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Impacts of marine fish farming with sea cages in Botany Bay

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IMPACTS OF MARINE FISH FARMING WITH SEA CAGES

IN BOTANY BAY

By

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CERTIFICATION

I, David Barker, certify that this thesis, submitted in part fulfilment of the requirements for the award of Honours Masters of Environmental Science, in the School of Earth and Environmental Sciences, University of Wollongong, is wholly my own work unless otherwise acknowledged. The document has not been submitted for qualifications at any other academic institution.

David Barker
1st March 2008

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ABSTRACT

Environmental impacts of a fish farm utilising sea cages, located in Botany Bay (NSW) have been assessed prior to and during commercial farming on the site. Sampling criteria and methods were devised to observe possible changes in bottom sediments resulting from fish wastes and possible increases in zinc resulting from placement of two zinc coated wire sea cages on the fish farm. Specifically, sediments from fish farm cage sites within the study site and from seven control locations within Botany Bay were analysed for sediment Total Phosphorus (TP), Total Organic Carbon (TOC) and Zinc (Zn). Sediment samples were also collected and archived for future benthic invertebrate analyses, if required.

Concentrations of sediment TP and TOC taken beneath the fish farm site displayed high spatial and temporal variability but were not significantly different from concentrations taken from control sites. These concentrations were comparable with available data for sediments at other sites within the Sydney Region. Concentrations of TP and TOC beneath sea cages did not significantly increase throughout the period of this study, indicating that fish farming activities had no detectable impact on sediment nutrients. Concentrations of Zn in the surficial sediments in Botany Bay ranged between 2.58 and 282.87 mg/kg and also fluctuated spatially and temporarily. Observed concentrations of Zn beneath sea cages were also well within the range of Zn concentrations sampled throughout Botany Bay. However, and most importantly, concentrations of zinc in surficial sediments under zinc cages did not increase significantly compared to those from control sites. In addition, further investigations were performed to assess possible impacts of zinc coated wire sea cages on Zn levels in the water column and in the farmed fish. Zinc coating on cage netting is used to reduce the accumulation of biofouling organisms.

These results indicated that no significant increases of Zn could be detected in the water column as a result of the placement of the cages on the farm site. Fish (*Pagrus auratus*) grown within the coated wire sea cage had no noticeable increases in Zn

concentrations. The degree of fouling by algae and invertebrates on zinc coated wire was substantially less than that occurring on the “soft” netting currently used by the local aquaculture industry. Overall, no significant impacts from sea cage culture of fish, in terms of sediment TP, TOC, and Zn were found. In addition, zinc coated cages were found to offer considerable advantage in lower rates of marine fouling when compared to material currently used for sea cage netting.

CHAPTER 1. INTRODUCTION

1.1 General introduction

The culture of marine fish is a viable commercial prospect in many countries throughout the world. In Japan, China and throughout Asia the aquaculture of fish in sea cages is a huge industry producing hundreds of thousands of tonnes per annum of a range of estuarine and pelagic species (FAO 1998, ABARE 2001). Preliminary estimates made by the Food and Agricultural Organisation (FAO 1998) reported that in 1997 world aquaculture production had increased to 28.8 million tonnes and comprised 23.6 % of total world commercial seafood supply (FAO 2008). By 2007, world aquaculture production had reached 53 million tonnes. In countries where aquaculture is a highly developed industry, its contribution is much greater. For example, in 1997, aquaculture supplied 60.2 % of the total Chinese fisheries landings of 39.9 mt (FAO 1998).

By comparison, the contribution of Australian aquaculture industry to total Australian fisheries production is relatively low at around 30% or 43,600 tonnes. However fisheries statistics indicate that over the period from 1990-91 to 1999-2000 the Australian fisheries production increased in value by 56 % while the value of aquaculture increased by 150 % (ABARE 2001). This increase seems to have plateaued as aquaculture contributed 35% of the total value of Australian fish production in 1994-1995 (ABARE 2006). The majority of the Australian gross aquaculture value is comprised of a relatively few high value species such as tuna, pearls, salmon, oysters and prawns.

The technology and infrastructure required to establish large scale finfish aquaculture production in Australia is now readily available to industry, and with an increased

demand and value in seafood in Australia and throughout the world, further growth and success of aquaculture in Australia can be expected (ABARE 2001). Over the past few decades, the development of marine hatcheries in South Australia (SA) and New South Wales (NSW) has resulted in an increase in production from sea cage fish farms (ABARE 2001).

In SA, mulloway and yellowtail kingfish have been hatchery produced and farmed and southern bluefin tuna have been captured and also successfully produced in sea cages (ABARE 2001). Australia's largest sea cage operations are situated in Tasmania where Atlantic Salmon *(Salmo salar)* is farmed (ABARE 2001).

As the aquaculture of marine fish in open and sheltered water in Australia is a relatively new industry, it is vital that farming practices are established and adopted which cause minimal impacts upon the surrounding environment (Battaglene 1996, Beveridge 1997, SEPA 2000). The development of environmental monitoring criteria and programs are of great importance in assessing the sustainable management of the selected farm site area (Bridger 2000, SEPA 2000).

The most common impacts caused by sea cage culture are changes to the seafloor beneath fish cages. As a result, the monitoring and assessment of the sediments beneath the farm site and the surrounding environment is a primary focus of environmental studies of aquaculture impacts (see Gowen & Bradbury 1987, Wu et al. 1994). Another possible impact is the degradation of the sea cage itself. For example, zinc coated sea cages are used in some aquaculture farms as they offer increased strength and security and as they are thought to reduce the degree of biofouling, and therefore reduce the cost of maintenance on the cages. However the use of zinc coated cages may increase the level of zinc in the environment.

1.2 Purpose of this research

There have been numerous studies focusing on the possible environmental impacts of sea cage aquaculture throughout the world, however at the start of this project there had been no other published data on the possible impacts of sea cage culture in Australian waters. The purpose of this research was to establish and examine a number of environmental criteria likely to indicate if any detectable changes had occurred to a farm site, or surrounding habitats, within Botany Bay, NSW. Information gained from this study is likely to form the basis of further investigations into possible environmental impacts of sea cage aquaculture at this and other locations in NSW.

Much of this research has been carried out as part of studies required under a licensing agreement from the NSW Environmental Protection Agency (EPA) now NSW Department of Environment and Climate Change (DECC). These agencies collaborated to establish the criteria for ongoing environmental monitoring. Sutherland Shire Council (SSC) is now the consent authority for the fish farm licence and any associated environmental issues. The SSC decided it was necessary for the proprietor to produce an Environmental Impact Statement (EIS) for the trial farm site with assessment of particular areas (see SBA, EIS 1999). The EIS incorporates some of the information and findings of this study.

The future of fish farming in NSW waters depends on the ability to economically produce fish without causing unacceptable impacts on the surrounding environment. It is therefore critical that farms be competently managed and that proprietors provide adequate ongoing environmental monitoring and assessment.

As discussed later (see Chapter 2), the major potential impact due to sea cage aquaculture is on the sea floor, beneath the sea cages. Consequently this research concentrates on possible impacts on sediments. Sediment samples were analysed and

assessed for nutrients, Total Organic Carbon (TOC) and Total Phosphate (TP) and Zinc (Zn). Sediment benthos samples were collected, fixed and archived for future reference and photographic images of the seafloor (specifically for sea grasses) were also archived. Observations were also made to assess the biofouling properties of zinc coated sea cages and nylon netting to assess the possible reduction in biofouling and the degree of biological waste material produced by each mesh type.

1.3 Reason for this study

The rationale for this study arises from the need to evaluate the impact of sea cages used for fish farming in Botany Bay, as well as the use of zinc coated metal as an alternative material to traditional net cages. Specifically research included:

Assessment of impacts on the sediments beneath the sea cages in terms of:

- Analyses of sediments for nutrient and TOC TP and Zn using a modified Before, After, Control, Impact (BACI) experimental design.

- Assessment of biofouling properties of zinc coated wire sea cages with respect to mesh hole size over expected two year life period and a comparison of the biofouling properties of both coated wire mesh and traditional nylon mesh was made using replicated quadrats of mesh types over a twelve week period.

- Further monitoring of control locations around Botany Bay (see Chapter 2), so that a comparative assessment could be made between them and the farm site.

As well, photographs of seafloor beneath cages and of sediment cores were archived, for description of macro benthic fauna. This section of work is not reported in this thesis.

1.4 Objectives

The research conducted for this project aimed to use selected environmental indicators to examine possible detrimental impacts occurring as a result of the establishment of the fish farm. Specifically the main objectives were:

1. To review previous research methods used to monitor and evaluate the possible environmental effects of sea cage aquaculture.

2. To examine sediment concentrations, through time, beneath the sea cage sites and at seven control locations within Botany Bay and to:

   (a) Determine if there are any changes occurring in the level of TOC and TP that could indicate possible nutrification from accumulation of excess fish feed and faeces on the sea floor beneath the sea cages and

   (b) Determine if there are any changes occurring in the level of zinc that could be attributed to the placement of the zinc coated cage in this location. Levels of zinc were assessed in the water using oysters as bio-indicators.

3. To develop and test methods to determine the growth of biofouling occurring on the zinc coated steel sea cage, netting and nylon netting, in order to evaluate and compare the possible reduction in mesh size resulting from biofouling material. Biofouling can cause environmental impacts to the surrounding sea floor and waters from waste disposal.

1.5 Scope and limitations

At the time of establishment of the sea-cage aquaculture in Botany Bay, there was little previous local information available about possible impacts of marine sea cage aquaculture in Australia (but see Chapter 2 for more recent information). For this reason, key issues were determined through consultation with NSW DECC and NSW Department of Primary Industries (NSW DPI). The study was restricted to one location, as the aquaculture operation was only within Botany Bay. As well, the high cost of laboratory analyses and the cost of SCUBA divers to assist in collection of samples restricted the number of possible sampling locations.

The project included the sampling at seven control locations and 15 impacted locations within the area leased for the fish farm and throughout Botany Bay. At each location, sediment samples were collected for nutrient analyses, and benthos archiving, and photographs were taken of the sea floor every four months, over a two-year period. A further 12 months of sampling of sediments at all control sites and the six sea cage sites for zinc analyses was carried out. The development of methods and collection of data to determine the growth of biofouling occurring on the sea cage netting and nylon netting and the subsequent reduction in mesh size of each mesh type, has also been included.

The proprietor of Silver Beach Aquaculture provided funding for environmental monitoring to collect and analyse sediments. Benthos samples were archived, in case apparent changes occurred in the sediment nutrient and TOC levels beneath the sea cages. In such cases, the benthos samples could be further processed and analysed to examine possible changes in invertebrate biodiversity. Samples of sediment zinc and phosphate concentrations were analysed at the University of Canberra under a contract arrangement, with OneSteel Ltd, MarineMesh™ sea cage environmental assessment.

The seafloor beneath the sea cages was found to have almost no seagrasses and subsequently the proprietor was granted a permit to "destroy or remove" any small areas of seagrass on the lease site, caused through normal disturbances from establishment and operations (see SBA EIS 1999). As a result it was considered unnecessary to further process and analyse photographic images of the sea floor and seagrasses, and these data are not covered in this thesis.

The collection of photographic records of seagrass and preservation of benthic fauna samples, provide an excellent source of baseline sediment data and visual records of the farm site and all control sites through time, if any future problems arise. These data will be invaluable for any future environmental assessments of the fish farm lease at this location, but are not discussed in this thesis.

1.6 Thesis Outline

The remaining chapters of this thesis are briefly outlined below.

Chapter 2 presents a literature review providing a background to the development of aquaculture in Australia and throughout the world and also into general and selected environmental impacts of sea cage aquaculture. Chapter 3, Impacts on Sediments, provides a description of the study location and positioning data for all sea cage and control sites. The material and methods, tables of results and statistical analysis and brief discussion of results are included within Chapters 3 & 4. Chapter 5 covers the issue of Biofouling of sea cage netting. The final chapter (6) provides conclusions and recommendations for ongoing environmental monitoring of this operation and location.

CHAPTER 2. LITERATURE REVIEW

2.1 Introduction

It is difficult to quantify the level of development in sea cage aquaculture on a world scale. However world fishery production, including capture fishery and aquaculture, increased from 117 million metric tonnes (mmt) in 1995 to 121 mmt in 1996 (FAO 1997). This increase may have been attributed to better fishery management and regulations, as well as the expansion of the aquaculture industry worldwide (Liao 2000). Despite the short development period, sea cage farming has been recognised as a potentially high production industry and the level of research and technology needed to establish procedures and practices in this industry has increased quite rapidly, particularly in the last decade (ABARE 2001).

The demand, consumption and import of seafood is increasing world wide. For example, China imported some 1,516,000 t of seafood in 1997 (Pauly 2002). Other Asian countries, such as Japan and Korea, also import large quantities of seafood (Pauly 2002). It has been recognised that aquaculture must be developed to meet the estimated shortfall in world supply of seafood from the commercial fishing sector (Pauly 2002). This recognition has resulted in governments of many countries focussing on the promotion of responsible aquaculture practices, such as those developed by the Global Aquaculture Alliance (GAA). The GAA is an international industry-based organisation that is developing a code of practice to guide the aquaculture industry toward sustainable development (Pauly 2002). In countries such as Scotland, where sea cage aquaculture is extensive, the Scottish Environmental Protection Authority (SEPA) has introduced specific guidelines for environmental assessment of possible impacts. Monitoring requirements are dependent on farm size and local hydrographical conditions (SEPA 2000). In addition, several non government organisations have formed to promote sustainable aquaculture. The basic

methods outlined for environmental assessment in these international publications were used as a basis for this study.

2.2 Development of sea cage aquaculture

This section of the literature review describes the international and Australian development of sea cage aquaculture.

2.2.1 International development of sea cage aquaculture

Sea cage aquaculture has been steadily developing over several decades in many countries, such as Japan, China, Norway, Italy, Sweden, Denmark, Ireland and Scotland. These industries have mainly been producing salmonid species. The culture of higher valued marine species, particularly in sea cages, is still a developing industry in many countries. Japan is generally considered the world leader in aquaculture research, claiming the ability to spawn and rear the larvae of many difficult marine species. For example, Japan produced over 100 million seed stock of red sea bream (or snapper) alone in 1993 (Nobuhiko & Perez-Enriquez 2000) and cultured 80% of its total kingfish production. Japan is also culturing northern bluefin tuna (Miyako pers com. 2003) the highest valued fish on the international market. It also produces and cultures a variety of invertebrates such as spiny rock lobsters, sea urchin, abalone and pearl oysters (Nobuhiko & Perez-Enriquez 2000).

The USA now has many farm operations producing a variety of estuarine and pelagic fish species, such as channel catfish and dolphin fish (Mahi Mahi) as well as many shellfish species (ABARE 2001). The Channel catfish and Red Sea drum hatcheries in the USA have produced fish on a large scale over many years and now support an industry producing thousands of tonnes of product annually (ABARE 2001).

South African researchers are investigating the potential for farming mulloway on the west coast (the same species as farmed in NSW, Australia) on the west coast in the Western Cape Province. The potential culture of rainbow trout *Onchororhyncus mykiss*, tilapia *Oreochromis niloticus* and common carp *Cyprinus carpio* are also being considered (see Van Roey et al. 2000).

In India, the extensive fish culture in coastal aquaculture systems successfully produces about 12000 t per annum of local species such as milk fish, grey mullet, sea bass, sand whiting, grouper and pearl spot (see Marichchamy 2000). There are also extensive sea cage and marine aquaculture farming systems scattered throughout South East Asia and in areas around Hong Kong and the Philippines. Many tropical regions and Pacific Islands State, such as Tonga, Fiji, New Caledonia and the Solomon Islands have cultivated a number of tropical marine fish species, such as tilapia since the 1970's (ABARE 2001).

The development and success of sea cage culture in each region has been affected by various local factors. In many under-developed countries, the limited available resources have hampered the growth of industry technology, such as the development and production of commercial pelleted diets (Battaglene 1996). The lack of available dry feeds has often forced the use of soft fish feeds, which reduce growth rates and pose a large environmental impact through wastes and through the capture of low priced fish for aquaculture feeds. The lack of regulation and control over production has also resulted in environmental problems (Battaglene 1996).

In many countries, other limiting factors have reduced the potential development of sea cage aquaculture. For example, there is often a distinct lack of suitable farm sites which limits the number of farms and the size of already established operations (Chou 1997). In Singapore, there is a limited area for sea cage farming, as few sites are available that do not conflict with activities that contribute significantly to the economy, such as shipping, oil refining (Chou 1997). In NSW recent studies have

concluded that the major viable opportunities for commercial scale sea cage farming are in off shore or open ocean farming locations (Liszka 2000).

The availability of different aquaculture species for industry can also be a factor limiting the expansion and development of the industry (Pillay 1977). For example, in the late 1990's, NSW had two established sea cage farms that were unable to properly stock cages with fish mainly due to a lack of suitable hatchery production of larvae (Battaglene 1996). In many countries, the only source of juvenile marine fish is through the capture of wild fish from native stocks, as hatcheries and the required technologies have not yet been developed.

2.2.2 Australian development of sea cage aquaculture

The production of marine fish in Australia is a relatively new industry compared to the freshwater aquaculture industry and the marine finfish aquaculture in other countries (Kearney1996). The culture of marine finfish can be achieved in land based facilities, in ponds or in recirculating tanks, or in sea cages or floating net pens offshore in open seas or in rivers, lochs or bays. The development of the finfish aquaculture industry in Australia has been slow but progressive (ABARE 2001). This is not surprising considering that Australia has large and viable commercial fishing industries, producing large quantities of fresh seafood, which enjoys a high reputation world-wide for quality, due to our pristine waterways. This contrasts with European, Asian and American countries which have found the culture of finfish a necessity as natural stocks have already been exploited and the supply of wild-caught fish cannot meet the demand for seafood (Beveridge 1997).

