britain and extrapolation of data from the Mediterranean (Beveridge, 1996) suggests that there is oxygen depletion in sediments where the input is greater than 1 gC/m^2/d, which approximates to 360 gC/m^2/yr.

From the modelled outputs SURFER can be used to calculate the planar area (m^2) at any contour level. This will estimate the area of seabed which will incur increased oxygen depletion, the "footprint" of effect. The areas accounted for this contour are given in Table 4.1.

Site	Planar area (m^2)
Broadwater	
Cranford A	16383
Cranford B	6310
Cranford C	6310
Total	*29003*
Northwater	
Kindrum	1424
Moross 1	5263
Moross 2	28897
Total	*35584*
Narrows	
Glinsk	24413
Millstone	40159
Total	*64572*

Table 4.1 Planar areas (m^2) for the 360 gC/m^2/yr contour for each fish farm site modelled.

The planar area for the 360 gC/m^2/yr contour ("footprint" of significant sediment oxygen demand) as given by the models are dependent on a combination of Food Conversion Ratio (FCR), hydrography, size of the cages and production at each site. For example, the large production sites within Mulroy Bay narrows (Millstone and Glinsk) are have large footprints and thus high sediment oxygen demands. Smaller sites where production is low will have smaller footprint and thus less absolute sediment oxygen demand. Cranford B has no significant demand (deposition is greater than 1gC/m^2/d) due to the low levels of fish production at this site during the modelling period.

4.3 Estimation of aquaculture capacity of Mulroy Bay

Two methods have been used for the estimation of the capacity for aquaculture within a bay system (C-Mar, 2000); the capacity for the two essential requirements for the cultured resource, food and oxygen.

4.3.1 <u>Capacity for shellfish culture within Mulroy Bay (food availability)</u>

For shellfish, Carver and Mallet (1990) consider aquaculture capacity as the maximum production level which can be achieved without negative impact on growth. This was defined in terms of the available food supply. To achieve sustainability within the system, it has been recommended that up to 50% of the available food supply be used for aquaculture (Rodhouse and Roden, 1987).

The study by C-Mar (2000) estimated the availability of food for shellfish culture within Mulroy Bay. This was based on values for primary production (carbon fixation, chlorophyll a and phytoplankton carbon biomass), particulate organic matter (POM), volume of water (tidal exchange) and shellfish filtration and clearance rates. The study showed that the total daily POM within the system was estimated as 71.9 tonnes of which 47.9 tonnes was attributable to phytoplankton carbon fixed trough primary production. Models indicated that 47.5 tonnes of available POM was available to grazing by mussels and 24.5 tonnes by scallops. Clearance rates, taken from literature sources (Bayne and Widdows, 1978; Okumus and Stirling, 1994; Bayne et al, 1976), suggested that cultivated mussels were consuming between 6.0 and 5.3 t/d and scallops between 0.6 and 1.7 t/d. Therefore food consumption by the cultured bivalves within Mulroy Bay was between 19% and 48% of the available POM when allowing for additional use due to bivalve seed grazing pressure.

4.3.2 Capacity for fish culture within Mulroy Bay (oxygen availability)

The balance between oxygen demand and oxygen production within an embayment is important for organisms living within water and sediments. Aquaculture is a net user of oxygen dissolved within the water column, though it is important for the continuance of aquaculture that an adequate supply of oxygen is available. The oxygen demand of Mulroy Bay was calculated on the basis of sediment oxygen demand, demand due to aquaculture and natural oxygen demand through bacterial breakdown. This can be offset against the oxygen production within the system, due to phytoplankton and macroalgae.

Sediment oxygen demand is calculated based on the "footprint of effect" for each cage block given in Chapter 2. This is based on the input of particulate waste material over a year's production of fish (Marine Harvest Ireland, data). There is no information available for shellfish farming. The value used for sediment demand within the "footprint of effect" was that calculated by C-Mar (2000) of 594 mg $0_2/m^2$/hr. This value is considered high when compared to the literature, as Kupta-Hansen et al (1991) for levels of 360 mg $0_2/m^2$/hr beneath salmon cages, where as Kempf et al (2002) found 214 mg $0_2/m^2$/hr. However, this is used as a "worst case" in these circumstances. The value of sediment demand used outwith the estimated "footprint of effect" is 90 mg $0_2/m^2$/hr (Kupta-Hansen, 1991). Again this is the highest value given in the literature for this type of sediment, allowing a safety margin for the calculation of sediment oxygen demand for Mulroy Bay. The value obtained by C-Mar (2000) of 247 mg $0_2/m^2$/hr, and suggested as "background" was not used here. There are no locations given for the sampling position, and comparison with literature values, even within the summer, shows that this is between 2.5 to 10 times higher than for unaffected sediments of a similar type (Kempf et al, 2002, Kupta-Hansen, 1991, Hargrave, 1994). It is suggested therefore that these samples were taken in areas influenced by aquaculture production.

