Environmental impacts of cage aquaculture in the southeast arm of Lake Malawi: water and sediment quality and food web changes

by

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Lake Malawi is a great lake not only because of its size (30,800 km^2) but also because of its unique fish diversity. The lake contains the highest number of freshwater fish species in the world. The fish species are hypothesized to have radiated within the lake, which is 1-2 million years old. The collapse of the capture fishery in Lake Malawi between the 1970s and 1990s led to the launch of cage culture of indigenous fish species in 2004 in the south east arm of the lake. While cage culture has been practiced for many years in temperate lakes and seas, the fish farm in Lake Malawi is the first in the African Great Lakes and, therefore, not much information currently exists that is relevant to the impact of cage culture on such a large, species-rich tropical lake. Consequently, a study was done between January and December, 2007, at the fish farm in Lake Malawi to determine potential impacts of cage wastes on the environment. The study found that, just like in temperate systems where 70-87% of C, N and P added through feed get dispersed into the environment, discharges from fish cages in Lake Malawi were between 71-88% of the nutrients added through feed. The discharges were proportional to the amount of feed added so that as production and feed supply increase over time, more cage wastes would be generated and released into the environment. The discharges were exacerbated by poor stocking and feeding regimes. Production periods were longer (mean of 376±42 days) than if recommended stocking and feeding rates were followed. Feed quality may also have affected production performance and waste generation in the cages, but was not studied. The cage wastes were incorporated into the food web and support the wild fishes in the vicinity of the fish farm. Impacts of the cage wastes on the water column and sediments in the vicinity of the cages were

minimal during the study period, probably because of rapid and efficient dispersion of the wastes by strong water currents, that averaged 9.3 cm s^{-1}, through the cages and high consumption of the cage wastes by large numbers of wild fishes which aggregated around the cages. The wild fishes also helped to disperse the cage wastes over a larger area through consumption, translocation and defecation. However, as production increases, the amount of cage wastes generated may overwhelm mitigation by dispersion by water currents and consumption by wild fishes, particularly if many cages are deployed close together and interfere with current flows. Based on my observations, a fish farm that produces 15,000 tonnes fish/yr in Lake Malawi would generate 1249, 113 and 21 megamoles/yr of C, N and P, respectively, that are comparable or higher than DOC, TDN and TDP loadings observed in the most disturbed large river systems draining into Lake Malawi. The impacts of these river systems in Lake Malawi have been well documented, particularly around river mouths and in the more densely populated and shallower southern portion of the lake, where algal communities and their sedimentation rates have begun to change. Cage culture discharges may accelerate these changes.

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Dedication

To my wife Monica, my daughter Anita Temwani, my son Augustine Leo, and to my sisters and brothers. Your encouragements and prayers carried me through this program.

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Chapter 1
Cage culture in Lake Malawi

1.1 Introduction

Malnutrition remains an important public health problem in many developing African countries such as Malawi affecting many lives, particularly women and children under-five years (Lauderdale, 2000). Malnutrition has many causes including micronutrient (e.g. iron, vitamin A, zinc and iodine), energy and protein deficiencies (Lauderdale, 2000). While some nutritionists question the existence of protein deficiency, others strongly believe that it remains an important dietary problem especially in countries such as Malawi where diets are predominantly plant based (Lauderdale, 2000). The capture fishery has been the primary source of food fish that provides the cheapest source of dietary animal protein in most African countries, including Malawi. In the 1970s, fish contributed more than 70% of the total dietary protein consumed in Malawi, but rapid human population growth, over-fishing, use of illegal fishing gear and environmental degradation over the years have reduced capture fish stocks in Malawi's natural water bodies, including Lake Malawi, Lake Malombe and the upper Shire River (Fig. 1.1). As a result, the contribution of the fish to human dietary protein intake in Malawi has declined by more than 50% from a peak contribution of 70% in the 1970s (Banda et al., 2005). Recent reports (e.g., Banda et al., 2005) show that the annual per capita fish consumption in the country has dropped from 14 kg of fish in the mid 1970s to below 3 kg in 2003 so that diets have become predominately plant-based. This has resulted in widespread malnutrition, particularly in rural communities across the country (Mumba & Jose, 2005). The collapse of the capture

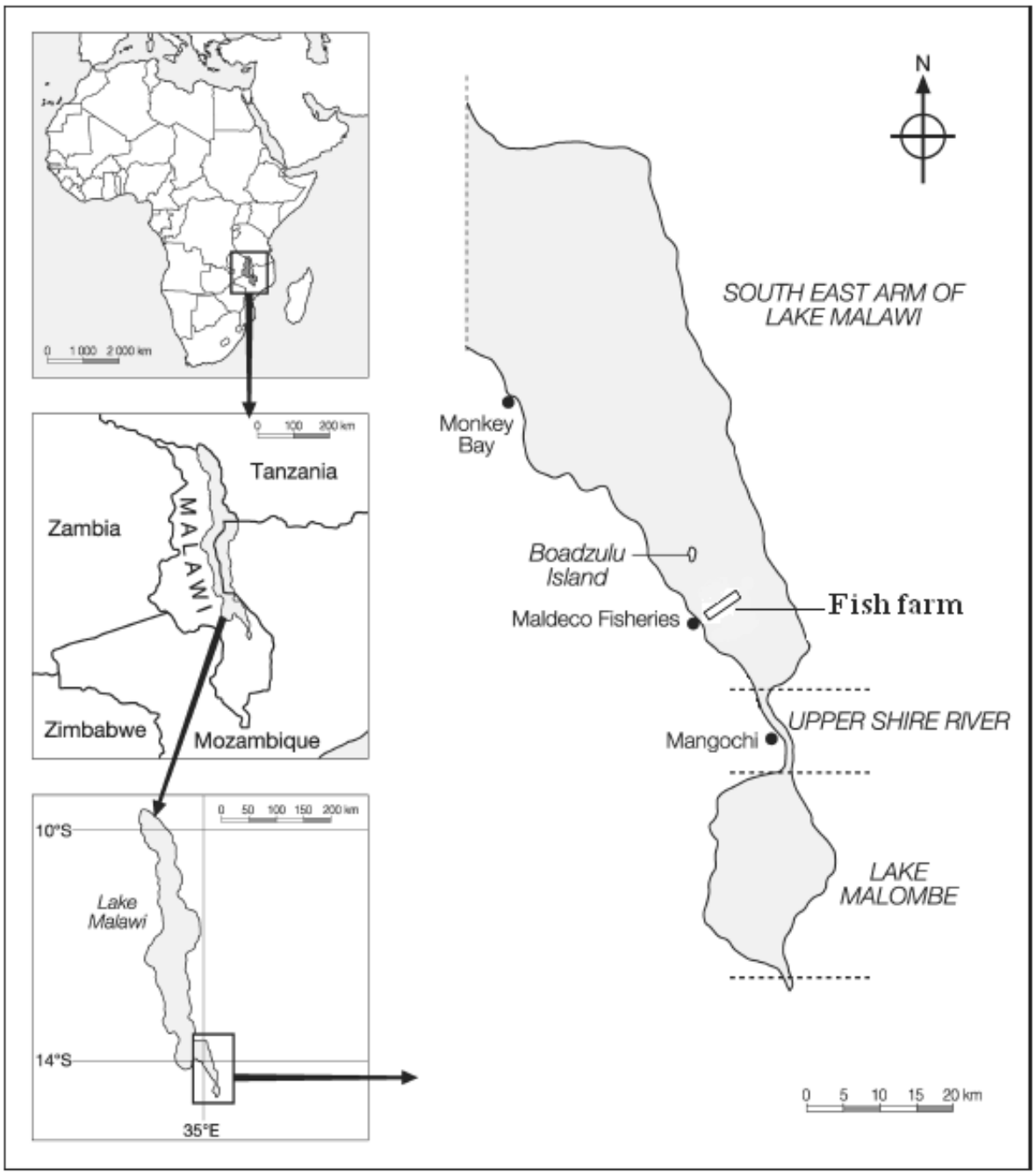


Fig. 1.1. Lake Malawi, the southeast arm showing the cage fish farm at Maldeco, the Upper Shire River and Lake Malombe. Fish farm not drawn to scale. Modified from Hara (2006).

fisheries also affected incomes of about 1.6 million people involved in fisheries-related activities such as fishing, fish processing, marketing and trading, boat and gear making and allied industries (Banda et al., 2005).

The decline of capture fishery production in Malawi's water bodies has significantly contributed to the recent (since 1980s) rapid growth of smallholder land-based aquaculture production through a government initiative to reduce malnutrition problems in the country. However, despite the significant rise in the number of earthen fish ponds and farmers in the country, fish production per hectare of pond still remains low so that the intended objective of improving protein consumption at the household level has not been achieved yet. The current annual fish production of 800 tonnes produced in 7000 earthen ponds in the country (National Aquaculture Center, 2004 *In* Maluwa & Gjerde, 2006) is less than 10% of the loss in chambo production in Lake Malombe alone since 1980, and illustrates that pond aquaculture will not soon be able to be a quantitatively important supplement of fish to Malawian diets. Although participation is growing in pond aquaculture, there are serious constraints, such as lack of suitable land for pond construction, seasonal sources of water, frequent droughts, lack of access to fish fry for stocking, lack of technical knowledge, lack of on-farm resources, and inaccessibility to potential markets that continue to constrain development at the smallholder scale of operation.