Only in the past few decades has Australian fisheries experienced the effects of over-fishing, and declines in catches of several fish species, such as the gemfish and orange roughy fisheries (Andrew et al. 1997). The sustainability of many other commercially important fisheries is threatened by over-exploitation (Andrew et al. 1997). The subsequent decrease in local product has seen an increase in the demand for species

such as snapper (*Pagrus auratus),* mulloway (*Argyrosomus japonicas),* yellow tail kingfish *(Seriola sp)* and yellow fin bream (*Acanthopagrus australis)* (FAO, 2000). These species are the focus of research and development in NSW and SA. The potential for operators to capitalise on this market has generated the development of new aquaculture techniques and the growth of the industry itself.

However, there are many limiting factors affecting development of sea cage aquaculture in Australia. A major limiting factor is the lack of suitable sites, in that the southern coastline of Australia experiences intense weather patterns and currents and there are few suitable protected bays which are not already being utilised by other user groups or are not located amongst high-priced residential real estate (Battaglene 1996, Liszka 1996, Glendenning and Read 2003). The value of land in Australian cities, particularly in Sydney, is extremely high, forcing potential farmers to seek land sites far from the city. This can impact upon the economics of conducting and staffing a business.

Another major impediment has been the environmental concern of establishing industries producing potentially highly nutrified waste waters. Australian waterways are managed to strict guidelines under a number of government bodies including the NSW DECC, NSW DPI and local councils (Battaglene 1996).

It is difficult to establish any new aquaculture industry associated with the waterways in any of the Australian states, other than South Australia and Tasmania where a number of farming permits have been granted. For example, in NSW there are a large number of government bodies involved in the approval process for sea cage farming. These include; NSW DPI, the Department Planning, the Department of Natural Resources, the NSW DEC and the Department of Regional Development. Often development proposals are dealt with by the NSW Premiers Department, to ensure co-ordination between agencies (Liszka 1996, Glendenning and Read 2003). It is also necessary for an independent EIS be prepared, before an aquaculture permit is issued.

Development of sea cage aquaculture in New South Wales.

The first marine finfish aquaculture trial in NSW was conducted in Botany Bay, as a research platform to investigate the potential of fish farming in NSW waters (Quartararo1996). This trial resulted in several tonnes of snapper *Pagrus auratus* and mulloway *Argryrosomous japonicus* being cultured or grown, using anchored sea cages (Quartararo 1996). The success of this trial has been the basis of commercial marine fish farms being established in NSW waters. This research fish farm later became the site for the first commercial marine sea cage fish farm in NSW, Silver Beach Aquaculture. The second commercial fish farm, Pisces Marine Aquaculture P/L, began operation in 1998. This second farm site was situated approximately two kilometres off the NSW coast at Port Stephens and in 2000 successfully produced approximately 40t of snapper (PISCES 2003). There is a potential for these farms to dominate the local market for some species, as well as develop new export markets. However the availability of fingerlings has been a limiting factor in the development of fin fish farms in NSW. There has been a bottleneck in the large-scale production and supply of fingerlings for the "grow out" industry. A reliable supply of juvenile fish or seed stock is a fundamental requirement for fish farming and is a critical factor in the commercial success of new marine fish farming ventures (Shepherd and Bromage 1988).

In recent years, the successful production of several hundred thousand snapper and mulloway fingerlings in NSW has allowed the aquaculture industry to grow substantially larger crops (SBA, pers. com 2002). This development in fingerling production should positively impact on the development of the sea cage industry in NSW waters.

Although other species such as Atlantic salmon fingerlings were commercially available to fish farmers, they are not suitable to the temperate waters of NSW which are too warm for the culture of salmonid species in sea cages (Battaglene 1996).

Also, the introduction of new species is not allowed under the NSW DPI (Fisheries) aquaculture policy. The research and development of the hatchery production of fingerlings on a commercial scale in Australia has been slow, mainly due to the high establishment and operational costs involved with developing a marine hatchery (Battaglene 1996).

In NSW, the culture of finfish has mainly involved land based activities usually culturing fresh water species. For example, in 2001 the silver perch aquaculture industry had an estimated value of AU$5 million annually (ABARE 2006) and the barramundi *Lates calcarifer* industry is currently valued at $1.3 million annually (ABARE 2006). The barramundi industry is much larger in Queensland and the Northern Territory, as the production and containment of fry less than 60mm is not permitted in NSW. Although this factor has inhibited the growth of the industry in NSW, barramundi fingerlings are purchased and translocated from Queensland hatcheries and grown to market by several NSW Based farmers, using saline ground-water fed ponds. While the potential value of the barramundi industry is considered to be high, the production has increased far less than was anticipated by Cable (1996), when the industry was first established.

It should be noted that the overall value of aquaculture in NSW is dominated by the culture of Sydney rock oysters, which originally formed the largest and most profitable aquaculture industry in Australia. The production of oysters in NSW is estimated to be worth approximately AU$36 million per annum (ABARE 2006). The sustainability of this oyster industry has been affected in recent years by a number of factors such as disease and water quality problems (Nell 2002).

Development of sea cage aquaculture in South Australia

In South Australia, marine sea cage aquaculture has developed rapidly in recent years. Mulloway and yellowtail kingfish have been hatchery reared and successfully grown to market size. The production of over 200 t of kingfish in 2000 (ABARE 2001) was more than double the wild capture fishery for the region (ABARE 2001). The highest valued sea cage operations in SA are based on southern blue fin tuna (SBT) valued at $139 million annually (ABARE 2006). However the aquaculture of SBT is based on the grow out of wild caught juvenile fish. This industry is limited to farm operators with an endorsement to capture or access wild SBT. This reliance on wild caught fish will hinder the future development of tuna culture in SA, and hatchery production of fingerlings is under investigation (ABARE 2001).

Development of sea cage aquaculture in Tasmania

In Tasmania, aquaculture production increased by 16 % to 15,900 t 1999 between and 2000. The basis of this success was due to the farming of Atlantic salmon, valued at $112 million in 2004-2005 (ABARE 2006). Although the practice may be criticised, as the species is exotic, the industry has been financially successful and several farms are still in operation after a decade. Again the limiting factor for industry growth has been the availability of other fingerling species for aquaculture. Farmers have been limited to salmon stock and unable to diversify, being forced to produce and sell a species not endemic to the region.

Development of sea cage aquaculture in Queensland

In Queensland, the total wild caught species in 2004-2005 was around 25,000 t, valued at $190 million (ABARE 2006). In comparison, the total production from aquaculture was around 29t but was valued at $252 million. The high value per tonne is a result of the aquaculture industry targeting high valued species. In Queensland, aquaculture is dominated by prawn farming which was valued at $45 million in 2004-

2005 (ABARE 2006). The primary finfish cultured in Queensland is barramundi and production is mostly from land-based operations. The Great Barrier Reef surrounds much of the Queensland coastline and coastal waterways are considered sensitive areas (QLD DSD 2003). The introduction of aquaculture farms has been limited by concerns raised by many environmental and tourist groups. For example, Sun Aqua P/L have been seeking approval to farm snapper and kingfish on the eastern side of Morton Bay since 1999 (QLD DSD 2003). Other recent proposals include sea cage aquaculture of barramundi in Hinchenbrook channel. The Queensland Department of Primary Industries (QDPI) and two commercial hatcheries (one in Queensland and one in Western Australia) are also undertaking research on the development of aquaculture techniques for grouper or estuary cod *Epinephelus coioides* (Rimmer 1998).

Development of sea cage aquaculture in Victoria

The total value of Victorian fisheries was estimated at $109 million in 2004-2005, of which aquaculture contributed around $24 million (ABARE 2006). Freshwater trout comprise 50%, by volume, of the states total aquaculture in 2004-2005 (ABARE 2006). Black mussels are also successfully farmed in Victoria. Recently the Victorian Government announced that it will dedicate a number of areas for marine sea cage aquaculture, which should result in an increase in this activity. A recent aquaculture plan suggested that Victoria could triple its fish farming production over the next decade (Anon, 1997).

Development of sea cage aquaculture in Western Australia (WA)

WA has Australia's largest commercial fishing industry after Queensland and provides Australia's largest employment in the marine fishing sector, although NSW and Tasmania were the largest employers in the aquaculture sector (ABARE 2001). Western Australia possesses the natural resources to support a significant land-based aquaculture industry (Fisheries WA) and has the second highest valued aquaculture

industry in the country, after SA (ABARE 2001). Aquaculture in WA is mainly for pearl oysters, yabbies and mussels (ABARE 2006). The oyster industry was valued at $122 million making up 95% of the total aquaculture production for WA (ABARE 2006).

Although the barramundi industry in WA is considered to still be in its infancy producing less than 10 t in 1997/98 (Thorne 2002), it has shown considerable growth in recent years and was identified by WA Fisheries as a considerable prospect for large scale aquaculture in Lake Argyle in the Kimberley (Thorne 2002). Up to 2006, there were no sea cage fish farm operations in WA.

Northern Territory.

The Northern Territory of Australia has a high potential for aquaculture development. Although remote, it has a largely untouched coastline, a pristine environment and a very small population (Field 2000). The total fisheries production in the Northern Territory was valued at around $59 million in 2004-2005 (ABARE 2006). Pearl farming was the only aquaculture activity recorded in the Northern Territory and was valued at around $28 million in 2004-2005 (ABARE 2006).

2.3 General impacts of sea cage aquaculture on the environment

Environmental impacts from sea cages vary greatly depending on the nature of the operation, from small impacts at the farm site to regional impacts from large-scale developments (Pearson and Rosenberg 1978, Silvert 1992, SEPA 2000, Crawford, 2003, Macleod et al. 2004, Fernandes et al. 2007). Impacts also vary with factors such as tidal cycles, season and the operational stage (Pillay 1992). The general impacts of sea cage aquaculture can be classified into a number of categories, such as sediment eutrophication, red tides from fish wastes, genetic impacts on wild fish stocks through the accidental release and stocking of fish into the environment. Of

these general impacts, water and sediment enrichment and biological impacts from sea cage aquaculture are of most concern and form the basis of this review.

2.3.1 Water and Sediment Enrichment

Aquaculture was traditionally considered an environmentally sound practice but increased production, inadequate planning, inadequate environmental controls and an increased awareness of the aquatic environment, have resulted in restrictions being placed on the expansion of aquaculture in many countries (Pillay 1992).

The most commonly observed problem associated with sea cage aquaculture is the increase in nutrients in the water column and in underlying sediments caused by the build-up of uneaten fish feed and faeces. This can result in nutrification and then possibly eutrophication in local waters (Battaglene 1996, Pillay 1992, Kibria 1996). Water quality impacts may involve increased nutrients (ammonia, phosphorus, nitrogen and total carbon), increased turbidity, and lowered oxygen (Seymour 1991, Tsutsumi et al. 1991, Pillay 1992). Solid wastes from fish and uneaten fish food may impact on sediments and benthic fauna (Brown 1987, Tsutsumi et al. 1991, Wu et al. 1993, Johannessen et al. 1994, Battaglene 1996), see also further examples in Section 2.3. Waste generated by a sea cage fish farm may result in the flow of carbon and nutrients in liquid and solid forms into the surrounding environment (as described in Section 2.3.1). Low levels of nutrification such as increased nutrients and organic substances within the sediment, can alter chemical balance within the sediment (Wu et al. 1994, Battaglene 1996). This change in chemistry of the substrate can result in changes in the structure of the benthic community (Wu et al. 1994, Macleod et al. 2004, Fernandes et al. 2007). Eutrophication may lead to the increase of hydrogen sulphide (H_2S) production from the sediments beneath the cages, which can reduce the available dissolved oxygen (DO) levels in the overlying water. This localised reduction in DO can cause mass mortality to the fish stocked in nearby cages (Tsutsumi et al. 1991, Pillay 1992, Battaglene 1996, SEPA 2000). In most

circumstance, the nutrification and eutrophication of sediments beneath sea cages generally have the greatest impact on the fish grown in the cages, and can result in economic losses to the fish farmer (Pillay 1992).

In recent years, there has been a focus on the environmental protection and rehabilitation of waterways, and thus on the industries situated on or adjacent to waterways, such as aquaculture operations. This increase in environmental awareness, as well as the improvement in technologies associated with feed production and farming techniques have brought about a reduction in reported incidents of eutrophic conditions resulting from poorly managed fish farms. Kibria (1996) discusses this theory with regards to aspects of phosphorus pollution from aquaculture operations, where improved feed production and nutrition technologies result in far less phosphorus being released into the environment, posing less environmental threat. Heinig (1994) states that the early development and trials of aquaculture occasionally resulted in substantial environmental degradation beneath and adjacent to sea cage sites. This degradation was attributed to a number of factors including inexperience, lack of feed technology, insufficient oversight and limited competition in the market place. Importantly, the farm equipment and feed technology, management and monitoring have now improved (Kibria 1996, Heinig 1994, Crawford 2003).

There have been many studies focusing on the changes in benthic communities beneath sea cages during fish farming operations (e.g. 1988, Frid et al. 1989; Ritz et al. 1989, li-Xun et al. 1991, Tsutsumi et al. 1991, Chareonpanich et al. 1994, Johannessen et al. 1994, Wu et al. 1994, Porter et al. 1996, & Pawar et al. 2000). Probably the most valuable study of benthic fauna beneath marine sea cages was that by Tsutsumi et al.(1991) where the environmental conditions and the benthic communities have been studied in a particular Japanese cove since 1966. Although results from this study indicate that fish farming in sheltered areas is often followed by organic pollution of water sediments, Tsutsumi et al. (1991) discusses the fact that such disturbances in the benthic ecosystem in coastal areas are caused not only by fish

farming, but also by various commercial activities throughout Japan on a much larger scale.

The detection of chemical changes in sediments beneath sea cages has led researchers to further investigations of resident invertebrate communities, and studies to further determine if associated change in benthic communities had occurred (Brown et al. 1987, Tsutsumi et al. 1991, Macleod et al. 2004). It has been established that the first sign of ecological changes in the benthic communities below sea cages is the appearance of opportunistic species, such as polychaete worms (Bergheim et al. 1991, Macleod et al. 2004). In some studies, where visible and chemical changes have occurred in the sediments beneath sea cages the impact on sediments adjacent to the sea cages was established and has been referred to as the zones of effect. The impacts have been classified as ranging from heavily affected to clean or unaffected (Brown et al. 1987, Yokoyama 2002). There is evidence of large spatial and temporal variability in the benthic communities, often related to sediment type (Morrisey et al. 1992, Underwood 1997) and raising questions about the value of benthic biodiversity studies. It should also be noted that the study of benthos communities is generally a very costly exercise. This is mainly due to sample collection, processing and species identification being highly skilled and labour intensive. Also, the large volumes of sediment required per sample, (100-200 cm^3) and the large number of replicates required per site, result in considerable quantities of sediment to be fixed and processed.

Another possible impact or biological change occurring from aquaculture is the stimulation of phytoplankton by the organic compounds in the waste from fish farms (Nishimura, 1983). Phytoplankton growth is influenced by turbulence and turbidity, as well as by effects of light and nutrient availability (Battaglene 1996). Under suitable growth conditions, increased organic enrichment can result in red tides (Gowen and Bradbury 1987).

As discussed above, the liquid waste materials generated by aquaculture have been the possible cause of problems associated with hypernutrification of the water column (Pillay 1992). In many underdeveloped countries the development of aquaculture farms has been uncontrolled, resulting in many environmental problems (e.g. Wu, et al. 1994). Intensive sea cage culture, such as raft farming in Asia and Japan, where fish farm platforms are linked together to form huge structures, over hundreds of hectares in area, has been suggested as possibly contributing to the occurrence of algal blooms and red tides and other associated environmental impacts (Wu et al. 1994). However, it is difficult to determine the precise influence of fish farming on their occurrence (Takahashi and Fukuzawa 1982, Wu et al. 1994). The incidence of "red tides" and other algal blooms is increasing throughout the world (Anderson 1994), and large algal blooms occur naturally in Australia, even in pristine areas such as Jervis Bay, NSW (Battaglene 1996). It therefore difficult to isolate the role of sea cage aquaculture as a primary source of pollution that causes algal blooms.

In summary, although it may be difficult to quantify the amount of pollution entering the environment from a fish farm, and equally as difficult to measure and determine any direct impacts, it is important that these impacts be managed and controlled. As outlined above it has often been observed that poor farm management and husbandry practices can lead to large scale degradation and pollution of the environment, particularly in the close vicinity of the farm site.

2.3.2 Biological impacts

Apart from the obvious impacts from eutrophication that aquaculture can have on the environment there are a number of possible biological implications and concerns that could result from aquaculture practices, as well as some possible positive environmental impacts regarding the future development of marine aquaculture.

The first and most concerning biological impact that could occur is farmed fish stocks escaping, mixing and possibly impacting upon the survival and genetics of wild fish stocks. The release of farmed fish can occur through operator error, storm damage or vandalism. It has been recognised that the escapement and establishment of self-sustaining introduced species can cause alteration of native fish gene pool (Arthrington et al. 1996), which can impact detrimentally on wild stocks and biodiversity. Studies have also established such impacts to be a factor in the decline of several native fish species in the USA (Lassuy 1995).

The most recent report of an environmental impact possibly caused by aquaculture in Australia occurred in Spencer Gulf, (S.A) in 2002/2003, where kingfish aquaculture has been blamed by community, angling and environmental groups for the huge increase in natural kingfish populations (Haxton 2003, Crisell 2003, SA Recreational Fishing Advisory Council (SARFAC) 2003, Recreational Angler's Network of South Australia (RANSA) 2003). The raised many associated issues, including the reduction of local cuttlefish populations and juvenile fish stocks. The SARFAC has attacked the Aquaculture Advisory Committee (PIRSA) implying they have not correctly used research funding to investigate the environmental impacts of releases.

Unfortunately, there is insufficient information in Australia to assess the risk of release of genetic material on snapper and mulloway stocks and of disease transmission (Battaglene 1996), a subject that still requires research today. Overseas studies have been done in an attempt to assess possible impacts of Japanese stock enhancement programs and aquaculture activities of Red Sea bream. It is considered that the artificial propagation in hatcheries may result in the loss of genetic variability and an increase in inbreeding, which may provide negative effects on the genetic diversity, and fitness of both hatchery stocks and wild populations (Nobuhiko and Perez-Enriquez 2000). It is also worth noting that snapper in Japan have been cultured and released into the environment for over thirty years, without any apparent ill effects (Foscarini 1988).