The sediment oxygen demand for the footprint of effect areas and the unaffected areas within the regions of Mulroy Bay are given in Table 4.2.

Region	Total Area (m^2)	"Footprint" (m^2)	Unaffected area (m^2)	"Footprint" SOD (kg O_2/d)	"Unaffected" SOD (kg O_2/d)	Total SOD (kg O_2/d)
Broadwater	19671653	29003	19642650	413.7	42428.1	42842.8
Northwater	5255609	35584	5220025	507.5	11275.2	11782.8
Narrows	17240178	64572	17175606	921.0	37099.3	38020.3
Total						92644.9

Table 4.2 Sediment oxygen demand (SOD) calculated from modelled areas of effect and non-effect from fish farms for regions within Mulroy Bay. Data based on production data for 2001 (Marine Harvest Ireland Ltd).

Oxygen consumption due to aquaculture production was based on fish production figures for 2001 (Marine Harvest Ireland, data) and approximate shellfish production biomass (C-Mar, 2000). Values for oxygen use by the cultured organisms are those given by Wheaton (1977) as 2.60 g O_2/kg fish/d and 1.61 g O_2/kg shellfish/d. Calculation of the oxygen consumption values are given in Table 4.3.

Region	Total (kg O_2/d)
Broadwater	3982.876
Northwater	1625.333
Narrows	2946.055
Total consumption	8554.264

Table 4.3 Oxygen consumption due to the cultured fish and shellfish within regions of Mulroy Bay using fish production data for 2001 (Marine Harvest Ireland Ltd) and shellfish production data from C-Mar (2000)

Phytoplankton production is based on values from light and dark bottle experiments (C-Mar, 2000) for waters within Mulroy Bay. The samples were taken during the summer months for Broadwater and Northwater regions. Data for the Narrows is not available but the value for Broadwater used in this instance. Oxygen production in the light bottle represents production of oxygen through phytoplankton photosynthesis and loss through respiration of both algae and bacteria. The loss of oxygen in the dark bottle gives the values of respiration. By using the difference in values the net oxygen production due to plankton may be calculated taking into account consumption of oxygen by phytoplankton and bacteria (biochemical oxygen demand of the waters). The oxygen production from phytoplankton is shown in Table 4.4.

Region	Area (m^2)	Volume (m^3)	Net O_2 prod ($mg\ O_2/L/d$)	O_2 prod ($kg\ O_2/d$)
Broadwater	19671653	59014959	1.09	64326.31
Northwater	5255609	15766827	0.71	11147.15
Narrows	17240178	51720534	1.09	56375.38
Totals				131848.84

Table 4.4 *Oxygen production for each area of Mulroy Bay due to phytoplankton. Net production uses values from Mulroy Bay during summer 1999 (C-Mar, 2000). Volume is calculated assuming primary production is concentrated in the surface 3 m of water.*

Additional oxygen production from macroalgae used levels estimated for summer 1999 (C-Mar, 2000) for Broadwater and Northwater.

The overall oxygen budget for Mulroy Bay is given in Table 4.5. This shows the balance between oxygen consumption due to the cultured resources – fish and shellfish – within the difference regions and the oxygen evolution due to primary production.

Region	Sediment Oxygen Demand (Kg O_2/day)	Aquaculture O_2 consumption (Kg O_2/day)	Phytoplankton O_2 production (Kg O_2/day)	Macroalgal O_2 production (Kg O_2/day)	Overall budget (Kg O_2/day)
Broadwater	42841.80	3982.88	64326.31		
Northwater	11782.80	1625.33	11147.15		
Narrows	38020.31	2946.05	56375.38		
Totals	92644.91	8554.26	131848.83	4948	**+ 35597.66**

Table 4.5 *Oxygen budget for Mulroy Bay, based on data provided by Marine Harvest Ireland and C-Mar (2000).*