In view of the declining capture fisheries in Lake Malawi and the resulting low economic returns, Maldeco Fisheries Ltd, which is the largest commercial fishing company in Lake Malawi, formed Maldeco Aquaculture Ltd which established the first lake-based aquaculture

industry (hereafter cage culture/farming) of *Oreochromis karongae* in the shallow southeast arm of the lake in 2004. *Oreochromis karongae* is one of the three closely related tilapiine cichlid species which are endemic to Lakes Malawi and Malombe and the upper Shire River (Fig. 1.1). The three species, *O. karongae, O. lidole* and *O. squamipinnis*, are marketed as "chambo" and have historically attracted high prices at the market. As a consequence, they have been overfished by both commercial and artisanal fishers leading, in part, to their collapse relative to other species in Lake Malawi, Lake Malombe and the upper Shire River (Fig. 1.2). Occasionally, a few cages have also been stocked with *Oreochromis shiranus*, locally called "makumba", in monoculture or polyculture with *O. karongae* (Fig. 1.3). *Oreochromis karongae, O. shiranus, O. mosambicus* and *Tilapia rendalli* constitute the four tilapia species used in pond aquaculture in Malawi, which started in 1960 (Maluwa & Bjerde, 2006). Cage culture is a new aquaculture technique in Malawi and Africa in general about which not much information currently exists. It overcomes significant limitations of pond aquaculture such as water and land availability neceessary for earthen pond construction. Its establishment in Lake Malawi by Maldeco Aquaculture Ltd is important to the country, particularly because it also aims to restore the declining endemic chambo fishery by relieving fishing pressure on the wild stocks and by releasing part of the production back into the wild. Cage culture will also provide more fish for markets which have , historically, been of great economic and nutritional value to the people of Malawi. To expedite restoration, the southern tip of the south east arm, which is generally considered to be the breeding ground for the chambo fishery, has been closed from commercial and artisanal trawling. The cage production of chambo is expected to

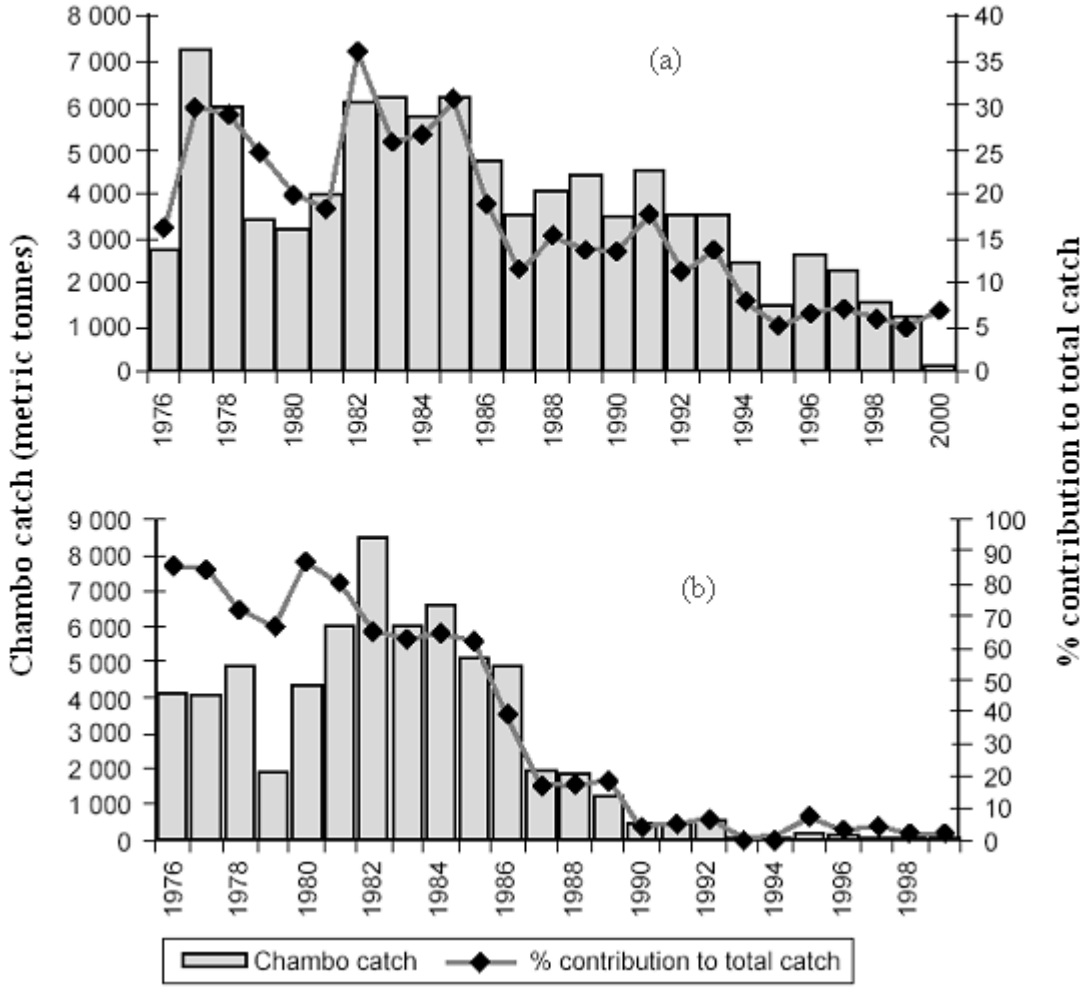


Fig. 1.2. Annual Chambo catches and their percent contribution to total catches in Lake Malawi (a) and Lake Malombe (b). Source: Banda et al., 2005.

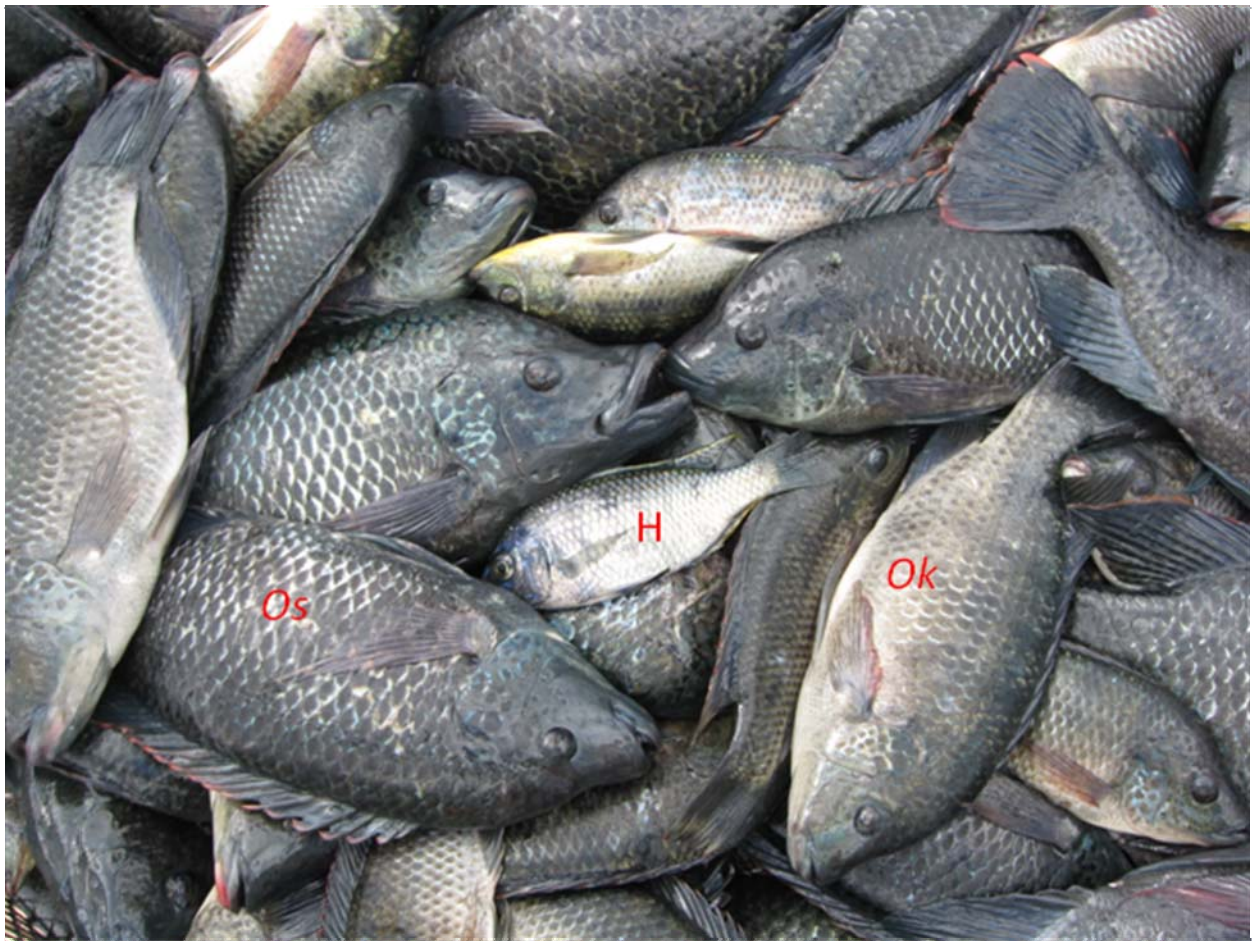


Fig. 1.3. *Oreochromis karongae* (*Ok*) *and Oreochromis shiranus* (*Os*) harvest from fish cages in the southeast arm of Lake Malawi. Sometimes wild haplochromine fish (H) can be trapped inside cages during net installation. See **Appendix 3** for a video file of the *Oreochromis* fish species in cages in Lake Malawi.

reach a maximum annual production of about 3,000 tonnes from circular grow-out cages measuring 14.7 m in diameter and ~6 m in depth. Although cage culture is, in principle, a simple and less costly means of rearing fish (Beveridge, 1984) that overcomes the constraints of water and land which limit pond culture, cage culture has been shown elsewhere (e.g. Pillay, 1992; Wu, 1995; Troell & Berg, 1997) to be associated with numerous environmental impacts. This is primarily because production involves large input of high-quality artificial feeds to fish cages of which only a small portion is consumed and assimilated by the cultured fish species (Folke & Kautsky, 1989, Troell, 1996) leading to large discharges of organic and inorganic wastes to the surrounding environment. The discharged organic wastes, rich in nitrogen (N) and phosphorus (P), have the potential to pollute the waters and underlying sediments causing local eutrophication, increased turbidity, loss of biodiversity and other impacts. Using mass balance studies in temperate systems, Gowen and Bradbury (1987), Holby and Hall (1991) and Hall et al. (1990, 1992) have shown that only about 24% of C, 31% of N and 31% of P added through feed are removed at harvest as fish biomass. In addition to affecting the quality of the water and sediments around and underneath the cages, the cage-derived organic wastes may present an important allochthonous source of organic matter for many consumers in the receiving ecosystem and may therefore modify the ecosystem's planktonic and benthic food webs.