If restocking programs were to be further trialled in Australia, a detailed EIS of possible associated impacts would be necessary. It is also essential that regulations and guidelines be established to ensure that hatcheries producing marine fish fingerlings have acceptable genetic variations in brood stock to ensure the genetic identity of commercially farmed species (FAO 1992).

One common problem faced by fish farmers is the control of pathogens. The populations of endemic parasites can reach extremely high levels in sea cage aquaculture (as described in Chapter 5). This is generally a result of the constant dense populations of fish held in the cages, which allows larval parasites to easily find a host. Parasite populations generally increase at an exponential rate and the onset of parasites may be so rapid and severe that fish mortality can escalate accordingly. An example of such an impact is the transfer of the monogenean parasite of salmon (*Gyrodactylus salaris*), from resistant wild populations from the Baltic Sea to Norwegian salmon, resulting in the loss of many wild populations (SCERU 2002). Although the results of pathogen infestations in sea cage culture can be economically catastrophic for the farmer, there appears to be only a limited threat of any impact resulting on the environment (see Battaglene1996). This resulted in the extinction of many wild populationsThe major environmental problem associated with control of pathogens is the use of chemicals and drugs to treat them (e.g. Read et al. 2003).

There is little documentation published on the impact of sea cage aquaculture on bird populations, as the impact is probably only minimal. According to Pillay (1992) the impact of aquaculture on birds and aquatic animals has not been adequately studied. Most sea cage operations have covers or netting to prevent sea birds from entering and poaching fish from the cages. Birds may get entangled in loose netting possibly resulting in stress, injury or death.

There are other potential problems associated with the control of other aquatic intruders such as seals, sharks and other predatory fish. The high density of fish held in cages and the increased availability of food in the area attracts predators of all

types. It should be noted that a seal damaged a sea cage and killed a small number of large, mature mulloway broodstock held on the farm in Botany Bay (SBA pers.com 2002). Although it was thought that the seal was unharmed during this event, the fact that these mammals may be at risk of injury or mortality is a considerable environmental concern.

2.3.3 Positive impacts of marine fish farming

Recognition should also be given to the positive impacts, resulting from the development of marine aquaculture. As previously discussed, production from almost all the worlds' major fisheries is either static or in decline and it is unlikely that the natural fish stocks of the world will be able to cope with the steady increase in world population and demand for fresh seafood (FAO 2002). It seems inevitable that the future supply of seafood depends upon the development of environmentally acceptable and sustainable marine aquaculture industries.

Another possible positive impact of the aquaculture industry meeting the demands of world markets and relieving the need and pressure from commercial fisheries is that it may result in higher recreational angling catches and better conservation of biodiversity, in areas where commercial fishing is removed or reduced (Smith et al. 1997). This prospect has already been recognised by fisheries scientists and managers in other countries, and the number of commercial fishing licenses has been reduced, with the intention of restoring waterways and native fish stocks. For example, in Australia both NSW and Victoria have introduced recreational angling licenses to raise revenue, partly to buy out commercial fishing licenses in key waterways. This, in turn, is expected to promote the recreational and tourist value of waterways. It may also increase the demand for cultured fish.

2.4 Selected impacts of sea cage aquaculture

The impacts of sea cage aquaculture resulting from wastes and biofouling are reviewed below, as they are an important aspect of the current research.

2.4.1 Impacts from wastes

The effects of fish farm wastes on the environment are the probably the main focus of concern for most studies observing impacts of aquaculture. These observations have been made since the 1970's and early problems associated with waste material are well documented. Useful reviews of the environmental impacts of sea cages have been undertaken by Woodward (1989), Pillay (1992) and Purser (1992). However most of these studies have been conducted on cold water salmonid farms. Studies on the impact of marine sea cage farms operating in conditions similar to those in NSW are rare (Battaglene 1996).

Intensive cultivation generates large amounts of organic and inorganic waste (uneaten food, faecal and excretory material) all of which are continually produced and released at single point sources into the environment (Tsutsumi 1991). Due to this fact concern has been expressed both by conservationists and by fish farmers who fear that the interaction between a farm and its environment could result in harmful feedback, which may have an adverse effect on the economic viability of the farm (Gowen & Bradbury 1987).

To observe and study the possible impacts of waste generated from sea cage aquaculture it is necessary to understand the input and concentration of substances in fish feed products, the metabolic processes of digestion of these products by the fish, and the flow of resulting waste materials into the surrounding environment (SEPA 2000). Phosphate is a nutritional requirement for animals, including fish, and pelleted fish diet contains phosphorus to ensure adequate growth. It is therefore possible that sediment phosphate levels may increase as fish wastes and uneaten fish food could accumulate beneath the sea cages (Kibria 1996). This increase in phosphates may also alter benthic community structures and the ecology beneath the sea cages. However the release rate of phosphorus from fish feed and faeces into the environment is dependent upon a number of physio-chemical characteristics of the surrounding environment including pH, temperature, oxygen and microbiological

activity (Persson 1988, SEPA 2000). The approximate flows of carbon and nutrients (e.g., phosphorus (P) and nitrogen (N)) through a typical sea cage system have been illustrated in numerous studies, including Battaglene (1996) and Kibria (1996). These illustrations of salmon farms make the assumption, or estimate, that 20% of feed is uneaten, falling to the seafloor. However, Kibria (1996) makes note that the presence of phosphorous in fish feed is far less than originally that used by industry and the quality of fish pellets has also improved. The new extruded diets are more digestible for fish (Kibria 1996), resulting in less waste being generated. Although data from salmon farms was used in these studies it would be reasonable to assume that the approximate ratio and flow of carbon and nutrients generated by these farms would be similar and applicable to this and other studies. However the assumption that 20% of feed is uneaten and lost into the environment (Kibria 1996) is much greater than the estimated 10% of feed being lost at the site of the current study (SBA EIS 1999). A review of current literature revealed no reports of catastrophic problems associated with nutrients from sea cage fish farms.

The productivity of many aquatic systems is limited by the amount of nutrients (particularly nitrogen and phosphorus) available (Battaglene 1996, Kibria 1996). It is therefore possible that the increase in these specific nutrients from sea cages could increase the biological productivity of the surrounding environment. The term hypernutrification is used to describe any increase in the concentration of dissolved nutrients in the water and under certain conditions can lead to an increase in phytoplankton growth and productivity termed eutrophication (Pillay 1992).

Eutrophication occurs when the nutrient supply increases to an amount beyond that useable by the natural community in the system (Brodie 1995). When the demand for oxygen exceeds the supply, the sediment becomes anoxic and at this point there are major changes in the sediment chemistry and the ecology of benthic organisms (Gowen and Bradbury 1987). As a result, the large amounts and constant deposition of waste beneath cages can cause the development of an anoxic zone, which is devoid of macrobenthic organisms (Stewart 2006). In the immediate vicinity of the farm,

macrofauna is likely to be impoverished and dominated by opportunistic species (Gowen and Bradbury 1987).

Although this effect is usually highly localised (Frid and Mercer 1989), the resulting impacts of this circumstance can be substantial on benthic communities and often fatal for fish held in affected cages. The reducing capacity of the sediments depends on the degree of enrichment, and can be assessed by the measure of the redox potential (oxidation /reduction level) of the sediment (Gowen And Bradbury 1987).

It should be noted that hypernutrification does not necessarily lead to eutrophication and that the impact of nutrients is very dependent on the degree of turbulence and turbidity (Woodward 1989). The dispersal of wastes from sea cages and the resulting detectable concentrations of nutrients is influenced by a variety of factors in different geographical and hydrological locations, as are the natural levels of available dissolved nutrients and productivity levels of specific habitats (SEPA 2000).

Although it may be considered favourable that flux levels (or dispersal rates from currents and depth) around sea cages be adequate to reduce the possible impacts of hypernutrification beneath the fish farm, the possibility of the dispersal directly affecting other nearby areas is also a potential threat (Battaglene 1996, Frid and Mercer 1989, Tsutsumi 1991, SEPA 2000). Environmental monitoring of caged fish farming in macrotidal environments suggest there is a potential threat for accumulation of farm wastes in nearby sedimentary sinks (Frid and Mercer 1989).

Interestingly, the most common causes of eutrophication worldwide are not aquaculture but agriculture, sewage and land-based industry (Wu et al. 1986). Although it is evident that aquaculture practices in some countries may have added to the eutrophication problem and associated problems of some waterways, it is often more likely that a combination of waste materials from numerous sources is responsible for the problem.

The growth of the sea cage industry in Australia and the subsequent required environmental monitoring will further increase understanding and knowledge surrounding the impacts of waste matter from sea cage aquaculture and undoubtedly result in the better management and the implementation of strategies and environmentally safe practices of the industry.

2.4.2 Impacts from biofouling

The fouling of sea cage nets by marine organisms poses substantial problems for fish farmers around the world (Lewis 1994). Traditional sea cage netting used on aquaculture farms is woven from multi-filament nylon strands, with each strand consisting of hundreds of nylon fibres. Although this configuration provides the net with great strength, the increased surface area and roughness of these strands are ideal for the colonisation of fouling organisms (Lewis 1994). An excess of biofouling on a cage net can restrict the flow of clean water through the net, potentially reducing the dissolved oxygen levels within the cage (Quartararo 1996). It can also provide an ideal habitat for parasites and other pathogenic organisms. For example, parasitic diseases pose problems in the culture of sparids, as well as other fish species (Roubal et al. 1992). Consequently, the settlement and subsequent rapid growth rates of micro algae, macro algae and invertebrates on immersed surfaces such as nets is one of the major problems faced by marine fish farmers in Australia and throughout the world (Lewis, 1994).

Settlement of biofouling organisms such as algae, barnacles, oysters and other invertebrates is often inhibited on surfaces coated with active metals such as copper and zinc (Mance 1987). Metal based products are commonly used in marine environments to protect structures from corrosion and biofouling (Lewis 1994). The changing and cleaning of nets is a labour-intensive process and greatly contributes to the overall production costs in finfish aquaculture (Lewis 1994). Furthermore, in NSW the waste material generated (i.e., the fouling organisms) must be disposed of in an environmentally friendly manner (EPA 1998) posing further costs to farmers. The

cost of controlling biofouling in the Australian aquaculture industry has been estimated to be approximately AUS$1billion per annum (Lewis 1994). There are implications for fish farmers cleaning nets, or removing biofouling whilst the cages are still in the water, as the biological waste material removed from the nets could cause degradation of the sea floor. The impacts of increased nutrient levels on the sea floor beneath sea cages are already well documented (see Section 2.3.1).

CHAPTER 3. IMPACTS ON SEDIMENTS

3.1 Introduction

Sea cage aquaculture is most likely to impact the sediments beneath and surrounding sea cages (see Chapter 2). Increases in sediment nutrients, particularly phosphates (Kibria, 1996), can alter conditions leading to sediment degradation and changes in benthic communities (Tsutsumi et al. 1991, Chareonpanich et al. 1994, Johannessen et al. 1994, Wu et al. 1994, Battaglene 1996, Porter et al. 1996, & Pawar et al. 2000). Increases in sedimentary Total Organic Carbon (TOC) are widely used as an indicator of biological loading from aquaculture in sediments (Burridge 1999, Karakassis 1999, Zitko 2001, Steeby 2002). As these chemical changes may impact upon the benthic community and the environment at the site of the fish farm, TOC and Total Phosphate (TP) were chosen in this study as indicators to monitor and assess the impact of the establishment of the sea cages on the sediments of Botany Bay. Total nitrogen (TN) was not used as it is difficult to quantify due to its unstable nature (Spooner 1999).

The first objective of this study was to measure changes that may have occurred in the sediment TP and TOC at the farm site caused by the fish farming operation. To achieve this it was necessary to collect, analyse and compare sediments from selected sea cage locations, and other selected control locations within Botany Bay. Sediment samples were collected at all sampling locations (see Figure 3.1) on five occasions, with a four month period between sampling. All sediment samples were analysed for TOC% and TP ug/g. These methods are explained in more detail below (see Section 3.2). Zinc was not initially considered to be a substance that would be generated in high levels when the fish farm and this research commenced. However two zinc coated wire cages were placed on the fish farm by OneSteel Ltd to investigate the durability and possible environmental impacts these sea cages may have on the surrounding habitat. OneSteel manufacture zinc coated wires (MarineMesh™) that have a far greater loading of zinc than traditional coated wires. The loss of zinc from

the wire cage still occurs and the life span of the product is still limited, as with traditional coated wire. However, if the life span of MarineMesh™ is higher than normal cages, then the economic value to the farm operator may also be greater.

The second objective of this study was to examine collected sediments for zinc so that detectable increases in zinc due to the placement of the zinc coated wire sea cages on the fish farm in Botany Bay could be assessed. Sedimentary zinc was examined from beneath the wire sea cages (placed in fixed locations throughout this study) and at all control sites through time, using the same methods used for collection of TOC and TP samples (see below).

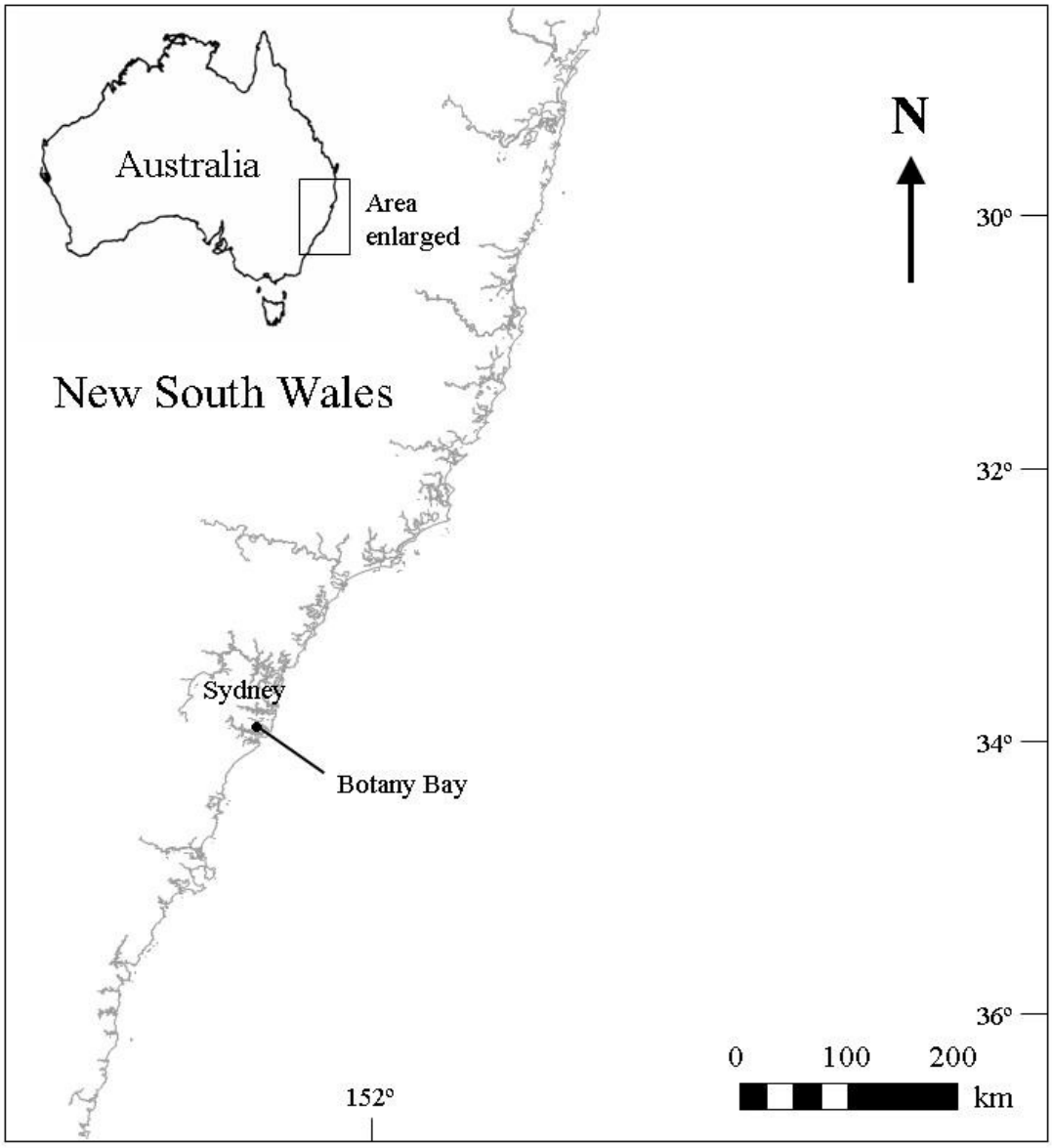
The dispersion of zinc through corrosion of the wire coating into the environment is an inevitable process. However, depending upon the rate of dispersion and the associated current flow of the surrounding waters zinc may accumulate within the sediments beneath the wire sea cages. It was therefore considered important to establish and quantify the loss of zinc coating from the wire cages over their expected life span. To do this it was necessary to assess the zinc coating remaining on the wire sea cage after it had reached its expected life span and was no longer being used to hold fish (see Figure 3.5).

3.2 Materials and methods

3.2.1 Study area and sampling regimes

The study was carried out in Botany Bay, which is located a few kilometres south of Sydney, NSW, Australia (see Figure 3.1). Silver Beach Aquaculture (SBA) has its farm site located on the south eastern foreshore of Botany Bay approximately 700 m off Silver Beach (Kurnell). The farm is approximately 100m from the Kurnell oil refinery wharf (see Figure 3.2). The Bay is generally shallow (< 6 meters deep) in most areas except for a 12m deep shipping channel at the entrance leading to major port and wharf facilities.