These figures make several assumptions in that they only allow for consumption through aquaculture and normal BOD and that production of oxygen comes from primary production. The latter is calculated for summer when levels are likely to be highest. The higher levels of BOD from terrestrial discharges have not been taken into consideration due to lack of substantiated data from agriculture and sewage. For example, figures from Donegal Council indicate that sewage discharge adds 1320 person equivalents (PE) to the Mulroy Bay system. However, there no data been made available as to the BOD levels from which this calculation was made, or any seasonal variability due to influx of tourists during the summer months. The area of highest level of discharge (1000 PE) is in the Narrows at Carrickart, whereas lower amounts are discharged into the Broadwater (200 PE untreated and 100 PE secondary treated) from Carrowkeel. Further data is required to enable accurate incorporation into the oxygen budget of the system. However, substantive oxygen demand from consistent inputs of nutrients from agriculture/forestry etc, is allowed for in the bacteria respiration in the light and dark bottle analysis of oxygen production (C-Mar, 2002).

The calculations also assume no additional oxygen replenishment from tidal renewal of water within the bay or from oxygen transfer due to wind acting on the water surface. Water exchange is calculated in Chapter 2, based on collected hydrographic data, and indicates that conditions within Mulroy Bay range from highly dynamic water flow in the Narrows (1.74 tides to exchange water) to slow water exchange in Northwater (8.87 tides to exchange water with the ocean). This would suggest good oxygen replenishment within the Narrows where water is renewed rapidly, but slower exchange within the Broadwater and the Northwater. Oxygen exchange due to wind may also be significant. C-Mar (2000) calculated that an additional 92213 kg O_2/d would be produced by a consistent wind of 5.25 m/s, blowing over the water surface of Northwater and Broadwater.

Additionally, the modelled planar areas or “footprints” of carbon deposition do not allow for use of modern aquatic feeding technology, such as uplift systems, presently used on most of the fish cage sites within the bay. These limit the amount of waste falling to the seabed.

These figures for capacity then can only be seen as a rough estimate of the oxygen balance within the Mulroy Bay system and thus enable comparison of the present levels of fish production to the capacity of the system. The figures for production of oxygen use data for primary productivity over the summer when production will be highest , suggesting that the overall budget will be lower in the winter. However, this is offset against lower temperatures and higher oxygen solubility within the water column. These figures should not be used in isolation and be confirmed by results of environmental quality and impact from monitoring studies given in Chapter 3.

The levels given in Table 4.5 indicated an overall surplus within the Mulroy Bay oxygen budget, primarily due to the phytoplankton production in Broadwater and the Narrows. Northwater had an overall negative oxygen demand, due to the relatively smaller area available for phytoplankton O_2 production. However, the sediment demand calculation is indicative of the high levels of background nutrients found in sediments of the upper Northwater basin (illustrated by faunistic data and sediment chemistry, see Chapter 3) and the long retention time.

5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Introduction

Mulroy Bay in Co. Donegal is one of the most intensively farmed aquatic areas in Ireland. Its enclosed nature has lead to concern that the capacity of the environment to support aquaculture has been exceeded and associated environmental degradation has occurred. This study was commissioned:

- to assess the state of the environment, within Mulroy Bay, at existing levels of aquaculture production, and

- to investigate the aquaculture capacity of the system in relation to present levels of production.

Mulroy Bay is a complex, enclosed marine system consisting of a number of deep basins separated by narrow sills. It is divided into three main areas, each with its own distinct hydrodynamic and bathymetric character, leading to different impacts on the environment from aquaculture production.

The Institute of Aquaculture of the University of Stirling, UK, has been monitoring water and sediment quality within Mulroy Bay since 1986. These data, along with those provided by Marine Harvest Ireland (the only fish farming company using the bay) and a recent report by C-Mar (2000) have been used to define the present environmental quality of the system. Hydrographic data collected by the Institute of Aquaculture have been used to model particulate waste dispersion from each fish cage in order to estimate a realistic sediment oxygen demand. This, along with data derived from experimental studies (C-Mar, 2000) has been used to compare present levels of shellfish and finfish production to carrying capacity in terms of oxygen budget and food availability.

5.2 Environmental quality within Mulroy Bay

5.2.1 Water quality

Hydrographic measurements suggest that the different bays within the system have different hydrographic characteristics. The Narrows are shallow and have very fast moving waters, and are thus highly dispersive, allowing high dilution of any aquaculture waste. Conditions within Broadwater and Northwater are deeper with considerably slower currents, relying on volume of water for dilution, making it likely that both soluble and particulate wastes will be more persistent around fish cages and mussel lines. This is confirmed by the calculated flushing times for the different areas (Chapter 2). Broadwater and Northwater take between 2 and 2.5 times longer to flush than the Narrows or allowing for complete exchange with the ocean approximately 3 to 4 times the flushing time of the Narrows.