For cage culture to be accepted socially, environmentally and economically, it requires a rigorous assessment of its impacts on the hosting ecosystem and its biodiversity. This is especially true when cage culture is conducted in multi-purpose lakes containing sensitive fauna such as the endemic and stenotopic fish species found in Lake Malawi. Lake Malawi is a great

lake not only because of its large size (30,800 km^2) but also because it is an ecologically unique ecosystem that harbours the world's highest diversity of freshwater fish species. Over 99% of the fish species, estimated between 700 and 1000 species (Turner, 1999), are cichlids endemic to the lake. From an evolutionary science perspective, Lake Malawi's cichlid species, which are hypothesized to have radiated within the lake, have been described to be of equivalent or greater value to evolutionary science as Darwin's finches of the Galapagos Islands (http://whc.unesco.org/en/list/289). In addition, the fish species in the lake are more diverse and abundant in the littoral zone of the lake, where the cages are being established, than in the vast ultra-oligotrophic pelagic zone (Fryer, 1957).

Ribbink (1994) described the Lake Malawi cichlids as being more sensitive to exploitation and habitat degradation because they are trophic specialists, lay fewer eggs than less specialized fishes, invest heavily in parental care and occur in small population sizes in addition to their stenotopic characteristics (limited geographic distribution). Because of the low fecundity, they are slower to recover from population depletion than their eurytopic counterparts (Ribbink, 1994). The risks of nutrient enrichment on these specialized fishes has recently been demonstrated in Lake Victoria where loss of diversity of cichlid species has been related to reduced water clarity caused by eutrophication (Seehausen et al 1997). Sensitivity of the cichlid species in Lake Malawi to fishing pressure also has previously been demonstrated when the initial intensive, demersal trawling in the lake resulted in a rapid decline in catches and species richness after only two years of fishing (Turner, 1977a, 1977b). Without careful attention to siting and management of fish cages, wastes generated in cages could severely impact the littoral

fish communities that currently sustain the artisanal fishing industry. With the great depths of Lake Malawi (mean depth of 264 m), cage culture siting will certainly be limited to very nearshore locations to reduce operational costs and reduce security risk which means that impacts on local water and sediment quality will be imposed on the littoral fish species and riparian populations using the lake for drinking water and domestic uses.

1.2 Thesis objectives and hypotheses

This study was the first evaluation of environmental aspects of the cage culture operation in Lake Malawi since its establishment in 2004 and it intended to find answers to two important questions with regard to the establishment of the cage fish farm (1) How will cage culture and associated waste products impact the quality of the water and underlying sediment at the cage fish farm in the southeast arm of the lake and (2) How will cage culture and associated waste products affect the pelagic and benthic food webs of organisms including wild endemic fish species around the cage area? The study focused on three main objectives as follows:

Objective 1-Chapter 2: Construct mass budgets for C, N and P in the cage fish farm in the southeast arm of Lake Malawi to establish cage-specific potential for point source loading of organic material and nutrients to the lake. This was done using production records gathered from the cage culture operators, Maldeco Aquaculture Ltd. The hypothesis was that a significant proportion of C, N and P in feed supplied to the fish cages would be lost to the surrounding environment in form of dissolved and particulate nutrients in the water column and underlying

sediments respectively. Based on international experience primarily in temperate waters, ≤31% of the added nutrients will be harvested in form of fish biomass at harvest.

Objective 2-Chapter 3: Measure dissolved and particulate nutrients in water column along a transect through the cage area in the south east arm of Lake Malawi. Additional measurements were also obtained using a Sea Bird CTD profiler, Fluoroprobe, Tidbit Temperature loggers, Secchi depth disc, and FSI acoustic current meter. The hypothesis was that turbidity, chlorophyll and dissolved and particulate C, N and P concentrations in the water column would be higher closer to than further away from the fish farm. However, the realized impact was expected to depend on the rate of water renewal to the cages which might be rapid in this large dynamic lake with steep coast lines.

Objective 3-Chapter 4: Trace the flux of particulate and dissolved C and N derived from the cage wastes into the planktonic and benthic food webs in the SEA of Lake Malawi using $\delta^{13}C$ and $\delta^{15}N$ ratios in feed and feces of cultivated fish, muscle tissue of cultivated and wild fish, phytoplankton, zooplankton, sedimenting material collected in sediment traps, benthic macroinvertebrates and sediment samples collected in the vicinity of the cages and at control sites. The hypothesis was that the organic wastes discharged from the fish cages would represent a primary source of organic matter for animal consumers in the planktonic and benthic food webs around the fish cages. Consequently, the impact of cage feeding may extend into the food web of the surrounding fish community and may disrupt natural feeding patterns.

Chapter 5: Chapter 5 presents general conclusions from this study and recommendations for sustainable development of cage culture in Lake Malawi.

Chapter 2
Carbon, nitrogen and phosphorus loading from tilapia fish cages in the southeast arm (SE Arm) of Lake Malawi

2.1 Abstract

Lake Malawi is a Great Lake not only because of its large size (30,800 km^2) but also because it is one of the world's most important hotspots of biodiversity, hosting between 700-1,000 fish species composed largely of sensitive endemic cichlids. The fish in the lake are more abundant and more diverse in the littoral zone, because of high food abundance, than in the vast ultraoligotrophic pelagic waters. In addition, the littoral fish in the lake have very restricted habitat preferences (i.e., highly stenotopic) making them very sensitive to habitat degradation which is already occuring in the lake's narrow littoral zone. Degradation of the nearshore areas may come in very different ways including catchment degradation, overfishing, and poorly sited and managed cage culture facilities. Studies from across the world show that fish cages can be important point-sources of nutrients to hosting water bodies due to their ability to generate large amounts of wastes rich in carbon (C), nitrogen (N) and phosphorus (P) which are discharged into the surrounding environment. Cage culture in Lake Malawi started in 2004 in nearshore waters of the southeast arm of the lake, with a production target of 3,000 tons fish/year. Around 50 cages were operational in the lake by 2009 with a target harvest of 20 tons fresh fish/cage/year. In 2007 data were aggregated from cage feed and production records provided by cage operators to produce a mass balance for C, N and P added to cages in the form of feed and juvenile fish for

22 production cycles in order to estimate nutrient losses from cages to the surrounding environment. The study showed that nutrient losses from the cages are significant and exacerbated by poor feed quality, overstocking, stocking of premature fingerlings and use of lower than recommended feeding rates to grow the fish. Results also show that C, N and P nutrients were supplied to the cages primarily through fish feed accounting for between 90 and 99% of all three inputs. There was a strong correlation between feed supply and levels of C, N and P losses. Solute and sedimentation losses from cages to the surrounding environment accounted for a significant proportion of all the three nutrients (C, N and P) in feed inputs; between 81-91% for C, 59-80% for N and 85-92% for P. The inefficient production strategies necessitated longer production cycles (376±42 days) and more feed usage to achieve market sizes of 300g/fish than if recommended stocking and feeding rates were followed.

2.2 Introduction

In most African countries the importance of fish as the cheapest and main source of dietary protein has been affected by the global decline in capture fish stocks. In Malawi, for example, fish used to contribute more than 70% of the total dietary protein consumed in the country in the 1980s, but rapid population growth, environmental degradation and over-fishing combined with use of illegal fishing gear over the years have overwhelmed the capture fisheries in natural ecosystems reducing the protein supply from fish to the current <30% (Banda et al, 2005). Recent reports (e.g. Banda et al, 2005) show that the annual per capita fish consumption

in the country has dropped from 14 kg of fish in the mid 1970s to below 3 kg in 2003. The consequence of this has been an increase in protein deficiency cases in the country, particularly amongst vulnerable women and children (Lauderdale, 2000). It was for the purpose of mitigating the protein deficiency problem in the country that cage culture was established in Lake Malawi in 2004. The means to achieving the goal of protein sufficiency was to increase the supply of food fish in the country through the use of grow-out cages where fish were expected to grow faster and at higher densities with the use of high energy artificial feeds than the wild population, as well as through re-stocking the wild population by releasing part of the caged fish population into the wild.

Cage fish farming has been shown elsewhere (e.g. Jones, 1990; Pillay, 1992; Wu, 1995; Troell & Berg, 1997) to be associated with numerous potential environmental impacts on hosting ecosystems primarily because it generates large amounts of organic wastes rich in nitrogen (N) and phosphorus (P). Mass balance studies in a number of cage fish farms around the world (e.g. Gowen & Bradbury, 1987; Holby & Hall, 1991; Hall et al., 1990, 1992) have shown that only about 24% of carbon (C), 31% of nitrogen (N) and 31% of phosphorus (P) added through feed are removed at harvest as fish biomass. The wastes generated (e.g., uneaten feed, feces and urinary wastes) are released into the surrounding environment with potential consequences of polluting the waters and underlying sediments. Impacts associated with wastes, both organic matter and nutrients, discharged from fish cages include eutrophication, toxic algal outbreaks, increased turbidity, decreased oxygen concentrations and loss of biodiversity. For these reasons a rigorous assessment of potential impacts of cage wastes on the receiving ecosystem and

biodiversity therein is required for cage culture to be accepted socially, environmentally and economically.

Reducing the environmental footprint of cage aquaculture is especially important when cage culture is conducted in water bodies with specialized and unique fauna such as the endemic and stenotopic fish species found in Lake Malawi. Lake Malawi is a great lake not only because of its large size (30,800 km^2) but also because it contains a high diversity of fish species (700-1000 species), composed largely (>99%) of sensitive endemic cichlid species with very specific habitat requirements and limited geographic distribution. These cichlids are more abundant in the littoral zone of the lake where food resources (periphyton and phytoplankton) are more abundant than in the vast ultraoligotrophic pelagic zone (Fryer, 1957). Other workers (e.g. Ribbink, 1994) have suggested that the cichlid species in Lake Malawi could be more sensitive to exploitation and habitat degradation because they are trophic specialists, lay fewer eggs than the less specialised fishes (eurytopes), invest heavily in parental care and occur in small population sizes in addition to occupying very specific habitats. Because of their low fecundity, they are slow to recover from population depletion than the less specialised fishes with high fecundity (Ribbink, 1994). The great depth of Lake Malawi (mean depth of 264 m) and the prevalence of strong winds over the lake will certainly limit cage culture sites to very nearshore locations for good anchorage of the cages, safety of feeding crews and security of product and infrastructure. Nearshore siting means that impacts on local water quality will be imposed on the littoral fish species and riparian populations using the lake for drinking water and domestic uses. The risk of nutrient enrichment of the water to these specialized Lake Malawi fishes has recently been

demonstrated in Lake Victoria, where loss of cichlid species has been related to reduced water clarity caused by eutrophication (Seehausen et al., 1997). The low resilience of the cichlid populations in Lake Malawi to fishing pressure has also been demonstrated when the initial demersal trawling in the lake resulted in a rapid decline in catches and species richness after only two years of fishing (Turner, 1977a; 1977b). This is a clear demonstration that without careful attention to location and management of cage culture operations in the lake, fish farms could severely impact productive littoral endemic and stenotopic cichlid fish communities that are currently the basis of local subsistence fisheries.