Figure 3.1 Map showing Botany Bay, Australia, where this study was conducted



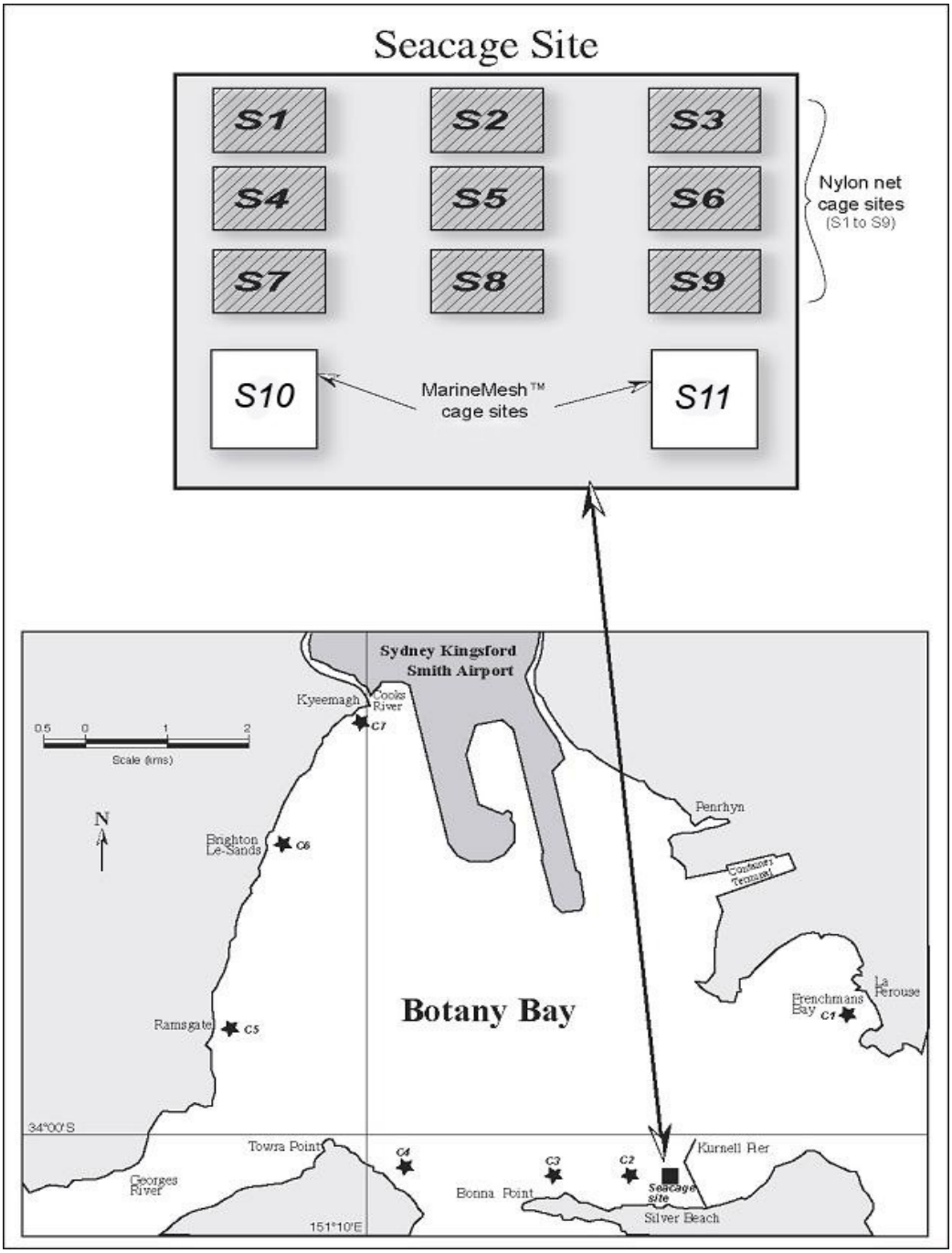
Sea Cage Sites

In total, there were eleven sea cage sites used during the period of this project. Three nylon net cages were rotated between 9 of these sites and, in addition, two zinc coated wire cages were permanently located at two sites (see Figure 3.2). The nylon cages were of dimensions 15 m x 8m, and the zinc coated cages were 10m x 10m. Each of the three nylon cages was rotated every four months by physically moving the cage to an adjacent site. Each site was used for a period of four months and then not used for another twelve months to allow for recovery of the seafloor.

Each of the nylon cage nets was removed and cleaned every 2-6 weeks depending on season, mesh size and the rate of fouling accumulating on the nets. For further details concerning fouling rates on these nets, see Chapter 4 (Biofouling).

A differential global positioning system (DGPS) was used for locating all sampling sites to ensure the exact coordinates of sites were recorded (Table 3.1) so they could be relocated at each sampling time as required. DGPS positions are accurate to within 5 meters. Distances from landmarks were also recorded and used to locate sampling positions at control sites. Distances from landmarks were recorded using distance-fixing binoculars. The mooring blocks for each cage site located the positions of sample sites beneath the sea cage sites. SCUBA diving was used to collect replicate core samples of surface sediment from each site (sea cage and control sites) on the same day for analysis, and photographs of all nursery sites were taken on the following day (where possible).

Figure 3.2 Botany Bay, NSW showing approximate positions of all control sites (C1-C7) and sea cage sites. Schematic diagram indicates positions occupied by nylon sea cages (S1 - S9) and zinc-coated sea cages (fixed at S10 & S11).



Control Sites

Seven control sites were selected in various locations around Botany Bay (Figure 3.1.2), these had similar characteristics to the sea cage site. The airport and container terminal areas (Figure 3.1.2), where recent major sediment disturbances had occurred due to dredging and construction works, were excluded as control site areas. Each control site has been numbered and named according to the nearest landmark or location. The physical characteristics, locations, and exact DGPS co-ordinates for all Control sites for are shown in Table 3.1.and Table 3.2.

Table 3.1 DGPS coordinates of Control sites. Landmark location information was also recorded and archived for future reference if required.

Control Site	Location	Latitude	Longitude
1	Frenchman's Bay	33^0 59' 17.194207916 "	151^0 13' 43.894326842 "
2	Silver beach	34^0 00' 18.880696575 "	151^0 11' 25.655354695 "
3	Bonna Point	34^0 00' 19.944901524 "	151^0 11' 37.116276487 "
4	Towra Point	34^0 00' 07.183064476 "	151^0 10' 39.078427196 "
5	Ramsgate	33^0 59' 08.060648523 "	151^0 08' 52.738693702 "
6	Brighton Le-Sands	33^0 57' 46.954659183 "	151^0 09' 23.931017513 "
7	Kyeemagh	33^0 57' 13.184710809 "	151^0 09' 50.402614961"

Table 3.2 Chemical, physical and biological characteristics of control sites in Botany Bay, chosen as part of studies of impacts of sea cages.

Site No	Location	Sampling location	Depth	Sediment characteristics	Flora	Fauna
1	La Perouse in Frenchman's Bay	A commercial mooring is the landmark used to identify the exact location for sampling, being 10meters due South of the mooring block	Approximately 6 metres deep	Mainly coarse grained sand, , consistent for first 5-10 cm of sediment across sampling site.	Mainly barren with very sparse patches of sea grass *Halophila* sp. The abundance of which varies as it is affected by heavy wave action and consequent sediment movements.	No fish or mammals were sighted during sampling
2	Silver beach Kurnell	The exact sampling location is found by using the alignment of two visual marks being the fourth rock groin off Silver Beach and the sea cage sites 1 to 3 on the most northern front of the fish farm.	Approximately 5 metres	Mainly coarse grained sand, some fine sand and very little silt, consistent for first 5-10 cm of sediment across all of Control site	Sparse patches of sea grass *Halophila* Sp. The abundance of which varies as it is affected by heavy wave action and consequent sediment movements.	Some fish were sighted at this location during sampling.
3	Bonna point, Silver beach	The exact sampling location is identified by measuring a distance of 237 meters due North from a special navigation mark, parallel to the sea cage site from silver beach.	Approximately 6 metres deep	Mainly coarse grained sand, some fine sand and very little silt, consistent for first 5-10 cm of sediment across all of Control site.	Mainly barren with very sparse patches of sea grass *Halophila* sp. The abundance of which varies as it is affected by heavy wave action and consequent sediment movements.	No fish were observed during sampling.
4	Towra point at the Southern entrance to the George's river	The exact location is found by the alignment of a number of visual landmarks in three directions around the site.	The depth is approximately 4 metres at mean low water.	Mainly coarse grained sand, some fine sand and some silt, consistent for first 5-10 cm of sediment across all Control site.	Patches of sea grass *Halophila* sp. The abundance of which varies as it is affected by heavy wave action and consequent sediment movements.	No fish or mammals were sighted during sampling.
5	Ramsgate beach	The exact location for sampling is 10 meters due east of the most northern corner pylon of the baths.	The depth across the farm site is 7 metres at mean low water.	The sediments are mainly fine-grained sand with some silt present.	Sparse patches of sea grass *Halophila* sp. The abundance of which varies as it is affected by heavy wave action and consequent sediment movements.	No fish or mammals were sighted during sampling
6	Brighten Le-Sands beach	A large State Emergency Service buoy is used to identify the location for sampling. The exact position is 10 metres due east of the mooring block for the buoy.	The depth is 6 metres at mean low water.	The sediments are fine-grained sand and silt.	No sea grasses present The seafloor is barren	No fish or mammals were sighted during sampling
7	Kyeemagh near the entrance to the Cooks River	The most southern pylon of the swimming baths, the exact position being 10 meters due east of this pylon.	The depth across the farm site is 5 metres at mean low water.	The sediments at this location is fine silty sand	No sea grasses present. The seafloor is barren	No fish or mammals were sighted during sampling.

3.2.2 Sampling procedures for TP, TOC & zinc in sediments

There were nine nursery cage sites on the fish farm (see Figures 3.2 & 3.3). A total of three sea cages were seasonally located on these nine sites. Each of these sites was sampled on one occasion prior to the placement of any sea cages on the farm, after which sampling occurred every four months, prior to the movement of a cage to a new location for fallowing (see Table 3.3).

Sediment core samples were collected from all impact and control sites over a 24-hour period. Samples of sediment were taken from directly beneath the cages in use and in the exact location that the cages would occupy on the site. Six replicate sediment samples were collected for analysis from each control site.

Only the first two centimetres of sediment were collected for the TOC, TP and zinc analyses. These samples were also used for sediment grain grading (see Table 3.2).

Table 3.3 Position of sea cages on each site at each time interval that sediment sampling occurred and selected sites sampled for TP and TOC.

N.B. The occupied sites are colour coded to identify movements and distinguish between the two sea cage platforms. (Period of occupation is 4 months).

Date	Sample Period	Nylon Sea Cage Sites			Wire Sea Cage Site
		S2	S5	S8	S10
Jan 98	1	(SAMPLED TOC, TP)			
May 98	2	OCCUPIED (SAMPLED TOC, TP)	(SAMPLED TOC, TP)	(SAMPLED TOC, TP)	(SAMPLED TOC, TP)
Sept 98	3	(SAMPLED TOC, TP)	OCCUPIED (SAMPLED TOC, TP)	(SAMPLED TOC, TP)	
Jan 99	4	(SAMPLED TOC, TP)	(SAMPLED TOC, TP)	OCCUPIED (SAMPLED TOC, TP)	OCCUPIED
May 99	5	OCCUPIED (SAMPLED TOC, TP)	(SAMPLED TOC, TP)	(SAMPLED TOC, TP)	OCCUPIED (SAMPLED TOC, TP)

3.2.3 Loss of zinc coating from wire sea cages

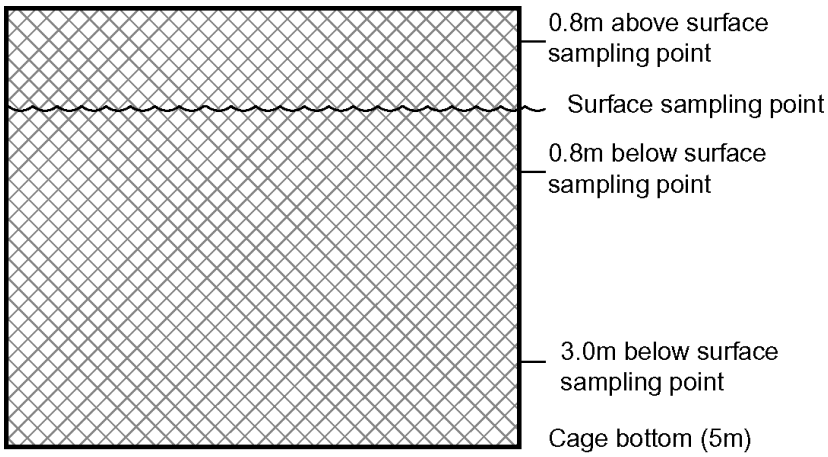
The zinc coated sea cages were comprised of five panels (including the base) of wire mesh. Each of the four sidewalls was virtually submersed, approximately 1m of mesh was held above the waterline to stop fish from escaping.

To estimate the depletion of zinc coating, pieces of wire mesh were removed from a sea cage after a period of 26 months and examined to determine the amount of zinc remaining on the wire itself. This period was considered to be the full life span of the wire sea cages. The mass of zinc lost and the approximate rate of depletion can be easily estimated by comparison to new zinc mesh. These factors are important in the overall environmental assessment of the cages, as the lower the release rate of zinc into the water, the greater the flux of zinc from water currents.

Samples of wire mesh were collected by SCUBA diving by removal of sections from the north-facing panel of the cage. The sampling strategy was established by randomly selecting samples of wire mesh from the side of the sea cage that was divided into three horizontal sections (Figure 3.4).

Figure 3.3 Zinc coated wire sea cage

Figure 3.4 Side wall of zinc coated wire sea cage showing sampling locations



3.2.4 Analytical procedures

Chemical analyses of sediment samples for TP and zinc were carried out by the University of Canberra using standard laboratory methods. Briefly, the sediments were acid digested following the methods of Baldwin et al. (1997) and Deaker et al. (1997). A Perkin-Elmer Elan 6000 Inductively Coupled Plasma Mass Spectrometer (ICP-MS) was then used to determine the concentration of zinc in the sediment digests. Standard Reference Material (SRM) was also routinely digested throughout the analytical phase to document the recovery rates and ensure quality control purposes.

The analyses of sediment samples for TOC were carried out, on a contract basis, by a NADA registered laboratory (see SBA 1999). A standard incineration method was used to recover and interpret the TOC % in sediment samples.

The analysis for estimated loss of zinc coating from the wire sea cages was done by OneSteel Ltd laboratories.

3.2.5 Statistical procedures

The results obtained for sediment TOC, TP and zinc have been analysed to assess changes occurring at selected sea cage sites. The statistical analyses was designed to test for before and after impacts, cumulative, long term impacts and recovery after impact. The design chosen for TOC and TP included impacted sites 2, 5, 8 and 10, and seven control sites, before and after the time that a sea cage occupied the site (See Table 3.3).

Ideally, the design for the analysis would involve replicated impacted sites and control sites. However, during the study, there was only one or two occupations of each of the sites (S1 to S9) by a sea-cage, so the design becomes asymmetrical with respect to these impacted sites. To overcome this problem and provide greater statistical power a larger number (7) of control sites was used. With respect to impact Site 8, the analysis is a 3 factor ANOVA, as the sea cage had occupied this site on two occasions. The Mean Squares (MS) were analysed and pooled with the Residual MS to provide greater degrees of freedom and statistical power in the analysis. Homogeneity of data was checked prior to ANOVA, but even when this assumption was not met, the

analyses was carried out as ANOVA is relatively robust in cases where numbers of samples is high (Underwood 1981, 1997).

3.3 Results

This section presents results of the studies of impacts on sediments of the sea-cage culture of fish at Silver Beach

3.3.1 Total Organic Carbon in sediments

Each of the following sections presents data for impact sites compared to control sites where TOC has been measured before and after the impact (i.e., sea-cage culture) (See Fig 3.5). Results of statistical analysis for changes in TOC at each of the impact sites are described here. The analysis for Sites 2, 5 and 10 are two factor ANOVAs, Site 8 was a 3 factor ANOVA.

Impact Site 2.

Results from the analysis for impact Site 2 (Table 3.4a), indicate there was no significant difference among sites and therefore no impact has occurred through time at Site 2. However Time (T) among Controls (C) is significant 2.89>2.42 (T x Amg C, Table 3.4a), indicating there is a significant difference among control sites through time.

In the second analysis, T x I (Impact) v C (30.98> 5.59) is significant ($P<0.05$)(Table 3.4b), which indicates a change has occurred between the two sampling times, however this may not be indicative of impact. The result may be due to the fact that there is only one sample set taken "before" time, prior to the impact, or more likely because there was a significantly greater concentration of TOC in the sediments at this sea cage site compared to the control sites in the "before" period (i.e., prior to the deployment of any sea-cage). This is suggested by the significant *F*-ratio for the Times Among Controls (T x Amg C) (Tables 3.4a & 3.4b). The concentration of TOC in the sediments at the sea-cage site was significantly greater in January 1998 sediments compared to all times sampled after the impact time (Fig 3.3). Moreover, the overall mean concentration of TOC at the sea cage Site 2 fluctuated through time at a lower concentration than that observed in the "before" period, indicating no impact or significant changes had occurred in the level of TOC at this time.

Impact Site 5.

Results for Site 5 were similar to that observed at Site 2. In the first analysis there is no significant difference among Time or Sites (Table 3.5a). However, there is a significant difference, and obvious variation among Control Sites. In the second analysis there is no significant difference among Time or Sites (Table 3.4b), indicating no impact has occurred at Site 5.

Impact Site 8.

The analysis for Site 8 was a 3 factor ANOVA, where sampling occurred twice before and twice after the placement of a sea cage on the site. The results also indicated no significant difference in TOC among Time or Sites. The data was further manipulated by pooling the MS Residual with the MS of Control sites to increase the degrees of freedom and therefore the statistical power of the test. The result (0.778) was also not significant, again suggesting that no impact has occurred at Site 8.