Comparisons between soluble nitrogenous levels and between plankton productivity (as chlorophyll ‘a’ levels) within the bay between 1991 and 2000 show considerable variation in measurements but no overall increase in nutrients and primary production with time, even though levels of aquaculture have increased. Annual production of mussels has increased from 20 tonnes in the early 1990s to 800 tonnes in 1999. In 1999 Atlantic salmon had a total production of approximately 1200 tonnes, consented by the Department of the Marine, at 8 sites, increasing it from 220 tonnes consented in 1994.

Periodically, however, levels of soluble nutrients have been elevated in the inner parts of Broadwater and Northwater, indicating potential for build-up in the slowest flushing areas of the bays.

Comparison of nitrate concentrations for coastal waters, defined by the Marine Institute (1999) show the levels recorded within the bay are considered "low" (i.e. have never exceeded 200 μg-N/L). See Figures 3.14 and 3.15. It is notable that in the period 1997 to 2001 nitrate levels increased during the winter of 1997/98 at all sites and during the summer of 2000 at the Millstone site. However, average levels have stabilised between 25 μg-N/L and 75 μg-N/L between Jan 1999 and present. Therefore levels of nitrogenous nutrients are not apparently detrimental to the environment within Mulroy Bay, at the present levels of aquaculture production.

In addition, values for biochemical oxygen demand (BOD) measured by C-Mar (2000) between June and September 1999 showed that, though there was some variability, all sites were classified as very clean (<1 mg/L) to fairly clean (<3 mg/L) (after Stirling, 1985; Royal Commission on Sewage Disposal, 1913) again indicating that water quality within the bay is acceptable at present aquaculture production levels.

5.2.2 Sediment quality

Sediment quality can be characterised by chemical and biological factors and results for these are given in detail for each fish farm site between 1997 and 2000 in Chapter 3. These results show clearly that sediment quality varies considerably in close proximity to the fish farm sites and also between sites, largely as a function of the sediment type and associated current flows. Sediments near to dispersive sites within the Narrows (Millstone and Glinsk) have the lowest measured sediment nutrient loading whereas the seabed near to cages in lower tidal flows within Broadwater or Northwater have higher loadings. Impact zones defined by both nutrient load and macrofaunal communities within sediments were largely confined within 25 m of the cages (see Table 3.12), which is considered as an acceptable zone of effect for fish cages by the Scottish Environmental Protection Agency (SEPA).

Sediment quality was consistently observed to be lowest at Kindrum using both macrofauna and chemistry as indicators. This is probably due to the poor dispersion of waste at this site and the length of time fish have been grown (this is the oldest site within Mulroy Bay). The background levels indicated by the reference sampling station in the area shows that the sediments here are depleted within this basin in the upper Northwater, possibly due to prolonged use for aquaculture, and would be slow to recover. Also, Milford fish production site in the southern Broadwater also has high sediment loading despite fish stocks being absent for some time. This may be

contributed to by terrestrial inputs of nutrients from a nearby river. The results suggest that these sites should be monitored carefully for water and sediment quality.
The movement of fish cages within the consented areas appears to have had a positive effect on sediment quality at most cages, particularly in mid-Broadwater. This along with regular rotation and fallowing between production periods has allowed the sediment within the areas farmed for fish to be acceptable.

There has been little work investigating the impacts of mussel farming on the sediments within Mulroy Bay. However, the reference sampling station for the Cranford fish farm sites, in Broadwater, was situated in a area where mussel farming has increased considerably since 1997. The results indicate no change in conditions within the sediment over this time.

5.3 Capacity for aquaculture within Mulroy Bay

Two types of aquaculture are prevalent within Mulroy Bay; finfish and shellfish. The carrying capacity for aquaculture, in terms of production tonnage, within the system is defined as the ability of the environment to support these activities. Two methods of estimating this were used; the calculation of food availability for sustainability of shellfish culture and, the estimation of the oxygen budget of the system to ensure the sustainability of both finfish and shellfish farming at current levels.

Calculations for the food availability were done using empirical data derived from the literature and field experiments (C-Mar, 2000). This was calculated for a “worst case” condition and showed that between 19% and 48% of the available food (71.9 t POM/d) would be utilised by the present level of shellfish culture within the bay. Carver and Mallet (1990) indicated that for sustainability of a system no more than 50% of the available food should be utilised by mussel culture. Thus, shellfish farming may be considered, at worst, near to its capacity. Due to the variability in the results, further work should be undertaken to confirm these figures.