The challenge in Lake Malawi's cage culture is, therefore, how to take advantage of the cage culture's positive aspect of mitigating protein deficiency in the country through increased supply of food fish to households while safeguarding the lake and the essential capture fishery against potential negative aspects of the operations.

In this chapter I report estimates of C, N and P mass balances of operating cages in Lake Malawi and examine possible steps to reduce loadings from the cages.

2.3 Materials and methods

2.3.1 Study site

Cage culture in the southeast arm of Lake Malawi was started in 2004 by Maldeco Aquaculture Ltd in response to the collapse of the capture chambo fishery in the lake (Fig. 1.1). The chambo fishery, which consists of *Oreochromis karongae*, *O. lidole* and *O. squamipinnis*, used to be the main target catch for Maldeco Fisheries Ltd, the largest capture fishery trawler in

the lake, until the fishery collapsed by >80% in all its endemic habitats of Lake Malawi, the upper Shire River and Lake Malombe in the late 1990s (Figs. 1.1 & 1.2). However, Maldeco Aquaculture selected *Oreochromis karongae* (Fig. 1.3) for cage culture cultivation in the lake. Occasionally, *Oreochromis shiranus* (Fig. 1.3), locally known as "makumba", was also cultivated in separate or mixed cages with *O. karongae*. Both Maldeco Fisheries Ltd and Maldeco Aquaculture Ltd are subsidiary companies of the Press Corporation Ltd, a group of 10 companies.

The fish farm has a land-based broodstock and nursery facility located about 2.5 km from the lake for fingerling production, and lake-based grow-out cages deployed in nearshore waters approximately 1 km from shore in the southeast arm of the lake (Fig. 1.1). The southeast arm is a relatively more productive part of the lake and used to be the main fishing grounds for the indigenous chambo fishery before stocks collapsed. To ensure a timely supply of good quality fingerlings, the broodstock and nursery facility has expanded during the first four years to 46 earthen breeding ponds, 14 earthen fry ponds (also called hapas) and 16 nursery concrete tanks where fingerlings are sorted into size classes. The whole facility is supplied with clean lake water pumped through a 2.5 km long pipeline from the lake.

The fish cages are circular (14.7 m diameter & ~6 m deep, Fig. 2.1) made of heavy duty nylon nets hanging on floating circular poly-vinyl chloride (pvc) pipes. The nets extend to about a meter above the water. The cages have predator nets below the water surface and above to keep piscivorous birds and other predators away from the reared fish. The cages were spread over a rectangular area which measured about 300x800 m, and 16 cages by mid 2007 deployed in four

Fig. 2.1. Fish cages in the southeast arm of Lake Malawi showing size and distance between cages and between cage rows.

rows with four cages per row. The distance between cages in a row and between rows was about 10 and 100 m respectively. Thirty two (32) new cages were added in between the old cage rows toward the end of the same year (2007) making a total of 48 cages as the distance between cage rows was reduced to less than 70 m.

Feeding in both land- and lake-based facilities is done by hand with high energy content artificial feed which was initially imported into the country from Spesfeed (Pty) Ltd in South Africa. The fish farm operators grouped the feeds as tilapia starter, tilapia grower, tilapia finisher and tilapia broodstock based on target fish sizes and type. Feed importation ended in 2007 when Maldeco Aquaculture Ltd installed and commissioned a feed mill complete with pelletiser at their company premises at Maldeco in Mangochi district. Feeding rates used by Maldeco staff were determined based on mean fish weights and total fish biomass in the cages based on sampling to estimate fish biomass, which was done once or twice a month.

2.3.2 Study approach

Estimates of feeding, feed consumption and nutrient losses presented here were made from production records, harvests and daily feed and mortality records, that were aggregated from Maldeco Aquaculture Ltd for all 22 production cycles[1] done between 2004 and 2008 in Lake Malawi. I estimated C, N and P losses from cages using a simple input-output mass balance model according to Schmittou (2006). This method of estimating nutrient loading from fish farms is preferred, particularly in high current environments such as in the open waters of Lake

Malawi where empirical measurements may not be possible. Most fish farms in the world have been located in sheltered or semi-sheltered, low-current areas (Carson, 1988) where organic enrichment of the underlying sediments and nutrient buildup in the surrounding waters make it easier to monitor impacts of the cage operations. In contrast, the cages in Lake Malawi are in near-shore but open waters, occasionally experiencing high currents from prevailing trade winds which blow over the lake. Feeding trips to the cages are at the mercy of the weather over the lake almost throughout the year. According to Eccles (1974) the prevailing winds for much of the year tend to blow along the axis of the lake, with some directional channelling by the lake's morphometry. During the rainy season, between November and March, the winds tend to be northerly while southerly winds become predominant between April and September. In October, strong easterly winds dominate, blowing across the lake particularly in the morning (Eccles, 1974). Eccles (1974) reported that these winds often attain speeds of up to 40 km/h generating surface waves 3-4m high over the lake. Associated currents caused by these wind events may quickly disperse discharged nutrients and organic matter from the cages, making empirical measurements rather difficult and sometimes unrealistic. However, there were many indirect signs around the cages in Lake Malawi which showed the significance of organic and nutrient loading from the cages. During feeding hours, large schools of wild fish species flocked to the cages to consume feed particles escaping from the cages (Fig. 2.2). The fish populations around the cages were dense enough that they could be easily caught by quickly dipping a hand net on unsuspecting wild fish near the water surface and trapping them against the inner cage net (Fig.

[1] Production cycle is the period between stocking and harvesting of cages.

2.3). I conducted a sampling program at the fish farm between January and December of 2007 to determine impacts of the cage operations from empirical measurements. Samples collected during the study included caged and wild fish species, water, plankton, sediments, benthic macroinvertebrates, and unconsumed feed and feces sedimenting below the cages and these results are reported in Chapter 3.

The input-output mass balance method of estimating nutrient loading requires that we accurately know the weights and chemical compositions of inputs (feed and juveniles) and outputs (mortality removal and harvest) as described by Schmittou (2006) and Sowles and Churchill (2004). The difference between inputs and outputs is generally assumed to have been discharged into the surrounding environment with potential to cause pollution. All data, except the chemical compositions of the inputs and outputs, were amassed from production record sheets we obtained from Maldeco Aquaculture Ltd. The record sheets had columns for production date, production day number, feed ration per cage (kg/day/cage), total feed from stocking to i^{th} day (kg), mean fish weight (g) determined through monthly sampling, total biomass in cage (kg), mortality removed, and total number of live fish in cage. Mid-morning and mid-afternoon surface water temperatures were also measured and recorded in these sheets. Water temperature was measured with an Hg-bulb thermometer until February 2007 when we deployed Tidbit temperature loggers at 0, 1, 2, 5, 10 and 13 m depths at the cage area. For each cage, a growth curve of mean fish body weight (g) vs time (days) was generated using production day number and mean fish weight (g) determined from monthly sampling. The curve was used to estimate daily total biomasses of live and dead fish in each cage needed to estimate

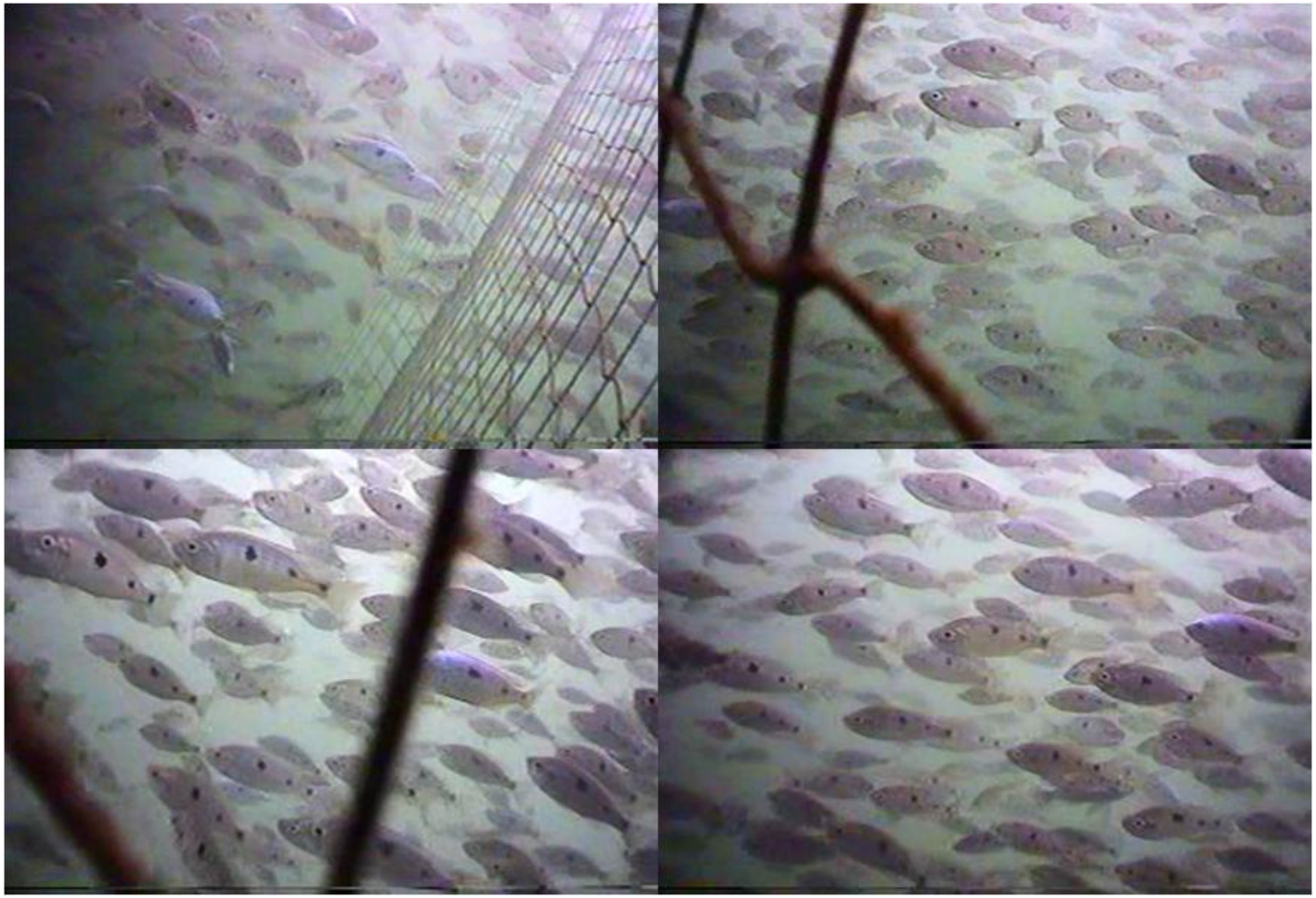


Fig. 2.2. The abundance of wild fishes around fish cages in the southeast arm of Lake Malawi. See **Appendix 4** and **Appendix 5** for video files of wild fish outside cages in Lake Malawi.