Impact Site 10.

There was no significant difference in TOC among Time or Sites at Site 10 (Table 3.6) again indicating no impact has occurred. As in Sites 2 and 5, there was a significant difference among control sites (44.31>2.42) for T x Amg C (Table 3.6).

Table 3.4a. Asymmetrical analysis of variance comparing the concentration of TOC in the sediments at sea cage Site 2 (I) with all control sites (C) sampled before (B) and twice after (A) deployment of the sea cage.

Time 1 (B) vs Time 2 (A) + C1 –C7 (Time 1 & Time 2)

Note: untransformed data used in analysis.

TOC

Site 2: A1, A2		**TOC**						
S of V		**DF**	**SS**	**MS**	**SS**	**MS**	**F**	**P**
Time		1	A1	T x Amg C	0.01	1.35E-01	0.28	ns
Sites					0.573333333			
	I v C	1	A1-A2	Amg C	0.373333333	1.866666665	9.768889	5.59
	Amg C	7	A2	Resid	0.2	0.191082802	5.842022	2.33
T x S					0.146666667			
	T x I v C	1	A1-A2	T x Amg C	0.047619048	4.81E-01	5.08	ns
	T x Amg C	6	A2	Resid	0.10	0.09	2.89	2.42
Resid		32	A1		1.046666667	0.032708333		
Total		48						

Table 3.4b Asymmetrical analysis of variance comparing the concentrations of TOC in sediments at sea cage Site 2 with all control sites sampled before (B) and after (A) deployment of the sea cage, after two occupations, over a long period of time at site 2.

Time 1 (B) vs Time 5 (A) + C1 –C7 (Time 1 & Time 5)

Note: untransformed data used in analysis.

TOC

Sites 2, A3, A4		**TOC**						
S of V		**DF**	**SS**	**MS**	**SS**	**MS**	**F**	**P**
Time-T		1	A1	T x Amg C	0.140833333	2.07	22.52	5.59
Sites-S					0.235833333			
	I v C	1	A1-A2	Amg C	0.16297619	2.2369281	24.357662	5.59
	Amg C	7	A2	Resid	0.07	0.091836735	3.7043389	2.33
T x S					0.249166667			
	T x I v C	1	A1-A2	T x Amg C	0.181071429	2.66	30.98	5.59
	T x Amg C	6	A2	Resid	0.07	8.58E-02	3.46	2.42
Resid		32	A1		0.793333333	0.024791667		
Total		48						

Table 3.5a Asymmetrical analysis of variance comparing the concentration of TOC in the sediments at sea cage site 5 with all control sites sampled twice before (B) and twice after (A) deployment of the sea cage.

Time 2 (B) vs Time 3 (A) + C1 – C7 (Time 2 (B) vs Time 3 (A))

Note: untransformed data used in analysis.

S of V		DF	SS	MS	SS	MS	F	P
Time-T		1	A1	T x Amg C	7.50E-03	5.53E-02	0.07	ns
Sites -S					0.3625			
	I v C	1	A1-A2	Amg C	0.005833	0.016355139	0.0207	ns
	Amg C	7	A2	Resid	0.356666	0.786764707	55.5363	2.33
T x S					0.135833			
	T x I v C	1	A1-A2	T x Amg C	0.000119	8.77E-04	2.93E-03	ns
	T x Amg C	6	A2	Resid	0.135714	0.30	21.13	2.42
Resid		32	A1		0.453333	0.014166667		
Total		48						

Table 3.5.b Asymetrical analysis of variance comparing the concentrations of TOC in sediments at sea cage Site 5 with all control sites sampled before (B) and after (A) deployment of the sea cage, after two occupations, over a long period of time at site 5.

Time 2 (B) vs Time 5 (A) + C1 –C7 (Time 2 & Time 5) .

Note: untransformed data used in analysis.

S of V		DF	SS	MS	SS	MS	F	P
Time-T		1	A1	T x Amg C	0.140833333	1.01	1.75	ns
Sites-S					0.235833333			
	I v C	1	A1-A2	Amg C	0.16297619	0.03218	0.05567	ns
	Amg C	7	A2	Resid	0.07	0.578042		
T x S					0.249166667			
	T x I v C	1	A1-A2	T x Amg C	0.181071429	1.55E-03	3.23E-03	ns
	T x Amg C	6	A2	Resid	0.07	4.80E-01	42.68	
Resid		32	A1		0.793333333	0.01125		
Total		48						

Table 3.6 Asymmetrical analysis of variance comparing the concentration of TOC in the sediments at sea cage Site 10 with all control sites sampled before (B) and after (A) deployment of the sea cage.

Time 2 (B) vs Time 5 (A) + C1 –C7 (Time 2 & Time 5)

Note: untransformed data used in analysis.

Site 10: A1, A2

S of V		DF	SS	SS	MS	F	P
Time-T		1	A1	0.12	6.94E-01	1.18	ns
Sites-S				0.2225			
	I v C	1	A1-A2	0.014405	0.069222	0.117535	ns
	Amg C	7	A2	0.208095	0.588949	53.33876	2.33
T x S				0.203333			
	T x I v C	1	A1-A2	0.030476	1.76E-01	3.60E-01	ns
	T x Amg C	6	A2	0.172857	4.89E-01	44.31	2.42
Resid		32	A1	0.353333	0.011042		
Total		48					

Table 3.7 Asymmetrical analysis of variance comparing the concentration of TOC in the sediments at sea cage Site 8 with all control sites sampled twice before (B) and twice after (A) deployment of the sea cage.

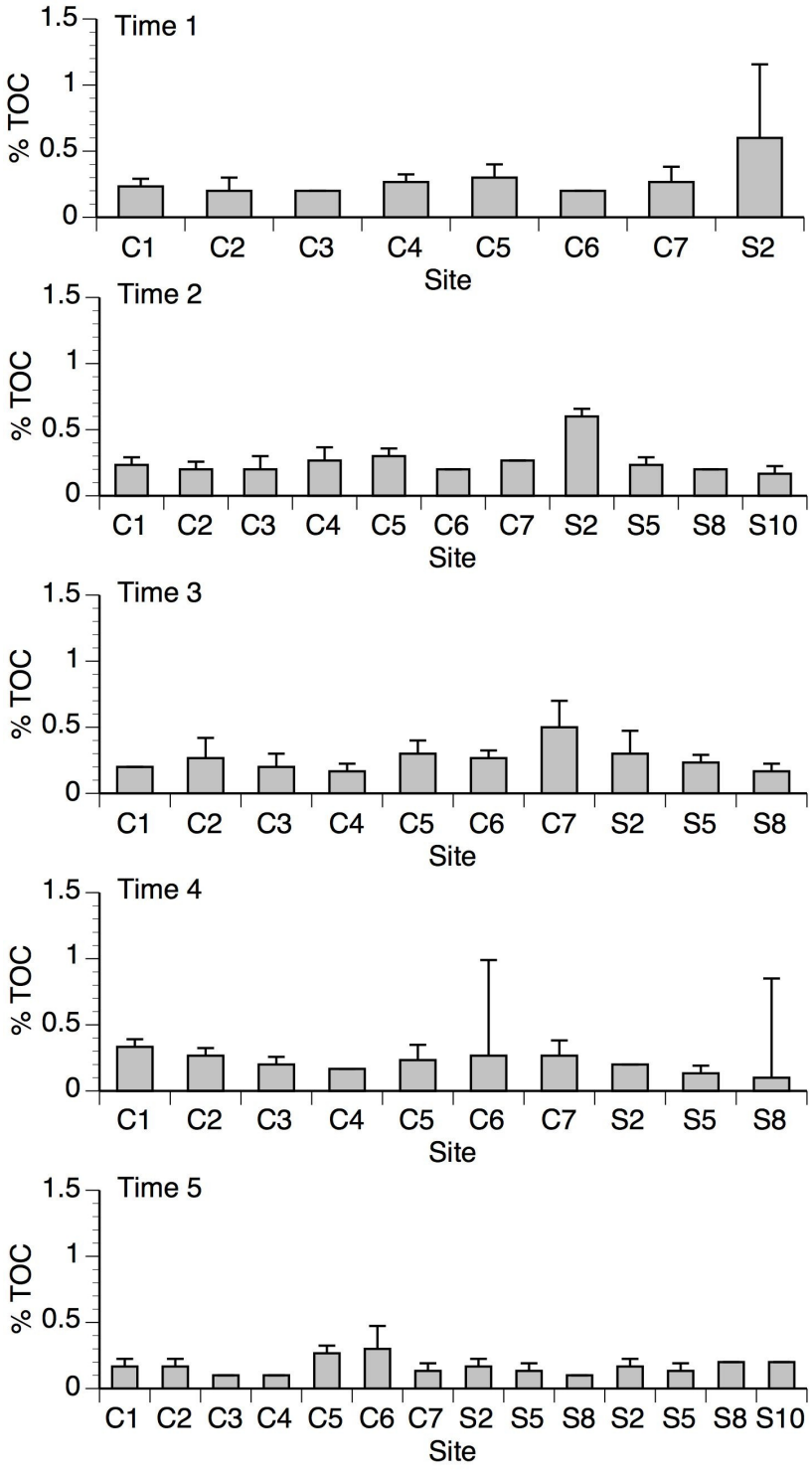
Time 2 + Time 3 (B) vs Time 4 (A) + Time 5 + C1 –C7 (Time 2, 3, 4 & 5)

Note: untransformed data used in analysis.

Site 8

S of V	Df	SS	SS	MS	F	P
B v A	1	A1	3.75E-03			
Time-T(BA)	2	A1	0.308333333			
Sites-S	7					
I v C	1					
Amg C	6					
BA x Sites	7		0.54625			
BA x I v C	1	A1-A2	0.207202381	0.207202	3.6668	ns
BA x Amg C	6	A2	0.339047619	0.056508	0.6911	ns
T(BA) x Sites	14		0.348333333	0.024881		
T(BA) x I v C	2	A1-A2	0.129761904	0.064881		
T(B) x I v C	1	A3-A4	0.000119048	0.000119		
T(A) x I v C	1	(A1-A2)-(A3-A4)	0.129642856	0.129643	1.5856	ns
T(BA) x Amg C	12	A2	0.218571429	0.018214		
T(B) x Amg C	6	A4	0.135714286	0.22619		
T(A) x Amg C	6	A2-A4	0.082857143	0.01381	0.1689	ns
Resid	34	A1	2.78	0.081765	(pool) 0.77975	ns
Total	107					

Figure 3.5 Mean total organic carbon levels through time at selected sea cage sites and all control sites. Bars represent standard deviation. Time 1: Jan 98, Time 2: May 98, Time 3: Sept 98, Time 4: Jan 99 and Time 5: May 99. Note extra replicates were taken in May 99 for sites S2, S5 and S8.



The graphical representation of results in Figure 3.3 provides a simple observation through time of the mean %TOC for all sea cage sites including both impact and control sites. Although variability is high, suggesting that more replication of samples may be required to provide more robust analyses, it should be noted that all results for TOC were less than 0.7 %. A detailed statistical analysis of these results was presented in Tables 3.5, 3.6 and 3.7

3.3.2 Total Phosphate in sediments

Phosphate sediment samples were collected at the same sampling time as TOC samples, and have been expressed as ug /g (or ppm) Phosphate. These data for TP are dealt with in more detail below and presented graphically in Figure 3.6.
A two-factor ANOVA was used for sites 2, 5 and 10, and a three-factor ANOVA was used for site 8.

Impacted Site 2.

Results of the analysis for Site 2 (Table 3.8a) indicated that no significant differences had been found among Time, Sites or Controls. In the second analysis (Table 3.8b) there is a significant difference among “impacted” sites indicating a change may have occurred through Time. However, again, this result does not necessarily indicate that an impact has occurred as there are also significant differences among Control sites. Similar to results for TOC at Site 2, the levels of TP varied greatly both spatially and temporally (Fig 3.6, Time1-5).

Impacted Site 5.

There was no significant difference among Time or Sites (Tables 3.9a and 3.9b), indicating that no significant differences in TP had occurred at site 5 (see Figure 3.6, Time 1-5).

Impacted Site 8.

The analysis for site 8 was a 3 factor ANOVA, where sampling occurred twice before and twice after the placement of a sea cage on the site. Results indicate there was no significant difference among Time or Sites (Table 4.1), indicating that no impact has occurred form P at site 8. (see Figure 3.6, Time 1-5)

Impacted Site 10.

There was no significant difference among Time or Sites (Table 4.0), indicating that no impact has occurred form P at site 5 (see Figure 3.6, Time1-5).

Table 3.8.a Asymmetrical analysis of variance comparing the concentration of TP in the sediments at sea cage Site 2 with all control sites sampled before (B) and after (A) deployment of the sea cage.

Time 1 (B) vs Time 2 (A) + C1 –C7 (Time 1 & Time 2)

Note: untransformed data used in analysis.

TP Site 2: A1, A2

S of V		DF	SS	MS	SS	MS	F	P
Time-T		1	A1	T x Amg C	4591.92362	5.97E-02	5.54E-03	ns
Sites-S					1568157.206			
	I v C	1	A1-A2	Amg C	92499.048	0.062683249	0.005814	ns
	Amg C	7	A2	Resid	1475658.158	10.781345	0.002521	ns
T x S					79142.53521			
	T x I v C	1	A1-A2	T x Amg C	2228.17275	2.90E-02	5.16E-02	ns
	T x Amg C	6	A2	Resid	76914.36246	5.62E-01	1.31E-04	ns
Resid		32	A1		136871.4347	4277.232334		
Total		48						

Table 3.8.b Asymmetrical analysis of variance comparing the concentrations of TP in sediments at sea cage Site 2 with all control sites sampled before (B) and after (A) deployment of the sea cage, after two occupations, over a long period of time at Site 2.

Time 1 (B) vs Time 5 (A) + C1 –C7 (Time 1 & Time 5)

Note: untransformed data used in analysis.

TP Site 2

S of V		DF	SS	MS	SS	MS	F	P
Time-T		1	A1	T x Amg C	0.140833333	2.07	22.52	5.59
Sites-S					0.235833333			
	I v C	1	A1-A2	Amg C	0.16297619	2.2369281	24.357662	5.59
	Amg C	7	A2	Resid	0.07	0.091836735	3.7043389	2.33
T x S					0.249166667			
	T x I v C	1	A1-A2	T x Amg C	0.181071429	2.66	30.98	5.59
	T x Amg C	6	A2	Resid	0.07	8.58E-02	3.46	2.42
Resid		32	A1		0.793333333	0.024791667		
Total		48						

Table 3.9.a Asymmetrical analysis of variance comparing the concentration of TP in the sediments at sea cage Site 5 with all control sites sampled before (B) and after (A) deployment of the sea cage.

Time 2 (B) vs Time 3 (A) + C1 –C7 (Time 2 & Time 3)

Note: untransformed data used in analysis.

TP Site 5: A1, A2

S of V		DF	SS	SS	MS	F	P
Time-T		1	A1	5996.209764	4.45E-02	3.14E-03	ns
Sites-S				1750367.238			
	I v C	1	A1-A2	52913.707	0.031172404	0.002195	ns
	Amg C	7	A2	1697453.531	14.20474533	0.003804	ns
T x S				134634.3983			
	T x I v C	1	A1-A2	28.1285	2.09E-04	1.86E-04	ns
	T x Amg C	6	A2	134606.2698	1.13E+00	3.02E-04	ns
Resid		32	A1	119499.0471	3734.345222		
Total		48					

Table 3.9.b Asymmetrical analysis of variance comparing the concentrations of TP in sediments at sea cage Site 5 with all control sites sampled before (B) and after (A) deployment of the sea cage, after two occupations, over a long period of time at Site 5.

Time 2 (B) vs Time 5 (A) + C1 –C7 (Time 2 & Time 5)
Note: untransformed data used in analysis.

TP Site 5: A3, A4

S of V		DF	SS	SS	MS	F	P
Time-T		1	A1	88702.02	0.69	0.03	ns
Sites-S				2239906			
	I v C	1	A1-A2	157714.3	0.075744	0.003717	ns
	Amg C	7	A2	2082192	20.37799	0.006382	ns
T x S				154671.5			
	T x I v C	1	A1-A2	26178.98	0.20	0.16	ns
	T x Amg C	6	A2	128492.5	1.26	3.94E-04	ns
Resid		32	A1	102178.5	3193.078		
Total		48					

Analysis of Variance (Site10) TP

Table 3.11 Asymmetrical analysis of variance comparing the concentration of TP in the sediments at sea cage Site 10 with all control sites sampled before (B) and after (A) deployment of the sea cage.

Time 2 (B) vs Time 5 (A) + C1 –C7 (Time 2 & Time 5)

Note: untransformed data used in analysis.

TP Site 10: A1, A2

S of V		DF	SS	SS	MS	F	P
Time		1	A1	96554.51	0.75	0.03	ns
Sites				2244314			
	I v C	1	A1-A2	162122.2	0.077861	0.003274	ns
	Amg C	7	A2	2082192	23.78215	0.008692	ns
T x S				144789.6			
	T x I v C	1	A1-A2	16297.1	1.27E-01	4.64E-05	ns
	T x Amg C	6	A2	128492.5	1.47	5.36E-04	ns
Resid		32	A1	87552.73	2736.023		
Total		48					

Analysis of Variance (Site8) TP

Table 3.12 Asymmetrical analysis of variance comparing the concentration of TP in the sediments at sea cage Site 8 with all control sites sampled twice before (B) and twice after (A) deployment of the sea cage.