Oxygen consumption and oxygen production within the bay was calculated using Sediment Oxygen Demand (SOD), aquaculture consumption, and phytoplankton and macroalgal production balanced against natural demand from bacterial respiration from total aquaculture and terrestrial nutrient inputs at time of sampling (see Chapter 4). SOD was estimated using modelled “footprints” of the zones around the fish cages where levels of organic input is likely to lead to elevated levels. The demand was then calculated for these and the overall area for each region using values of sediment oxygen consumption for affected and unaffected sediments taken from the literature.

Overall oxygen budget was calculated as giving an excess of 35,598 kg O_2/day. This used annual aquaculture production levels but summer figures for plankton productivity, thus giving a slight overestimation. The calculation, however, does not allow for water exchange through tides and oxygen transfer with the atmosphere due to wind aeration, calculated to be 92213 kg O_2/day for a wind of 5.25 m/s (C-Mar, 2000). In addition, the calculations only allow for demand from bacterial respiration from overall water nutrients but do not allow for seasonal variability in, for example, sewage input. More data is required for this to be included.

The calculated oxygen excess is low, but allowing for additional inputs through water and wind flows the calculations suggest that aquaculture at present levels are sustainable within the Mulroy Bay system. Different areas are affected by currents more than others, therefore the potential for further aquaculture should be limited to the more dynamic areas such as the Narrows. However, these figures should not be used in isolation and should be confirmed by results of environmental quality and impact from monitoring studies given in Chapter 3. These confirm that in general environmental quality of Mulroy Bay is acceptable.

5.4 Recommendations

The results above show that the environmental quality within Mulroy Bay is largely acceptable. Localised impacts have been found near to fish farms, which are mainly within acceptable environmental quality standards. However, the calculation for capacity of the environment to support aquaculture indicates that at the present levels the production is within this capacity but it may be approaching certain limits (e.g. food availability and Particulate Organic Matter (POM).

Many assumptions have been made from literature and experimental derived data and further work is required to confirm these conclusions, and data from additional seasonal inputs (sewage discharge) should be taken into account. On this basis, present levels of fish and shellfish production may be maintained by the environment of Mulroy Bay. Careful consideration though should be given to allowing increases in the present levels within the bay. However, it is essential that environmental management procedures are put into place, and these may include the following:

- Fallowing of fish farms, and possibly shellfish farms if quantitative data suggests localised impacts in future and also between production cycles to allow recovery of sediments. This should be for a minimum of three months, but rotation of sites between production periods is preferable. It is advised to allow natural recovery rather than direct action (e.g. removal of waste) as the sediments adjacent to finfish and shellfish farms are within defined acceptable limits at most sites within Mulroy Bay. Fallowing is particularly important at sites where environmental results show significant impact on sediments, such as at the Kindrum fish farm site, and should be instigated as a matter of course.
- Regular monitoring of water and sediment quality within the bay, particularly near to both finfish and shellfish farms, and at predetermined and fixed reference stations. This should employ both sampling/analysis and visual techniques at least once per year. These can be compared spatially, temporally and with defined environmental quality standards.
- Employment of new technology in feed and feeders to minimise waste should be considered. This will enable more efficient use of food and less waste to the environment through improved Food Conversion Ratio (FCR) (Telfer et al, 2001).
- Investigation for the potential for integrated fish and shellfish farming within Mulroy Bay. The potential for this has been recognised (Troell, 1996) but rarely implemented for mariculture in temperate areas. In an area such as Mulroy Bay

where both fin fish and shellfish are farmed intensively there may be scope for this approach.

- Systems are available for coastal management and assessing aquatic resources use. These are often GIS based and are in their infancy and can be used for site selection, environmental effects (Perez, et al, 2002) and resource management (Aguilar-Manjarrez and Ross, 1995). Potential for the use of such systems for the mangement of aquaculture and other users of Mulroy Bay should be investigated. Designation of the site as management areas or the implementation of a single bay management plan such as the CLAMS system, would help in the integrated management of aquatic resources within Mulroy Bay.
- Further work should be undertaken to refine the estimate of carrying capacity within Mulroy Bay as better information and data becomes available, such as inputs of nutrients from towns and holiday homes. This would allow seasonal variability to be taken into account, especially from the influx of holiday-makers during the summer months.

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