Fig. 2.3. Catching wild fish with a hand net in surface waters around fish cages in the southeast arm of Lake Malawi.

nutrient losses from the cages. C and N contents (%) of feed and fish were determined as a byproduct of stable ^{13}C and ^{15}N isotope analysis at the Environmental Isotope Laboratory at the University of Waterloo, Canada, using a Delta Plus Continuous Flow Stable Isotope Ratio Mass Spectrometer (Thermo Finnigan/Bremen-Germany) coupled to a Carlo Erba Elemental Analyzer (CHNS-O EA1108-Italy) while their P content (%) was determined by the molybdate-ascorbic acid method after persulfate digestion of ashed samples (Stainton et al., 1977). P was analyzed in 23 caged fish which weighed between 3 and 571g body wet weight sampled from 13 cages (2 fish/cage) at various production periods of 26 - 443 days. I also measured ratios of dry weight to wet weight of captured caged fish and their feed. The mass balance equations take the following forms:

$$\mathbf{C\ (Loading,\ kg) = (F*C_{DF} + J*C_{DJ}) - (H*C_{DH} + M*C_{DM})} \qquad \mathbf{(i)}$$

$$\mathbf{N\ (Loading,\ kg) = (F*N_{DF} + J*N_{DJ}) - (H*N_{DH} + M*N_{DM})} \qquad \mathbf{(ii)}$$

$$\mathbf{P\ (Loading,\ kg) = (F*P_{DF} + J*P_{DJ}) - (H*P_{DH} + M*P_{DM})} \qquad \mathbf{(iii)}$$

where F, J, H and M are dry weights (kg) of feed supplied, juveniles stocked, fish harvested and total mortality (dead fish removed from cages), respectively, as recorded at the end of each production cycle. C_{DF}, C_{DJ}, C_{DH} and C_{DM} are carbon contents in dry feed, dry juveniles, dry harvests and dry mortality, respectively, expressed as % of dry weight. N and P contents in N and P loading equations have been expressed in the same manner as C content in the C loading equation above.

Apart from estimating C, N and P loadings from the fish cages, production records were analyzed to determine production strategies, particularly in relation to feeding and stocking rates,

which could have exacerbated nutrient and organic losses from the fish cages. Fish stocking rates for the 22 production cycles and their daily feeding rates were compared to recommended rates suggested by Schmittou (2006). Nutrient and carbon loadings resulting from the realized feeding and stocking rates were compared to potential C, N and P loadings that would emanate from the fish cages if recommended feeding and stocking rates were employed to grow fish in the cages in Lake Malawi. The purpose of this analysis was to determine if following recommended feeding and stocking rates could be beneficial for cage culture operators and for the lake with regard to reduction of feed costs and nutrient (C, N and P) loading from the cages to the surrounding environment.

2.4 Results

Between 2004 and 2008, total production of fish from the cage fish farm in Lake Malawi was 283,123 kg wet weight from 22 production cycles that used 734,244 kg of artificial high energy feed. This represents an average feed conversion ratio (FCR = feed supplied/body weight gain) of 2.7. Among the 22 cages, annual cage fish production averaged (±SE) 12,319±834 kg and FCR varied between 2.1 and 3.9. FCR tended to be lower in recent production cycles. Production cycles were, on average (±SE), 376±9 days long.

The mean percent composition per dry weight of fish was 49.2% C, 14.0% N and 0.8% P, while the feed was 44.6% C, 5.8% N and 0.9% P (Table 2.1). The composition of P per wet weight of reared chambo was 0.2%, which is closer to %P in wild *Oreochromis lidole* (another chambo species) of 0.3% reported by Mumba & Jones (2005). The percent C and N in reared

Table 2.1. Carbon (C), nitrogen (N) and phosphorus (P) compositions (±SD) of dry weight of feed, juvenile fish, fish harvest, fish mortality, feces of caged fish and particulate cage wastes collected in sediment traps deployed under the fish cages.

	%C	%N	%P	$\%H_2O$
Feed	44.57±2.63	5.75±0.83	0.87±0.11	5.20±0.22
Fish	49.17±1.96	13.96±0.76	0.81±0.16	69.00±2.36
Feces	32.22±4.13	2.78±0.34	NA	NA
Cage-derived PM	25.70±7.40	2.87±0.85	NA	NA

Table 2.2. Carbon (C), nitrogen (N) and phosphorus (P) mass balances estimated from 22 production cycles completed between 2004 and 2008 at a cage fish farm in the southeast arm of Lake Malawi. The numbers for each mass balance are means±SE in kg/cage/year of C, N and P. Numbers in parentheses refer to losses as % of C, N and P added to the cages through fish feed.

Nutrient	Juvenile fish	Fish feed	Mortality	Fish harvest	Solute & Sediments
C	104.9±14.7	14211.9±1194.6	98.0±25.0	1877.7±127.1	12341.0±1090.1
	(0.2-3.4)	(96.6-99.8)	(0.1-4.3)	(9.4-17.7)	(81.5-90.8)
N	29.8±4.2	1833.5±154.1	27.8±7.1	532.7±36.1	102.7±125.4
	(0.4-7.3)	(92.7-99.6)	(0.2-9.5)	(20.6-38.9)	(59.3-79.7)
P	3.8±0.5	602.2±50.6	3.6±0.9	68.2±4.6	534.3±46.8
	(0.2-3.0)	(97.0-99.8)	(0.1-3.7)	(8.0-15.2)	(84.1-92.1)

C, N and P input sources include fish feed and juvenile fish while outputs include fish harvest and mortality. Losses are in form of solutes and sediments released from the cages into the surrounding environment. The ranges in the percentages represent variation amongst 22 production cycles which were analyzed. Mean fish harvest per cage between 2004 and 2008 was 12, 874 kg.

fish observed in this study are similar to levels in twenty fish species from Bark Bay, Lake Superior, studied by Tanner et al (2000). The percent C and N in the twenty fish species ranged between 42.8 and 48.4% (mean of 45.1%) and between 9.6 and 12.3% (mean of 11.3%) respectively (Tanner et al., 2000). Percent P in the Bark Bay fish species was however higher at 2.5% than levels observed in reared fish in Lake Malawi. Just as in the study by Tanner et al (2000), life stage had little influence on the C, N and P composition of the analyzed fish from Lake Malawi cages.

C, N and P mass balances (Figs. 2.3a, b & c; Table 2.2) show that C, N and P nutrients were supplied to the cages primarily through fish feed while stocked juvenile fish were a minor addition, as feed accounted for between 92.4 and 99.8% of all three inputs. On average, each cage released about 12,341, 1,303 and 534 kg/year of C, N and P respectively into the surrounding environment in dissolved and particulate forms. The dissolved and particulate losses accounted for a significant proportion of all the three nutrient (C, N and P) inputs to the cages through feed, with 81-91% of feed C, 59-80% of feed N and 84-92% of feed P being lost from the cages. There was a strong linear relationship between feed supply and C ($r^2 = 0.996$), N ($r^2 = 0.968$) and P ($r^2 = 0.997$) losses from the cages (Fig. 2.5). The slopes of the linear regressions were 0.024 for P but 0.040 and 0.385 for N and C, respectively.

On average, about 1000 kg of C, 106 kg of N and 43 kg of P wastes were generated per ton of fish harvested between 2004, when cage operations started, and 2008. Therefore, if the trend continues, a fish farm(s) producing 15,000 tonnes fish/year would generate and discharge approximately 15,000 tonnes of C, 1584 tonnes of N and 650 tonnes of P into the environment,