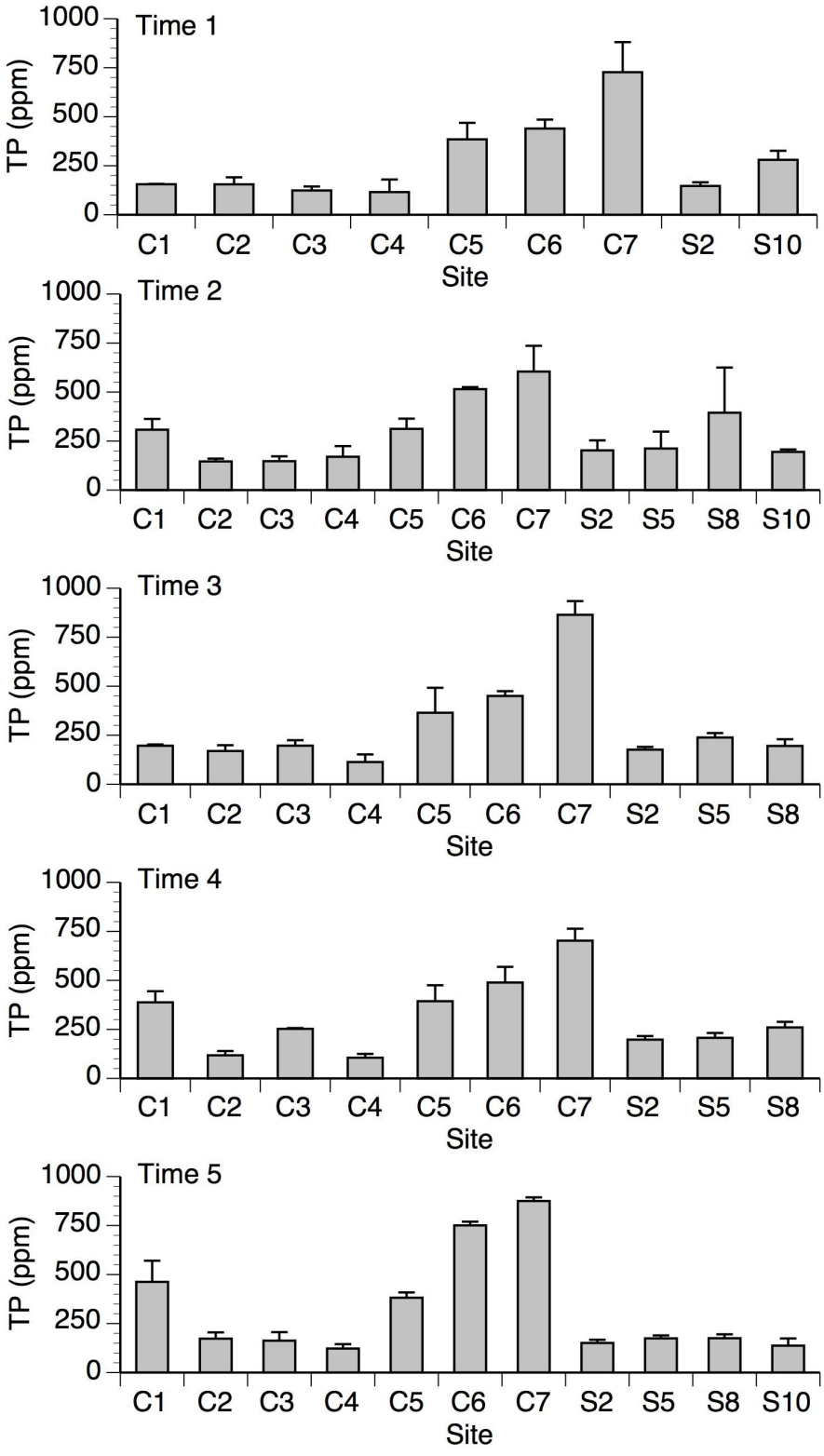
Time 2 & Time 3 (B) vs Time 4 & Time 5 (A) + C1 – C7 (Time 2, 3 (B) & Time 4, 5 (A)

Note: untransformed data used in analysis.

Site 8

S of V		Df	SS	SS	MS	F	P
B v A		1	A1	41430.1767			
Time(BA)		2	A1	29513.7872			
Sites		7					
	I v C	1					
	Amg C	6					
BA x Sites		7		145850.295			
	BA x I v C	1	A1-A2	48543.164	48543.16	2.993	ns
	BA x Amg C	6	A2	97307.131	16217.86	1.907	ns
T(BA) x Sites		14		353523.282	25251.66		
	T(BA) x I v C	2	A1-A2	94885.561	47442.78		
	T(B) x I v C	1	A3-A4	64101.135	64101.14		
	T(A) x I v C	1	(A1-A2)-(A3-A4)	30784.426	30784.43	3.6204	ns
	T(BA) x Amg C	12	A2	258637.720	21553.14		
	T(B) x Amg C	6	A4	134606.269	22434.38		
	T(A) x Amg C	6	A2-A4	124031.451	20671.91	1.5855	ns
Resid		34	A1	289099.421	8502.924		
Total		107					

Figure 3.6 **Mean total phosphate levels through at selected sea cage sites and all control sites. Bars represent standard deviation. Time 1: Jan 98, Time 2: May 98, Time 3: Sept 98, Time 4: Jan 99 and Time 5: May 99.**



The graphical representation of results in Figure 3.6 provides a simple observation through time of the mean TP (ppm) for selected sea cage sites compared with the seven control sites. Although there was high variability in sediment TP, suggesting that more replication of samples may be required to provide a more robust analyses, it should be noted that results for sediment TP were very similar between control and impact sites, for all time periods.

3.3.3 Zinc in sediments

Ideally, an assessment of any possible impacts of zinc from the wire sea cages would be based on replicated before and after control and impact sample design. However, as there was only one impact site, the experimental design was asymmetrical. Control sites were again replicated in this design and this had a large influence on the power of the tests for impact. Table 3.14 shows the design that was used to analyse the temporally-replicated data. Note that all terms denoted with an asterisk in this table involve an asymmetry in the levels of the particular factor.

This design has several important features that overcame problems that have beset approaches to impact assessment in the past (Otway 1995, Otway et al. 1996a, b). First, the design incorporates spatial and temporal replication thus overcoming problems of pseudo-replication (Hurlbert 1984). Second, temporal replication (before and after the disturbance) is done at a time before and at several times after, to identify temporal trajectories (also see discussion in Stewart-Oaten *et al*. 1986). Third, the design detects whether disturbances, in this case the loss of zinc from the wire sea-cages, causes detectable changes in the variable of interest at the impact site (i.e. the fish farm site). Last, the design detects impacts that occur at different temporal scales, that is, as 'pulse' or sustained (short-term longer-term) 'press' changes (Bender et al. 1984).

The repartitioning of the asymmetrical analysis of variance in Table 3.14 provides for temporal interactions with an *a priori* orthogonal contrast between the single putatively-impacted site and the control sites before and after the disturbance begins. It is this feature that permits tests for impact. The detection of impact depends on the duration of the changes caused by the disturbance and the space-time interactions,

which occur naturally, that is, in the absence of an anthropogenic disturbance. The detection of impacts at different temporal scales requires several tests and these are described below. First, a persistent impact is detected using the *F*-ratio of MS B v A x SC v Controls / MS B v A x Among C giving a test with 1 and 6 degrees of freedom (i.e. with 7 control sites sampled). Second, if there is no significant variation from 'before' to 'after' among the control sites (there are no short-term temporal interactions among the control sites). If the *F*-ratio of MS (B v A) x Among C/ MS Residual is not significant at $P = 0.25$, then the MS (B v A) x Among C term can be eliminated (pooled with the Residual) from the analysis. This results in a test with substantially more power, as the MS Residual and its associated degrees of freedom (an order of magnitude greater) are then used in the *F*-ratio for assessing impact.

An intermittent impact can be detected in one of two ways. First, when there are no short-term temporal interactions among the control sites after the disturbance begins, i.e. the F-ratio MS Times (After) x Among Controls / MS residual is not significant in Table 3.4, a short-term impact is indicated when the *F*-ratio of MS Times (After) x SC v Controls / MS residual is significant and the two-tailed *F*-ratios of MS Times (After) x SC v C / MS Times (Before) x SC v C and MS Times (After) x Among C /MS Times (Before) x Among C are significant and not significant, respectively. Second, when there are significant short-term temporal interactions among the control sites after the disturbance begins, i.e. the *F*-ratio MS Times (After) x Among C / MS residual is significant in Table 2, a short-term impact is evident when the F-ratio of MS Times (After) x SC v C / MS Times (Before) x Among C is significant and the two-tailed *F*-ratios described above are significant and not significant, respectively.

While the design maybe considered an advancement on the previous BACI (Before-After/Control-Impact) designs of Green (1979), Bernstein & Zalinski (1983) and Stewart-Oaten *et al*. (1986), there may still be some problems. Some of the tests may have low statistical power and thus, the probability of making a Type II error may be large. This could be overcome by relaxing the Type I error-rate of 0.05 to 0.10 as this will result in increased power and make the detection of impacts less conservative.

There was a significantly higher concentration of zinc in the sediments at the sea cage site compared to the control sites in the "before" period (i.e., prior to the deployment of the sea-cage) as evidenced by the significant *F*-ratio for the Times (Before) x SC v C term (Table 3.14). The concentration of zinc in the sediments at the sea cage site was significantly greater in January 1998 sediments compared to May 1998 (Table 3.14). Moreover, the greater overall mean concentration of 168.90 ppm at the sea cage site in the "before" period was significantly greater than the mean of 71.59 ppm at the control sites and responsible for the significant
B v A X SC v Controls *F*-ratio (Table 3.14).

The concentrations of zinc in the sediments at the sea cage site compared to the control sites after the deployment of the wire sea cage did not differ significantly (Table 3.14). There was also significant short-term temporal variation in the concentration of zinc in the sediments at the control sites, as indicated by the significant Times (After) x Among C *F*-ratio (Table 3.14). The concentrations of zinc at the control sites varied between 10.84 mg/kg and 177.42 ppm in January 1999 and 2.58 ppm and 282.87 ppm in May 1999.

(Note: that these results have been reported previously by Barker and Otway 2003).

Table 3.13 Mean concentrations of zinc (mg/kg) in the sediments at the sea cage site and control sites sampled twice before and twice after deployment of the wire sea cage.

Time		All Control Sites	Sea-Cage Site
Before			
	1	60.83	280.23
	2	79.98	57.56
After			
	1	46.49	97.31
	2	95.92	74.20

Table 3.14 Asymmetrical analysis of variance comparing the concentration of zinc in the sediments at the sea cage site with all control sites sampled twice before (B) and twice after (A) deployment of the wire sea cage.

Source of variation	df	MS		*F*	*P*
Before vs After = B vs A	1	2255			
Times(Before vs After) = Times(B vs A)	2	10508			
Sites	7				
Sea cage vs Controls = SC vs C	1	33547	0.55		ns
Among Controls = Among C	6	61547		29.00	**
B vs A X Sites	7				
B vs A X SC vs C	1	18497	14.68		**
B vs A X Among C	6	1260	1.42		ns
Times(B vs A) X Sites	14				
Times (B vs A) X SC vs C	2				
Times (B) X SC vs C	1	76752	74.66		**
Times (A) X SC vs C	1	7380.28			ns
Times (B vs A) X Among C	12				
Times (B) X Among C	6	1028	1.17		ns
Times (A) X Among C	6	2669	3.03		**
Residual	64	882			
Total	95				

A. Tests for Intermittent Impact

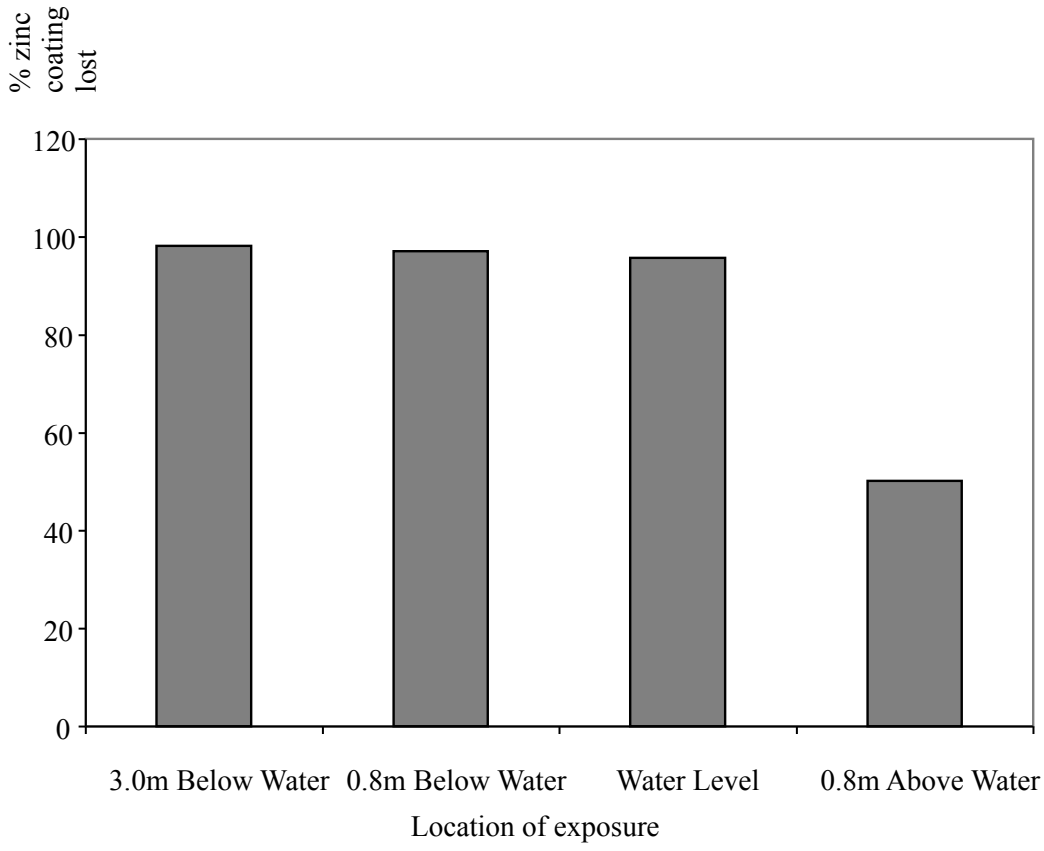
1-tailed test: MS Times(A) X SC vs C / MS Times(A) X Among C = 0.28 ns

2-tailed tests: 1. MS Times (A) X SC vs C / MS Times (B) X SC vs C = 0.01 ns
2. MS Times (A) X Among C / MS Times (B) X Among C = 2.60 ns

3.3.4 Depletion of Zinc coating

Results indicated that the zinc coating had depleted to a low level with some samples revealing that virtually all the coating had been removed from most samples (Figure 3.7). As expected, the sections of cage above the water line had also corroded but had substantial quantities of coating remaining. The fact that most of the zinc had depleted but some coating was still present on the wire would suggest that after nearly two years in use, the sea cage is very close to the end of its useable life span.

Figure 3.7 Zinc (%) lost from each observed region of MarineMesh™ sea cage.



3.4 Discussion

TOC and TP were chosen as indicators of a detectable impact in the sediment in the form of an increase in nutrients, due to the culture of fish in the sea cages at the Silver Beach Site. The results for both TOC and TP showed a wide range of concentrations in the sediments at the impact (cage) sites and control sites, and these concentrations fluctuated through time (see Fig 3.3 & 3.4). However, the levels of TOC and TP in the surficial sediments beneath the sea cages showed no significant increase through time compared to the seven control sites.

There was however a significant difference in levels of both TOC and TP among control sites, indicating fluctuations in these variables occurs throughout the Bay as a whole. These fluctuations in the concentrations of TOC and TP in the sediments at control sites are likely to be the result of natural spatial and temporal variations in the sediments in Botany Bay, which was greater than any change at the impact sites.

The results also indicated a range of zinc concentrations in the sediments at the control sites and cage site through time. These levels were all well within the range of zinc concentrations known to occur in the sediments in the Sydney region (Davies 1978, Gray 1995) and elsewhere (Chester & Stoner 1975, Everaarts 1989, Batley & Brockbank 1990). The levels of zinc in the surficial sediments beneath the sea cage (Impact) Site showed no significant (and consistent) increase through time, and were within the range of zinc concentrations exhibited at these latter sites. This clearly indicates that, within the proximity of the commercial fish farm, the fluctuations in the concentrations of zinc in the sediments at the control sites were most likely to be the result of natural spatial and temporal variations.

Finally, while the zinc-coated wire sea cages did not appear to contribute to a detectable accumulation of zinc within the sediments directly beneath the cages, the potential for this to occur in other more ecologically-sensitive areas could be even further reduced by ensuring that the wire sea-cages are cathodically protected (see, OneSteel 2003 for details).

The life-span of standard zinc coated wire used by fish farmers in Japan was approximately 12 months (OneSteel pers comm 1998). Although the conditions where, sea cages are situated in Japan may be different to those where the zinc coated wire cages were studied in Botany Bay, Australia, it would be prudent to say that the zinc coated wire sea cages would generally have a far greater life expectancy than standard zinc coated wire mesh. It is also possible that the addition and use of a cathodic protection system could further increase the life expectancy of the zinc coated MarineMesh™. (see, OneSteel 2003).

Overall, the results from this research indicated that the zinc coated wire cages have a greater lifespan than uncoated wire mesh cages used in Japan, and that no environmental impacts were found as a result of the zinc coating. The zinc coating also acted as an antifouling agent, significantly reducing the growth of fouling organisms, compared to other mesh types tested (see Chapter 5).

CHAPTER 4. FURTHER IMPACTS OF ZINC

4.1 Introduction

The corrosion of the zinc coating on the wire sea cages occurs at the interface of the coating and the water. The corrosion of the coating is due to electrolysis slowly removing zinc ions from the surface of the coating. The detection of the depleted zinc and its possible effect on the levels of zinc found in the water column at the cage sites, at any given time, is difficult to assess and quantify.

Using a similar method of analyses to that used for sediments in Chapter 3, a comparison was made of the levels of zinc found in oysters grown at control sites with those grown at (and attached to) the wire sea cage sites. Mussels and oysters are filter feeders and are known to accumulate metals, including zinc and have been used as biological indicators in other studies assessing the accumulation of metals in marine waters. For example, the Sydney rock oyster *Saccostrea glomerata* is endemic to this area and has previously been used successfully as a biological indicator (see Scanes 1992, 1996) in the Sydney region. This species was used in this to assess if bio-accumulation of zinc was greater in oysters grown near the MarineMesh™ cages compared to control sites.