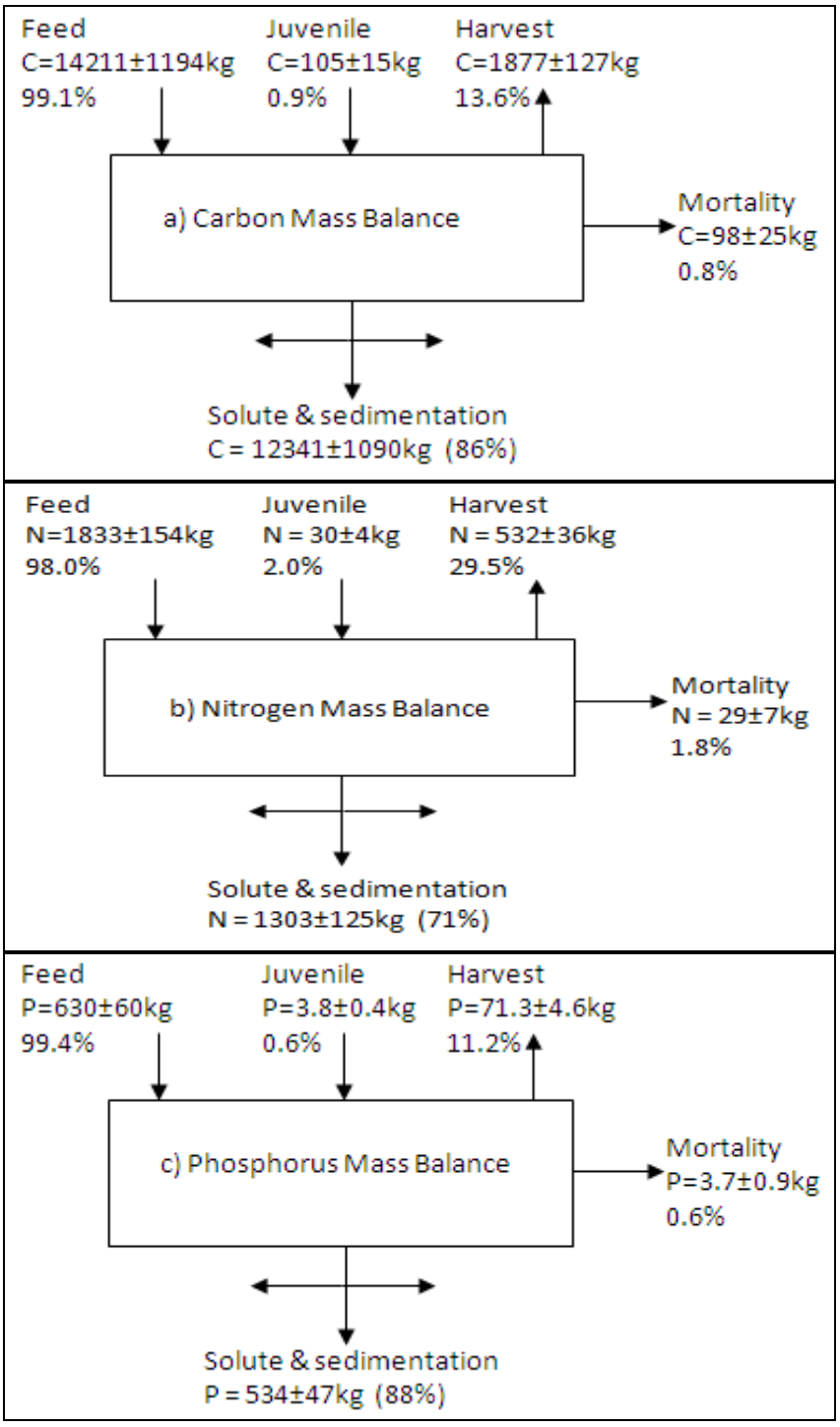


Fig. 2.4. Mass balances for carbon (a), nitrogen (b) and phosphorus (c) in fish cages per year in Lake Malawi. The annual amounts are mean±SE.

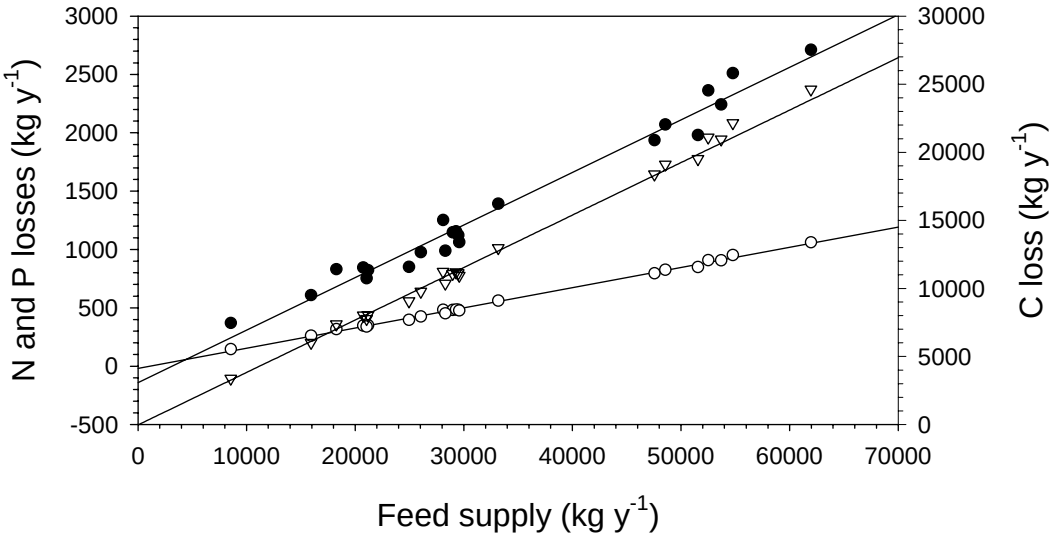


Fig. 2.5. Relationship between feed supply and N (closed circles), P (open circles) and C (open triangles) losses from fish cages in Lake Malawi. Notice the slopes of the linear plots. The equations of the regressions are N = 0.040*FS – 9.586 (r^2 = 0.968); P = 0.024*FS – 1.585 (r^2 = 0.997) and C = 0.385*FS –31.326 (r^2 = 0.996) on the right y-axis. **Note:** different y-scale for C.

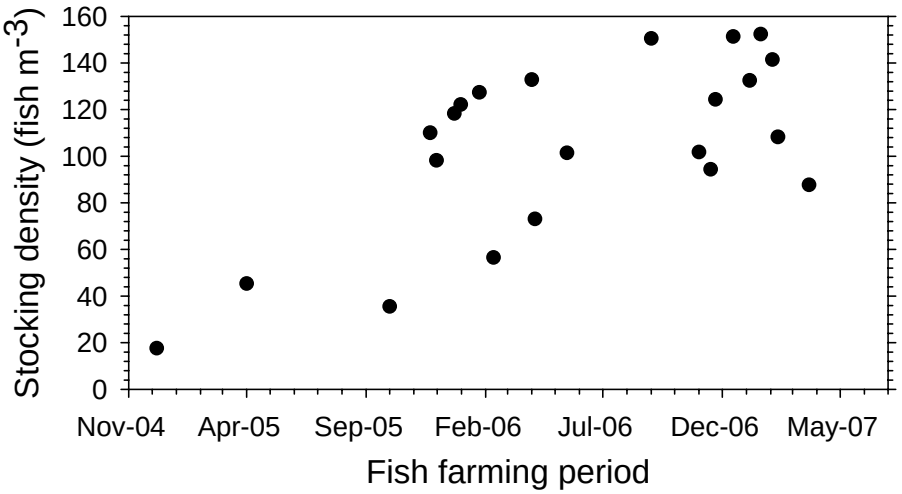


Fig. 2.6. Fingerling stocking density (# fish m^{-3}) in the fish cages in Lake Malawi. The stocking density has been expressed per unit volume of the cages (total cage volume = 764 m^3).

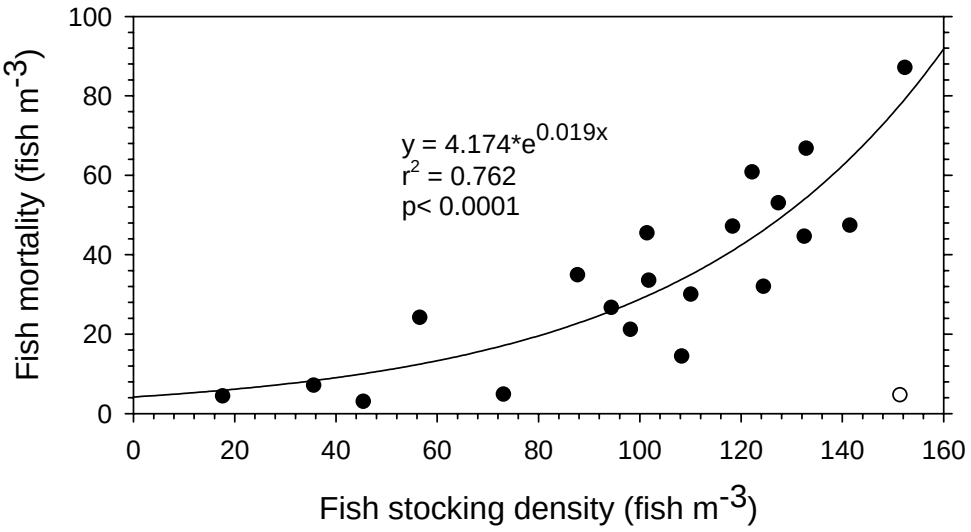


Fig. 2.7. Relationship between fish stocking density per unit cage volume (fish m^{-3}) and fish mortality per unit cage volume (fish m^{-3}). One cage (open circle) was omitted from the regression analysis because it was outlying.

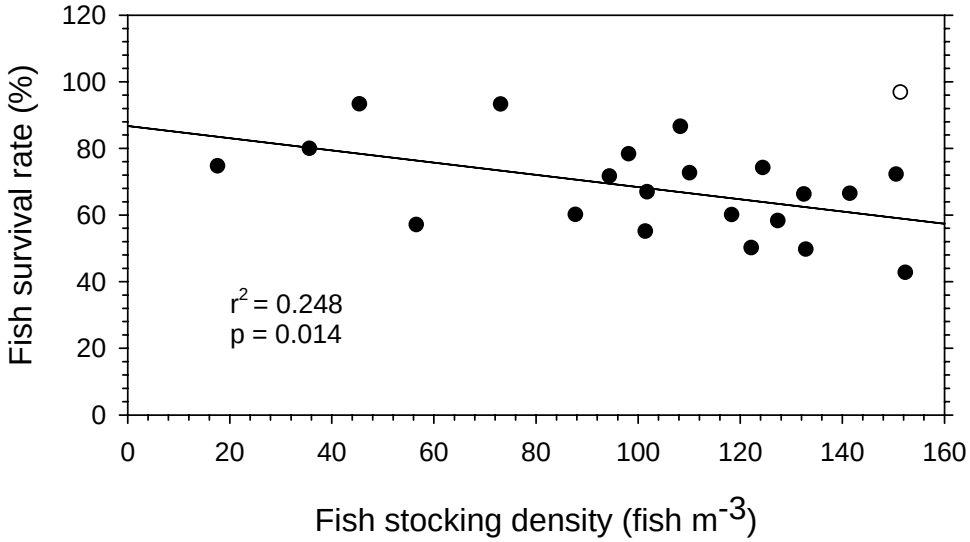


Fig. 2.8. Relationship between fish stocking density per unit cage volume (fish m^{-3}) and fish survival rate (%) in the cages. One cage (open circle) was omitted from the regression analysis because it was outlying.

with potential to cause pollution that may be detrimental to both the environment and the fish farm(s). These discharges are equivalent to 1249, 113 and 21 megamoles/yr of TC, TN and TP respectively. The megamoles of nutrients discharged have been compared to riverine DOC, TOC, TDN, TN, TDP and TP loads measured in 13 major rivers draining into Lake Malawi from the Malawi catchment (Table 2.3). The FF/RR ratios in Table 2.3 show that the amounts of C, N and P discharged from the fish cages would be 1-420 times higher than DOC, TDN and TDP levels in all the 13 major rivers draining into Lake Malawi, including the highly disturbed Linthipe, Songwe and Dwangwa Rivers. Similarly, the C, N and P losses from the fish cages at 15,000 tonnes/year production capacity would be 1-210 times higher than the TOC, TN and TP in all the rivers except Linthipe and Songwe Rivers (Table 2.3). The FF/RR ratios for "Total RR" (i.e., Total Riverine loading from 13 rivers) show that production of 150,000 tonnes fish/yr in cages in Lake Malawi would generate TC, TN and TP equal to or more than total loadings of TOC, TN and TP observed in the 13 rivers draining the Malawi catchment area.