Food surveillance authorities require strict quality control and assurance to help protect consumers from unsafe foods. The seafood industry is often difficult to regulate in terms of quality, as most of its product is harvested from the wild or wild fish stocks.

The aquaculture industry has the ability to supply a product that has been cultured in known conditions including location, housing, water and feed quality. The product itself can be routinely monitored to ensure that it is within the Australian & New Zealand Food Authority (ANZFA) guidelines for safe consumption. For fish, the guideline for the acceptable concentration of zinc in muscle tissue is a value less than 150 mg/kg (ANZFA Food Standards Code A12 - Metals and Contaminants in Food).

The objectives of this section were to: (1) to document the concentration of zinc in the muscle and liver of fish cultured in zinc-coated wire cages (including reference to associated research on zinc in water); and (2) to determine if these were within the regulations and regarded safe for consumption. The study also provided information about the amount of zinc in the pelleted fish diets. This was done to determine whether the pelleted diets were a possible source of zinc to the tissues of the fish.

4.2 Materials and Methods

4.2.1 Zinc in the water

The Sydney Rock Oyster *Saccostrea glomerata*, was used as a biological indicator for increases of zinc in water. At the sea cage sites (see Figure 3.2) the sampling units were attached directly to the sea cage wire in a central position on the sidewalls of each sea cage. The control sites were located around Botany Bay (see Figure 3.2). The oyster baskets were attached to stationary objects under the water, such as a permanent navigational mark or jetties, using SCUBA.

Oysters were purchased from a commercial oyster farmer in Quibray Bay, on the southern side of Botany Bay, NSW. The farmer provided history of the oysters, which showed that they were wild caught spat that had been initially grown in the Hawkesbury River, NSW. They were then transferred to Quibray Bay for final grow out to market size. The oysters were removed from Quibray Bay and purged for 24 hours in re-circulating filtration tanks, a standard harvesting procedure.

Twenty-five oysters were placed in each of nine cylindrical plastic oyster baskets with an approximate volume of 5000 cm^3 and mesh size of 20 x 20cm, that were clip-locked to ensure that oysters were not lost. Divers attached baskets to physical structures (e.g. pilings) at each control location and at the two zinc-coated sea-cages using plastic cable ties. Baskets were placed at a depth of 3.5 m below the mean tide level. The oysters were deployed at all the control and sea cage locations on 15th May 1999. Ten oysters were retrieved from each basket on each of two occasions separated by about six weeks (namely the 30th June 1999 and 15th August 1999). On each occasion, all oyster baskets were retrieved and 10 randomly chosen oysters removed from each basket. Each basket was then returned to its original location.

The oysters that were removed from each basket were placed into appropriately labelled plastic bags. Oysters from each location were then placed in separate 120-litre aquaria and purged in water filtered to approximately 100 microns for 24 hours. This process was established to remove excess sediments from within the oyster shells and improve the consequent analyses for zinc.

Oysters were shucked, rinsed and weighed (wet weight). Each oyster was then freeze dried for 48 hours. The freeze-dried oysters were then digested with acid, following the methods of Baldwin et al. (1994) and Deaker et al. (1995). A Perkin-Elmer Elan 6000 Inductively Coupled Plasma Mass Spectrometer (ICP-MS) was used to determine the concentration of zinc in the oyster tissue digests. Standard Reference Material (SRM) was also routinely digested throughout the analytical phase to document the recovery rates and ensure quality control.

Statistical Analyses

The resulting data were analysed using two 1-factor analyses of variance to examine the spatial variation in zinc concentrations after the 6 week and 12 week deployments. Homogeneity of variance was examined prior to analysis of variance and where variances were heterogeneous the data were transformed following the procedure of Underwood (1981). When the Sites term was significant in the analysis of variance, differences among means were identified using Dunnett's multiple comparison test, with a Type I error-rate of 0.05.

4.2.2 Zinc in fish from wire sea cages

Fish (snapper) were grown on the farm site from an approximate mean weight of 5 g to a mean weight of 305.4 g (SD = 38.3) in soft, nylon-mesh sea cages and then transferred into the wire sea cages. Prior to their transfer, 10 fish were selected in March 1999 and analysed for zinc in their muscle and liver tissues. A further 10 fish were supplied every 4 weeks over the period April – June, 1999. A final sample of 10 fish was collected in February 2001, at which time the fish had been successfully grown in the cages.

The fish were weighed and samples of muscle and liver removed. The tissue samples were then freeze dried and acid digested following the methods of Baldwin et al. (1994) and Deaker et al. (1995). A Perkin-Elmer Elan 6000 Inductively Coupled Plasma Mass Spectrometer (ICP-MS) was then used to determine the concentration of zinc in the fish muscle and liver samples. Standard Reference Material (SRM) was also routinely digested throughout the analytical phase to document the recovery rates and ensure quality control. In addition, a known sample was randomly included in batches for analyses to determine repeatability.

Two commercially available fish pellet diets were used to feed the fish prior to and during this study. Randomly chosen samples of both feeds (hereafter referred to as Feed 1 and Feed 2) were collected by the operator and stored in labelled plastic bags prior analyses.

Commercial Feeds 1 and 2 (fish pellets used to feed the fish) were processed in a similar manner to the fish tissues and then analysed for the concentration of zinc in an ICP-MS

Statistical Analyses

The data were analysed using single factor analyses of variance. Prior to analysis of variance the data for the concentrations of zinc in the liver and muscle tissues were tested for homogeneity of variance using Cochran's test (Underwood 1981). The data for the concentrations of zinc in liver were heteroscedastic, but homogeneity of variances was achieved after log (x+1) transformation of the raw data. The data for the concentrations of zinc in muscle tissue also yielded heterogeneous variances which, unfortunately, could not be stabilised using a variety of transformations. However, as analysis of variance is robust to heterogeneous variances when there are large numbers of replicates (Underwood 1981, 1997), analyses were done using the untransformed data.

4.3 Results

4.3.1 Zinc in the water

Spooner (1999) has reported on the results of an investigation of zinc in oysters placed in the zinc-coated cages (as part of a related study). The levels of zinc within oyster tissues did not differ among sites (refer to Spooner 1999, Tables 4.6 and 4.7, $P > 0.05$). However, the level of zinc within tissues of oysters tended to decrease over time at some locations (e.g. refer to Spooner 1999, Fig 3.1), whereas other sites remained virtually unchanged. There was no evidence of an increase in the concentration of zinc in the oysters at the wire sea cage site compared to the oysters at the control sites after 6 or 12 weeks of deployment. It is important to note that the results of the analyses of the 10 randomly chosen oysters sampled prior to the initial deployment in Botany Bay revealed high levels of zinc and other metals such as arsenic.

4.3.2 Zinc in fish farmed in wire sea cages

Zinc in Fish Tissues

The concentrations of zinc in muscle and liver tissues of the fish differed significantly through time (Tables 4.1, analyses of variance, $P < 0.05$). The concentration of zinc in muscle tissue initially decreased from the period before to after placement in the wire sea cage, but then increased and fluctuated through time (Table 4.1, SNK, $P < 0.05$). In contrast, the mean concentration of zinc in the liver of the fish was significantly greater from before to after their placement in the wire sea cage (Table 4.2, SNK test $P < 0.05$). Following placement in the marine mesh sea cage, the mean concentrations of zinc in liver tissue fluctuated significantly through time (Table 4.2, SNK, $P < 0.05$).

Analysis of Variance (muscle tissue)

Table 4.1 Analysis of variance of the concentrations of zinc in the muscle tissue of snapper grown in zinc coated sea cage.

Note: data transformed to log (x+1).

Source of variation	df	MS	F	P
Times	4	1.220067	7.320644	2.13E-04
Residual	45	0.166611		
Total	49			

Results of SNK test (muscle tissue)

March 99 (prior to stocking in Zn cage) > April 99 < May 99 > June 99 > Feb 01

Analysis of Variance (liver tissue)

Table 4.2 Analysis of the concentrations of zinc in the liver tissue of snapper grown in zinc coated wire sea cage.

Note: untransformed data used in analysis.

Source of variation	df	MS	F	P
Times	4	1.112818	7.322272	2.13E-04
Residual	45	0.151977		
Total	49			

Results of SNK test (liver tissue)

March99 (prior to stocking in Zn cage) > Feb01 > June99 > April99 > May99

Table 4.3 Mean (± SD) concentrations of zinc in the muscle and liver tissues of snapper grown in zinc-oated wire sea cage (n = 10).

Note: the ANZFA food standards code A12 states that for safe human consumption of fish the concentration of zinc in muscle tissue should not exceed 150 mg/kg. The initial sample, March 1999 was taken prior to the placement of the fish in the wire sea cage, the following samples were all taken after deployment.

DATE	MUSCLE mg/kg		LIVER mg/kg	
	Mean	SD	Mean	SD
MARCH/99	0.0321	0.0134	0.1619	0.1178
APRIL/99	0.0163	0.0036	0.0743	0.0059
MAY/99	0.0240	0.0152	0.0691	0.0334
JUNE/99	0.0456	0.0391	0.0903	0.0251
FEBRUARY/01	0.0380	0.0090	0.1261	0.0161

The mean (± SD) concentrations of zinc in fish muscle and liver (Table 4.3) did not exceed 0.0456 (± 0.0391) mg/kg and 0.1619 (± 0.1178) mg/kg, respectively. More importantly, the concentrations of zinc in the fish muscle tissue were 3 orders of magnitude below the levels of zinc recommended for safe consumption by the Australian and New Zealand Food Authority.

4.3.3 Zinc in Pelleted Feed

The mean (± SD) concentrations of zinc in the commercial fish feeds 1 and 2 had 181.5 (± 18.3) mg/kg and 54.5 (± 1.32) mg/kg of zinc, respectively.

4.4 Discussion

4.4.1 Impact of zinc on water surrounding zinc coated wire sea cages

The Sydney rock oyster has been successfully used as a biological indicator in relatively shallow open waters to observe accumulation and rate of effect over similar time period to those used in this study (see Scanes 1992, 1996). It was anticipated that the oysters purchased and selected for sampling would initially have had relatively low levels of zinc. However, the oysters were found to have quite high initial levels of zinc. As the analysis of all samples was done after the sampling process was complete, this problem could not be rectified. Therefore, results obtained from this component of the study may not be conclusive. The reduction in tissue metal levels indicates that the oysters were previously grown in a location exposed to greater metal ion concentrations compared to the locations chosen for this study, within Botany Bay. However, the levels of zinc found in oysters attached to the wire sea cages did not significantly increase through time in comparison with those of the oysters at the seven control locations. This suggests that the loss of the zinc from the cages occurred at a rate that did not cause increases in the concentration of zinc above those that were already present in the oyster tissues.

It is important to note that no detectable changes occurred in the levels of zinc in the superficial sediments analysed (see Chapter 3). Moreover, the depletion of the zinc coating of the wire sea cages resulted in a considerable mass of zinc being lost into the surrounding environment (see Chapter 3). It would be logical to conclude that the depleted zinc particles or ions may have been dispersed in the water column by the current flow which ranges from 8 – 12 cm/sec (Quartararo 1996) occurring at the sea-cage site. Visual observations made when collecting sediment samples indicated that obvious and sometimes dramatic movements of sediments had occurred, particularly after extreme weather conditions had been experienced resulting in heavy wave action within Botany Bay. It was estimated that the depth of sediments at a particular location could vary or alter up to 300 mm in depth between quarterly sampling periods. These movements were obviously occurring from strong currents throughout the water column, supporting the theory that it is most likely the zinc was dispersed in the water column by current flow.

4.4.2 Impact of zinc on fish grown within wire sea cages

The significant decrease in the mean concentration of zinc in the liver tissue of the snapper is contrary to what would be predicted if the fish were accumulating zinc from the coated wire sea cage or from the pelleted food. While the mean concentrations of zinc in the muscle tissue of snapper significantly differed through time, the mean concentration never exceeded the ANZFA guideline. These results provide and demonstrate compelling evidence that the coated wire sea cage had no detectable effect on the concentration of zinc in the muscle and liver tissues of the snapper after almost 2 years of growth in the coated wire sea cage.

The analyses of fish feeds 1 and 2 (See 4.3.3) indicated that both feeds contained reasonably high levels of zinc. As the zinc concentrations in fish tissues showed no significant increase, it is likely that any additional (unused) zinc in the pelleted feeds was passed through the gut of the fish without being accumulated.

It is also possible that the feeding behaviour of the snapper may have contributed to the very low levels of zinc found in the animal's tissues. Moreover, as zinc is mainly accumulated in body tissues by ingestion (Rainbow 1992) and snapper are opportunistic feeders (Henry 1988) which often graze on the invertebrates living on sea cage netting, it is conceivable that the fish may have directly ingested some zinc from the corroding zinc-coated wire whilst feeding on these invertebrates. However few, if any, invertebrates were present on the wire sea cages (see Chapter 5) and thus it is likely that little foraging occurred on the wire. Consequently, it is highly unlikely that the snapper would have ingested zinc directly from the wire over the duration of the study.

The other main objective of this part of the study was to examine whether fish grown in coated wire sea cages were safe for human consumption. The levels of zinc found in the liver and muscle tissues were substantially less than that specified by the ANZFA standards and the snapper grown in the coated wire sea cages were clearly safe for human consumption with respect to concentrations of zinc.

CHAPTER 5. BIOFOULING IMPACTS OF SEA CAGE NETTINGS

5.1 Introduction

Biofouling organisms are classified as either micro-fouling or macro-fouling organisms (Lewis, 1994). Micro-fouling organisms include bacteria, diatoms and protozoa, which form a thin layer or slime over submerged substrata. Macro-fouling organisms are further sub-divided into soft-fouling and hard-fouling organisms. Hard-fouling organisms are those that secrete calcium carbonate tubes, shells or skeletons. These include animals such as bivalves, barnacles, tubeworms, bryozoans and corals (Lewis, 1994). Soft-fouling organisms are those that lack such hard structural components. These organisms include most of the algae, and animals such as hydroids, sponges and ascidians (Lewis, 1994).

NSW DPI (Fisheries) used traditional, soft-mesh netting to hold finfish, to establish the commercial fish farm at the entrance to Botany Bay. The present proprietor of the farm uses the same nets and maintains them under the same maintenance regime as originally established. These soft-mesh nets are reported to require changing and cleaning every 10-28 days depending on the season and mesh-size (Quartararo, 1996). As described in the previous chapter, the coated wire cages, located close to these original cages used on the fish farm are exposed to similar environmental conditions and may offer economic benefits.

The objective of this part of the study was to quantify any reduction in the mesh-size (i.e. the size of the mesh holes) of the wire zinc coated mesh resulting from biofouling. The mesh sizes of sea cage nets are gradually reduced as biofouling organisms such as algae, soft and hard invertebrates attach themselves and grow on the nets. The greater the amount of biofouling, the greater the reduction in mesh size and consequently a reduction in water flow through the mesh. The reduction of water flow through the mesh can affect the available Dissolved Oxygen (DO) of the water within the net. As it is desirable for farmers to densely stock sea cage nets, the availability of DO to the fish being farmed is an important factor.

5.2 Materials and Methods

Two zinc coated wire sea cages were placed in operation on the fish farm site and were immersed in salt water for approximately 16 months (30/10/98 to 25/2/00), respectively. The cage that had been stocked with fish for the longest period of time was chosen to carry out the first experiment. It had been subjected to nutrients from fish feed and faeces, which may have enhance algae growth and contribute to biofouling and corrosion of the mesh, as often occurs with sea cages utilising soft netting (Lewis 1994, Quartararo 1996). Subsequently, this sea cage was sampled to assess the reduction in mesh size due to biofouling. This was done in two ways. Firstly, casual observations of the appearance of the sea cage were made intermittently throughout its deployment whilst carrying out the other sampling detailed elsewhere in this thesis. Secondly, the sea cage was sampled quantitatively after 16 months of deployment. To this end, photographic images (slides) were taken of 0.1 m^2 quadrats of the biofouled mesh and clean mesh quadrats adjacent to one another (Figure 5.2). The clean mesh quadrat was made by placing a piece of new mesh on a quadrat with black shade cloth backing to assist with photographic resolution (Figure. 5.1). Each side of the cage, (i.e., N, S, E, and W) were stratified into top and bottom regions. Three replicate 0.1 m^2 quadrats of biofouled mesh were chosen at random in each of the top and bottom regions of each side and labelled for subsequent identification. These were then photographed together with a 0.1 m^2 quadrat of clean (unfouled) mesh.

In the laboratory, the developed images were illuminated through a standard slide projector ensuring that the projector was level and perpendicular to the screen to reduce any optical distortions and/or errors. On examination of the images, 42 holes or "squares of mesh" could be observed in each 0.1 m^2 quadrat. Ten mesh squares were then selected at random from each of the control (ie. unfouled) and treatment (i.e., biofouled) meshes in each slide and the mesh-size of each "square of mesh" were measured. The actual measurement of each individual square mesh hole was made across the diamond, as the horizontal width, to the nearest mm.

The data obtained was analysed using two factor, nested analyses of variance, with factor 1 (net type, fixed) and factor 2 (frames, nested in net type). Prior to analysis of variance the data for each mesh type were tested for homogeneity of variance using Cochran's test (Underwood 1981). The data yielded heterogeneous variances which,

unfortunately, could not be stabilised using a variety of transformations. However, as analysis of variance is robust to heterogeneous variances when using a large number of replicates (Underwood 1981, 1997), analyses was carried out on un-transformed data.

Figure 5.1 Profile of cage wall with "fouled mesh" and "clean mesh" quadrats.