Analysis of fingerling stocking rates since 2004 when operations started at the fish farm showed that stocking rates in the cages were variable, ranging from a minimum of 18 fish m^{-3} (13,500 fingerlings per cage) in 2004 to a maximum of 152 fish m^{-3} (116,500 fish per cage) in 2007 (Fig. 2.6) with a mean stocking rate of 104 fish m^{-3} for the period. The stocking density has increased to 130,000 fish per cage in 2009 (M. Mkandawire –Maldeco Aquaculture's farm manager, pers. comm.). The sizes at which the fingerlings were stocked were also variable, ranging from 3.9 g to 24.8 g (mean of 8.9 g). Maldeco Aquaculture's target total harvest was 20 tonnes fish/cage/yr (~26 kg m^{-3}) and ≥300 g individual fish weight at harvest. Consequently,

Table 2.3. Comparison between TOC, TN and TP discharges from a hypothetical fish farm (FF) in Lake Malawi producing 15,000 tonnes fish/yr.and discharges of DOC, TOC, TDN, TN, TDP and TP in 13 major rivers (RR) draining into the lake. All discharges are in megamoles/yr. Maldeco Aquaculture Ltd in the southeast arm Lake Malawi has a permit to produce 3,000 tonnes fish/yr.

River	FF-TC	RR-DOC‡	FF/RR	RR-TOC‡	FF/RR	FF-TN	RR-TDN‡	FF/RR	RR-TN‡	FF/RR	FF-TP	RR-TDP‡	FF/RR	RR-TP‡	FF/RR
Nadzipulu	1249	13	96.1	72	17.3	113	1.1	102.7	5.3	21.3	21.0	0.14	150.0	1.5	14.0
Namkokwe	1249	10	124.9	63	19.8	113	1.4	80.7	5.2	21.7	21.0	0.12	175.0	0.6	35.0
Linthipe	1249	1,310	1.0	10,043	0.1	113	112.1	1.0	487.9	0.2	21.0	4.76	4.4	49.7	0.4
Bua	1249	735	1.7	1,217	1.0	113	42.2	2.7	70.7	1.6	21.0	1.10	19.1	5.8	3.6
Dwangwa	1249	114	11.0	1,153	1.1	113	12.1	9.3	76.9	1.5	21.0	1.62	13.0	10.0	2.1
Dwambadzi	1249	35	35.7	75	16.7	113	1.5	75.3	4.0	28.3	21.0	0.21	100.0	0.5	42.0
Mlowe	1249	7	178.4	15	83.3	113	0.3	376.7	0.8	141.3	21.0	0.05	420	0.1	210.0
Luweya	1249	55	22.7	92	13.6	113	2.8	40.4	5.7	19.8	21.0	0.17	123.5	0.6	35.0
S. Rukuru	1249	88	14.2	401	3.1	113	5.6	20.2	25.1	4.5	21.0	0.35	60	2.5	8.4
N. Rumphi	1249	47	26.6	507	2.5	113	2.6	43.5	33.7	3.4	21.0	0.25	84	2.7	7.8
N. Rukuru	1249	103	12.1	792	1.6	113	7.0	16.1	57.4	2.0	21.0	0.52	40.4	5.0	4.2
Lufira	1249	52	24.0	448	2.8	113	5.1	22.2	35.8	3.2	21.0	0.24	87.5	2.1	10.0
Songwe	1249	518	2.4	4,480	0.3	113	58.1	1.9	365.7	0.3	21.0	2.11	10.0	17.5	1.2
Total RR*	1249	3,087	0.4	19,358	0.1	113	251.9	0.4	1,174	0.1	21.0	11.64	1.8	98.6	0.2

‡ data from Hecky et al., 2003.
*Total RR = Total Riverine = total of annual loadings from all 13 rivers in the table.
FF/RR >1 means fish farm discharge is greater than riverine discharge.
FF/RR <1 means riverine discharge is greater than fish farm discharge.
S. Rukuru = South Rukuru, N. Rumphi = North Rumphi and N. Rukuru = North Rukuru.

these stocking rates suggest overstocking based on the Schmittou (2006) equation for estimating stocking densities:

$$N_c = W_c/w \qquad \textbf{(iv)}$$

where N_c is the number of fish to stock per m^3 of cage, W_c is the expected total weight of fish per m^3 of cage at harvest and w is the desired mean weight of fish also at harvest. Fish mortality in the cages increased exponentially with increasing stocking rates (Fig. 2.7). Conversely, fish survival rates declined as stocking rates increased in recent cages (Fig. 2.8). These mortality numbers in cages were generally highest at the beginning of production periods and declined exponentially as production continued to the end (Fig. 1.6). Spikes in mortality were sometimes observed, especially in cages with *Oreochromis shiranus*, after partial harvest for sale or for broodstock selection (Fig. 2.9). Similar observations have been reported from a number of fish cage studies (e.g., Sunde et al., 1998) after stressful size-grading operations to separate small from large individuals to avoid unnecessary social interactions between different size groups. There was higher mortality in recent cages than in earlier ones, as recent cages tended to have a higher stocking rate for immature fingerlings (Fig. 2.10). Due to the large variation in size of fingerlings stocked, the stocking density, mortality in cages and inadequate feeding, the number of fingerlings stocked in the cages was better related to total harvest at the end of production cycle ($r^2 = 0.692$) than to stocking biomass ($r^2 = 0.210$). Similar results have been observed with the African catfish *Claris gariepinus* in ponds (Hogendoorn & Koops, 1983) and in cages (Hengsawat et al., 1997), *Oreochromis niloticus* in cages (Daungsawasdi et al., 1986) and channel catfish *Ictalurus punctatus* also in cages (Storck & Newman, 1988).

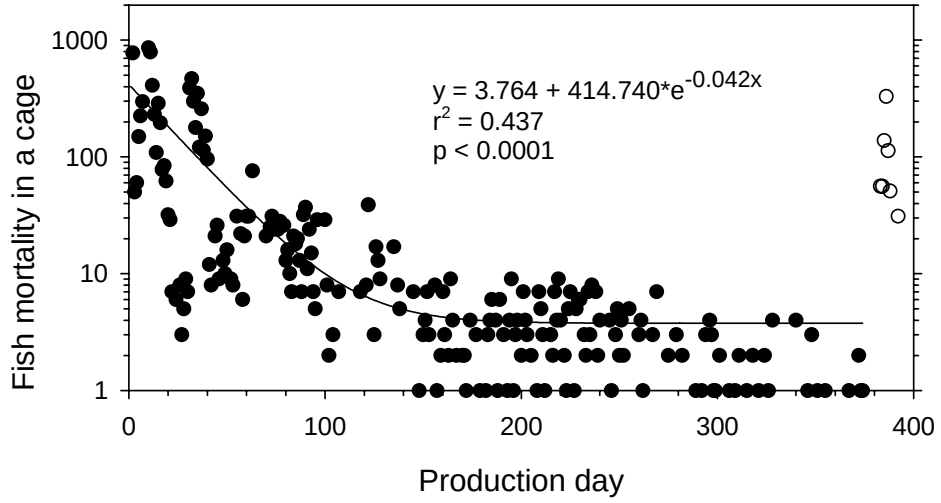


Fig. 2.9. An exponential drop in fish mortality with time observed in most of the fish cages in Lake Malawi. Note the spike in mortality late in the production cycle attributed to partial harvesting for broodstock selection and/or sales.

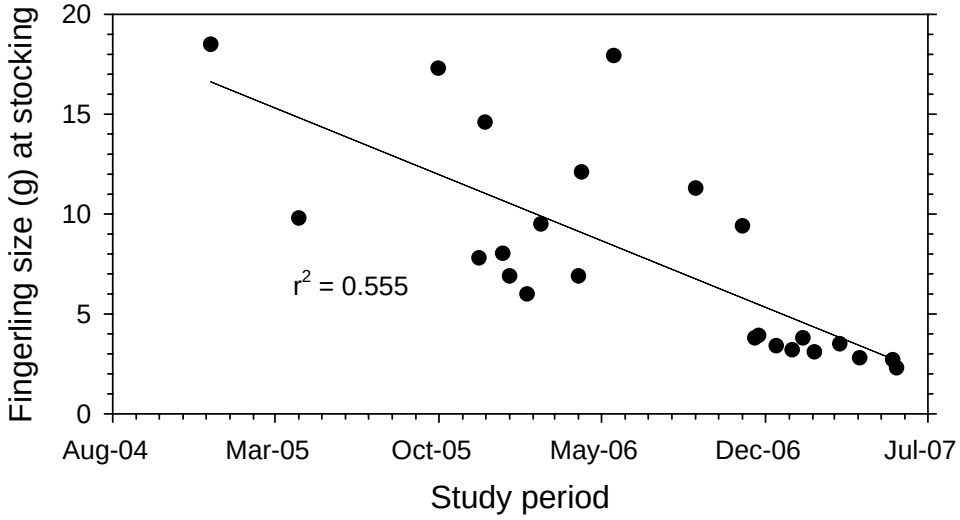


Fig. 2.10. Individual mean weight of fingerlings stocked in cages in Lake Malawi between 2004 and 2008.