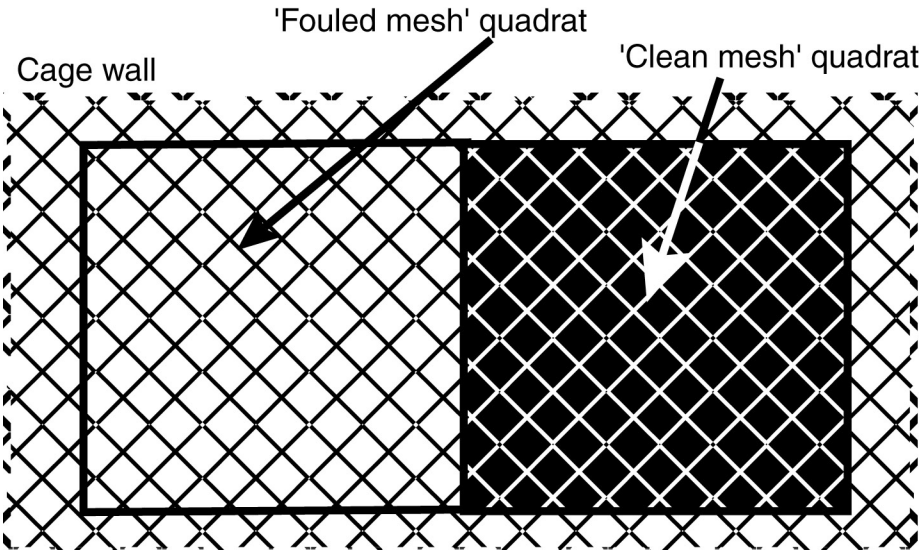
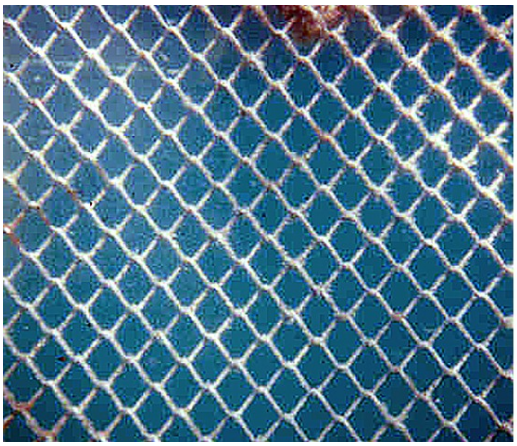
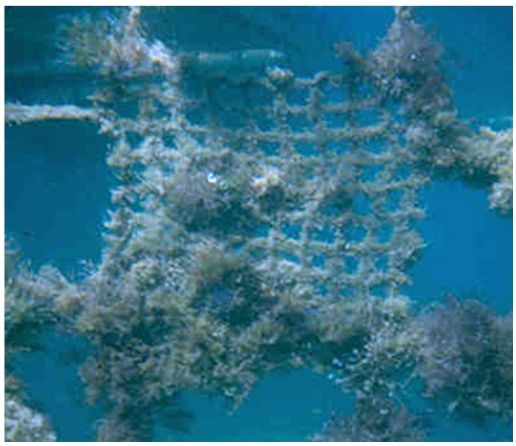


Figure 5.2 Image showing accumulation of biofouling on wire coated mesh (left) and nylon mesh panels (right) after twelve weeks of submersion.





5.3 Results

The photographic images of quadrats of unfouled and biofouled mesh proved to be a robust and reliable means of documenting changes in mesh-size (e.g., see Figure 5.2). The unfouled wire mesh had a mean mesh-size of 36.5 mm (SD = 1.08 mm) and exhibited relatively little variation among replicate areas. After the 16 month deployment, the mesh-size of the biofouled mesh varied among the replicate quadrats and this resulted in heterogeneous variances (Cochran's test, $P < 0.05$). Various transformations of the data were examined, but were unable to stabilise the variances. Given that there were numerous replicates and that analysis of variance is robust to heteroscedasticity and departures from normality (Eisenhart, 1947; Underwood, 1997), the analysis of variance was done using the untransformed data.

Following the 16 month deployment, the minimum reduction in zinc cage mesh-size was 4.6 mm (i.e., from 36.5 to 31.9 mm, Table 5.1) representing a maximum reduction of 12.6 % of the clean mesh size. This reduction in mesh-size was evident in the top region of the eastern wall of the sea cage. The maximum reduction in mesh size resulting from biofouling was 9 mm (i.e., from 36.5 to 27.5 mm, SD = 1.93 mm, Table 5.1). This result represented a reduction of 24.7 % of the original mesh-size and was evident in the top region of the northern wall of the sea cage. In spite of the varied reduction in mesh-size, the analysis showed that there were no significant differences in the reduction of mesh-size among sides of the cage or between the top and bottom regions (Table 5.2, $P > 0.05$).

These observations of clean wire coated mesh and the potentially fouled wire coated sea cage wall have provided evidence regarding the accumulation of biota on the coated wire mesh over the 16 month period, quantifying the effect of biofouling by observing the reduction in the hole size of mesh after fouling.

Table 5.1 Mean (+ SD) mesh-sizes (mm) of clean and biofouled coated wire mesh at different positions on the sea cage.

Note: clean (unfouled) mesh had a mean (+ SD) mesh-size of 36.5 (1.08) mm.

POSITION	CAGE WALL			
	North	East	South	West
Top	27.50 (1.93)	31.90 (1.99)	27.80 (1.77)	28.30 (1.84)
Bottom	29.70 (2.24)	30.10 (2.05)	28.80 (2.06)	29.90 (2.51)

Table 5.2 Analysis of variance comparing the mesh-sizes of the coated wire mesh at various positions on the sea cage after 16 months of deployment.

Source of variation	df	MS	F	*P*
Mesh Type = M	1	5.78E-03	3.20E-02	0.86
Sides = S	3	1.61E-03	8.92E-03	1.00
Position = P	1	0.554528	3.066306	0.09
M x S	3	2.30E-02	0.12735	0.94
M x P	1	4.33E-03	2.39E-02	0.88
S x P	3	1.07E-02	5.90E-02	0.98
M x S x P	3	4.09E-02	0.877639	8.78
Quadrats(M x S x P)	32	0.180846	0	0.00
Residual	432	1.83E-03		
Total	479			

Table 5.3 Analysis of variance, comparing the mesh-sizes of the coated wire mesh and soft nylon netting after submersion for a 12 week a period.

Source of variation	SS	df	MS	F	*p*	F vs
Mesh Type	10597.45	1	10597.45	110.1527	<0.001	Frames(mesh)
Frames (M)	384.8273	4	96.20683	4.599962	<0.001	RES
Residual	1129.394	54	20.9147			
Total	12111.67	59				

Results of SNK test (frames of mesh)

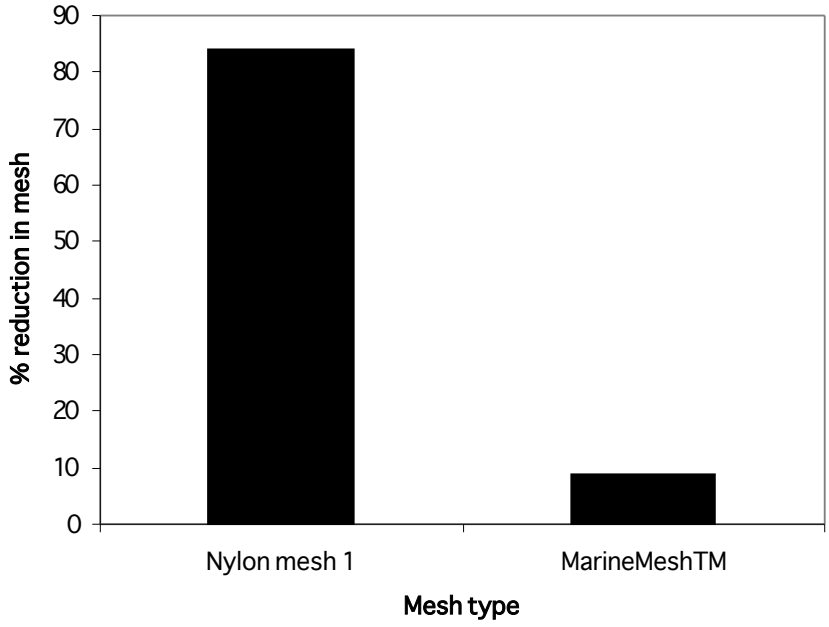
coated wire mesh > frame 1 = frame 2 = frame 3

nylon mesh > frame 1 < frame 2 = frame 3

The heterogenous variances that occurred are most likely a result of the spatial variation of fouling organisms growing on the nylon net frames. The biofouling on the nylon mesh was comprised of large species of algae with leaves that completely blocked some mesh holes. In comparison the wire coated mesh had a small amount of biofouling which more evenly covered the wire in a film of biota. Data from each frame of mesh was tested (see SNK test Table10) to establish if any differences could be found among replicate frames of each mesh type. Results indicate significant differences among nylon mesh frames and no significant differences amongst wire coated mesh frames, suggesting there is far more variability amongst the fouling of nylon mesh. Analysis of variance results indicated a significant difference between nylon and coated wire mesh with respect to the reduction in mesh hole size resulting from biofouling.

Following the 12 week submersion period of the mesh frames, the mean reduction in mesh-size of nylon mesh was from 36.5 to 5.9 mm, representing a reduction of 81% of the clean mesh size. Coated wire mesh had a reduction in mesh size from 36.5 to 33.1mm, representing a reduction of 9% of the clean mesh size. These results are presented as a % reduction in mesh size for each mesh type (see Fig 5.3).

Figure 5.3 Reduction (%) in mesh hole size of coated wire mesh and nylon netting.



5.4 Discussion

This study used a visual, photographic method combined with statistical analyses to assess the reduction of biofouling occurring on the of the coated wire mesh. These observations provided information for each side wall of the coated wire mesh sea cage. Each side of the cage, (i.e. N, S, E, and W) was stratified into top and bottom regions to possibly detect any difference in the degree of biofouling occurring, possibly affected by factors such as depth or exposure to sunlight. Results clearly showed that statistically there was no significant difference between fouled and clean mesh after the 16 months of deployment. Nor were there any detectable differences between sidewalls at any depth. This indicates that there was no detectable difference in the degree of biofouling over the study period, using this method of assessment.

Results from a direct comparison of nylon and wire coated netting made over a twelve week period, clearly show the wire coated netting was far less affected by biofouling organisms than traditional soft nylon netting. The reduction in mesh hole size of nylon netting was significantly greater than that of the coated wire netting. The author also made visual observations on S.C.U.B.A and from the surface. These casual observations made intermittently throughout the 16-month deployment of the sea cage showed that there was little, if any, biofouling of the wire mesh during the first 9 months as evidenced by the still metallic shine of the wire. During the following 7 months, a very thin, algal film became evident in patches on all sides of the cage. While not quantified, the film appeared to reach maximal coverage during late spring, 1999. Soon after the algae appeared to simply fall off with current and wave action.

The fact that the coated wire sea cage was still in operation after 16 months use is significant in determining the functional qualities of the cage. It was necessary for the operator of the fish farm to change and clean the traditional soft mesh net cages used on the farm every 10-28 days, as also experienced by NSW Fisheries (Quartararo, 1996). In terms of biofouling reduction the coated wire sea cages required no maintenance or cleaning over their period of use and were in good condition at the end of this study.

The degree of biofouling that grows on a structure is relative to a number of factors, including its exposure to settling biofouling organisms (Lewis 1994). Future observations or studies of wire coated netting may determine the biofouling reduction

properties to be very different to what was found in this study. However the location of this study site was at the entrance to Botany Bay (see Figures 3.1 & 3.2) which is a renowned catchment area for Sydney rock oyster and mussel spat (Nell 2002) and other molluscs. In addition, the sea cage flotation structures used on this farm experience a tremendous amount of biofouling comprising a diversity of organisms.

Considering these factors the chosen study site and period of observation this study should provide a sound observation of the typical biofouling that could be expected to occur using zinc coated wire sea cages in a similar temperate marine environment. A major constraint to the introduction of zinc coated cages may be the greater intial outlay in capital costs.

CHAPTER 6. GENERAL DISCUSSION AND CONCLUSIONS

This thesis examined several aspects, including sediments, water and biofouling, which may be indicators of possible environmental effects that may be caused by the operations of a commercial fish farm operating in Botany Bay, NSW Australia.

6.1 Sedimentary studies

6.1.1 Assessment of nutrients in sediments

The levels of sedimentary Total Organic Carbon (TOC), Total Phosphates (TP) and zinc beneath the sea cages and amongst seven control sites through time, located in Botany Bay, were examined.

The monitoring of sediments for TOC and TP indicated that these parameters did not significantly alter through time beneath the sea cages in comparison to levels found throughout Botany Bay at the seven control sites. However, as the sea cages were not stocked to their maximum potential there is a possibility that detectable impacts may occur under higher stocking densities. The maximum stocking density of any individual cage during the study period was 20 kg/m^3. This low stocking density and therefore low overall feed levels compared to other commercial farms (e.g., see Miyako, pers. comm., 2003), also makes it difficult to compare this data to other publications which investigate problems that have already occurred on large scale production sites with much greater stocking densities, such as those described by Seymour (1991), Tsutsumi (1991) and Pilay (1992).

As no significant differences were found between TOC and TP data collected from control and "impact" sites, the data collected from each site was combined and this provided a much larger and more powerful data set, representing typical sediment TOC and TP for the farm site as a whole. This also indicated there were no significant differences between the control and impact (cage) sites.

Macroscopic observations of the surface sediments also indicated no apparent changes to the sea floor.

6.1.2 Assessment of zinc in sediments

The next objective of the sediment studies was to assess possible increases in zinc in sediments, resulting from the use of coated wire sea cages

The detection of metals in marine waters and habitats has been investigated in other studies in NSW. Methods currently available to identify environmental indicators were examined to decide the most cost effective and appropriate method to assess possible impacts resulting from the use of zinc in the coated wire mesh sea cages. The depletion of zinc is an inevitable process. The rate of depletion of zinc from the wire cages was unknown, but it is likely that the depleted zinc would be dispersed into the water column and/or the sediments beneath the sea cages. It is also possible that the fish being grown within the cages may absorb some of the zinc by means of ingestion, whilst grazing on biota growing or residing on the sea cage.

Previous collections of sediments within the farm lease area and at other sites around Botany Bay had taken place as part of the farm's required environmental assessment program. This provided data for twelve months prior to the introduction of the zinc coated sea cages. A sampling strategy was devised to collect sediments beneath the coated wire cages prior to their placement on the farm and from other control locations within Botany Bay. Sediments from all locations were analysed and statistical analyses of results revealed no detectable increases in the level of sedimentary zinc beneath the sea cages through time.

Zinc in the water

Sydney Rock Oysters *Saccostrea glomerata* were chosen as biological indicators to detect and possibly assess the impact of zinc being dispersed into the water column. The possible detection of an increase in zinc in oyster tissues from samples taken beneath the wire cages could indicate two things: (1) that zinc levels within the water column were higher than those in other areas within Botany Bay, suggesting that the wire cages or zinc contained within the fish feed had contributed to this increase; and (2) that biota, such as molluscs, residing within close proximity to the wire cages, are affected by this increase of zinc in the water column.

This method has been used successfully to determine the levels of metals found in waters surrounding deep-water ocean sewage outlets of Sydney, NSW (Scanes 1996).

An unexpected situation occurred with this component of this study, in that the oysters obtained for the study were already laden with metals including zinc (which is not unusual for bivalves – see Rainbow (1992) for a review). This made the interpretation of results more difficult. However, statistical analysis of results indicated that there was no significant increase in zinc within oyster tissues at the sea cage sites compared to the control sites. These results suggest that the amount of zinc depleted from the coated wire sea-cages was outside the detection limits of this study, and unlikely to cause any environmental concern, as they were below standards.

6.2 Biofouling studies

6.2.1 Biofouling reduction of wire coated sea cages

This study provides the first quantitative analysis of the reduction in mesh size of "zinc coated", wire mesh sea cages, due to biofouling. From the results obtained, it is clear that statistically there was no significant difference between fouled and clean mesh, however, it was difficult to determine whether the reduction in mesh size was significant in terms of affecting the function of the net cage. The fact that the "zinc coated" wire cages were still in operation after 16 months use is significant in determining the functional qualities of the cage. By comparison, it is necessary for the operator of the fish farm to change and clean the traditional soft mesh net cages every 10-28 days (Quartararo, 1966). The coated wire cages have undergone no maintenance over their period of use and are still in good working order with no apparent corrosion.

6.2.2 Comparison of biofouling between coated wire netting and nylon sea cage netting

In this study it was found that other parameters may be examined to help determine the overall performance of the wire cages. As the mesh size is reduced, measurement of dissolved oxygen levels could display some effect due to the current level of biofouling. However, as the cages are presently only stocked at low densities it is unlikely that any significant differences would be detected between dissolved oxygen levels inside and outside the cage.

6.3 Limitations and Recommendations for future study

There were no fundamental limitations. However, there were several factors identified that may have affected the outcomes of this study, these included:

- Low stocking of fish farm, particularly during the period that sediments were sampled.
- The monitoring period for sediments was also limited due this time being the first two years of the farms commercial operation.
- Further analyses of benthic fauna within sediments collected may have been beneficial to detecting changes occurring in sediments. Due to a lack of resources these sediment samples were not analysed, but archived for future analyses if required.

There are a number of recommendations that could be observed in any future work or studies investigating the environmental aspects of the fish farm site. These include include:

- Further sediment sampling should be done when the farm is stocked to a greater level of production.
- Monitoring of sediments should also be done over a longer period of time and continually throughout the life of the farm.
- Increased replication of sediment samples may provide a more powerful analysis.

6.4 Final Conclusion

Overall, the study found no detectable impact on sediments due to the introduction of the fish farm. There was also no detectable impact from the placement of the wire sea cages on the farm site on sediments, fish grown within the wire cages, or the surrounding waters.

The zinc coating on the wire sea cages showed a significant difference in the reduction of biofouling growth, compared to traditional nylon netting, which may reduce the possible environmental impact caused from biofouling waste material.

In conclusion, each of the objectives of this study has been achieved, and the results indicate that the current operation of the sea cage farm culture of fish in Botany Bay meets the national and state guidelines for food safety and environmental quality.

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