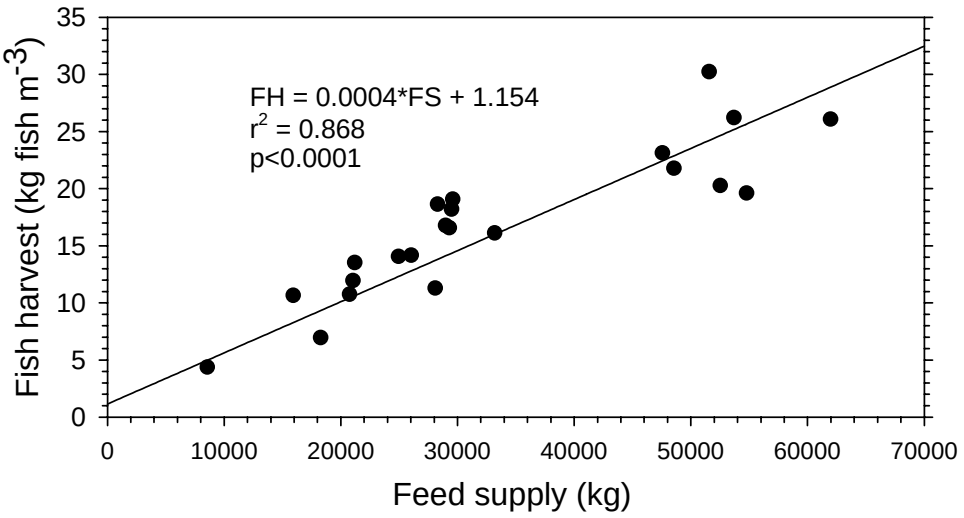


Fig. 2.11. Relationship between feed supply (kg) and total fish harvest per unit cage volume (kg fish m^{-3}).

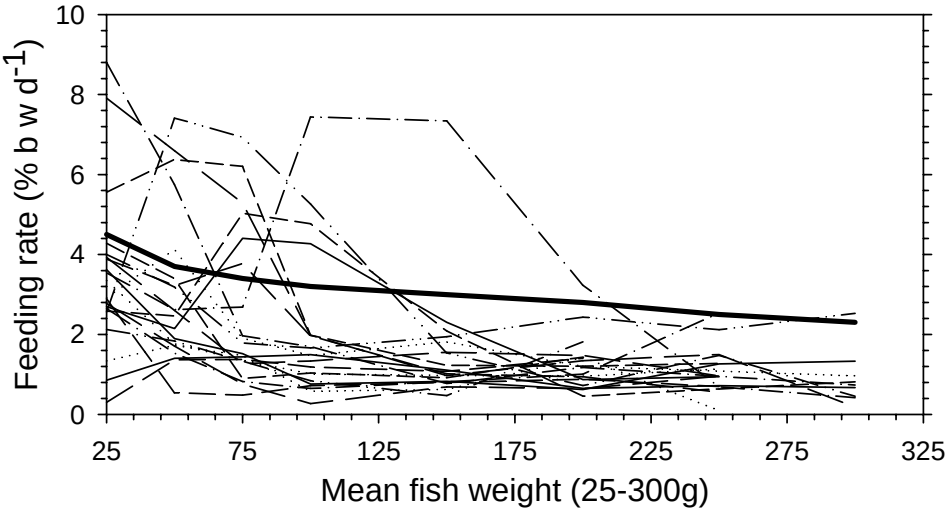


Fig. 2.12. The relationship between mean fish weight (g) and feeding rates (% body weight d^{-1}) in fish cages in Lake Malawi (thin lines) compared with recommended feeding rates (thick line) suggested by Schmittou (2006). Notice that most feeding rates in Lake Malawi fish cages were consistently lower than the recommended rates except for a few rates which were initially higher but dropped abruptly to below recommended rates as fish grew to larger size.

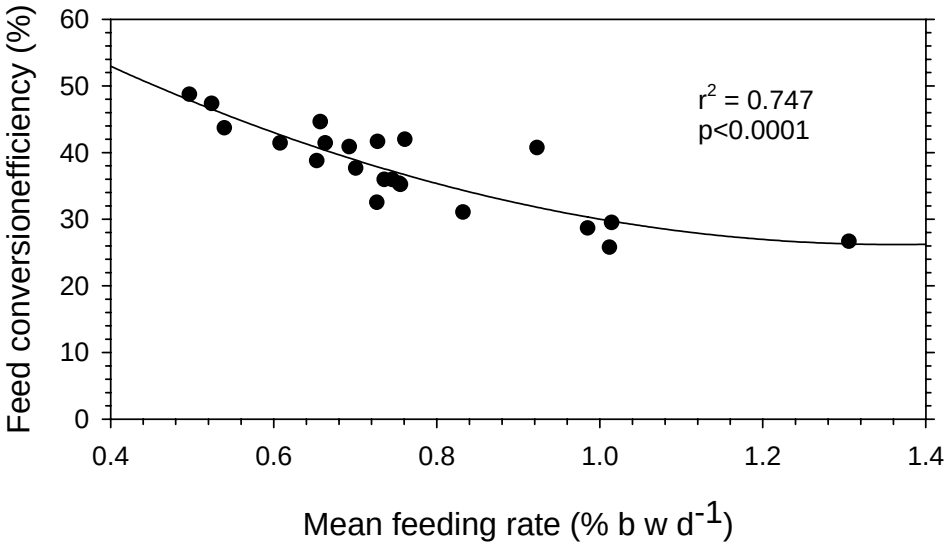


Fig. 2.13. Relationship between mean feeding rate (% body weight d^{-1}) and feed conversion efficiency (FCE, %) in Lake Malawi fish cages. Mean feeding rate was calculated from initial body weight, final body weight and number of days in each production cycle as in equation (vi). FCE = body weight gain/feed supplied*100

Feed quality and quantity, as well as how it is provided to reared fish (i.e. feed rations and intervals between feedings), are important aspects for consideration in fish farming since harvest from cages is directly related to the quality and amount of feed supplied. There was a strong linear regression between fish harvest (kg/m^3) and feed supply (kg) (Fig. 2.11). Feeding techniques (feed ration sizes and intervals between feedings) can also affect waste generation from cages. The wastes generated in cages can also be directly related to the amount of feed supplied (Fig. 2.5). Feed utilization efficiency in the cages was low as indicated by high mean feed conversion ratio (FCR) and low mean feed conversion efficiency (FCE±SD) values of 2.75±0.52 and 37.54±6.48%, respectively. The feeding rates (Fig. 2.12) show that rates used in the fish farm in Lake Malawi were consistently lower than the rates recommended by Schmittou (2006) which suggests that the caged fish were probably underfed. Lower feeding rates were more apparent in recent production cycles than in earlier ones. The underfeeding suggestion is supported by higher FCE at lower feeding rates than at relatively higher feeding rates (Fig. 2.13). Feed quality was not measured in this study but may be inferred from mean feed conversion ratio (FCR) and feed conversion efficiency (FCE) which are supposed to be low and high, respectively, with high quality feed. The mean fish weight at harvest declined from ≥300 g in 2005 to ~200 g in 2008 despite rearing the fish longer in the cages (Fig. 2.14a). The low mean fish weights at harvest were due to declining fish growth rate in the cages (Fig. 2.14b) associated with high stocking densities and inadequate feeding rates in the fish cages.

Estimates were made of the time and amount of feed that would be required under recommended feeding rates to raise the fish to the same body weights using the same feed, and

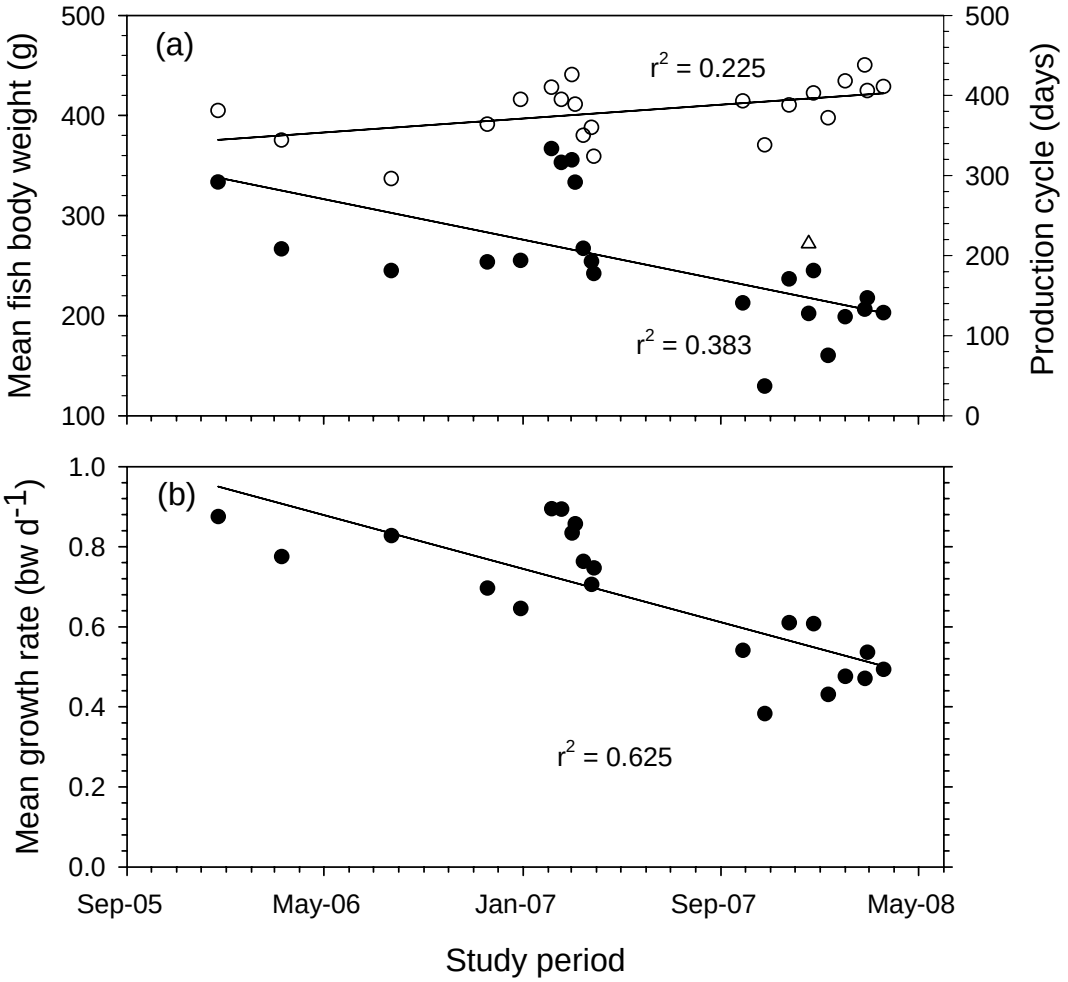


Fig. 2.14. Declining mean fish body weight at harvest ((a), closed circles; $r^2 = 0.383$) and growth rate ((b); $r^2 = 0.625$) over time attributed to inadequate feeding rates. Consequently, production cycle became longer ((a); open circles; $r^2 = 0.225$). Note an outlying production cycle of 272 days (open triangle) in (a).

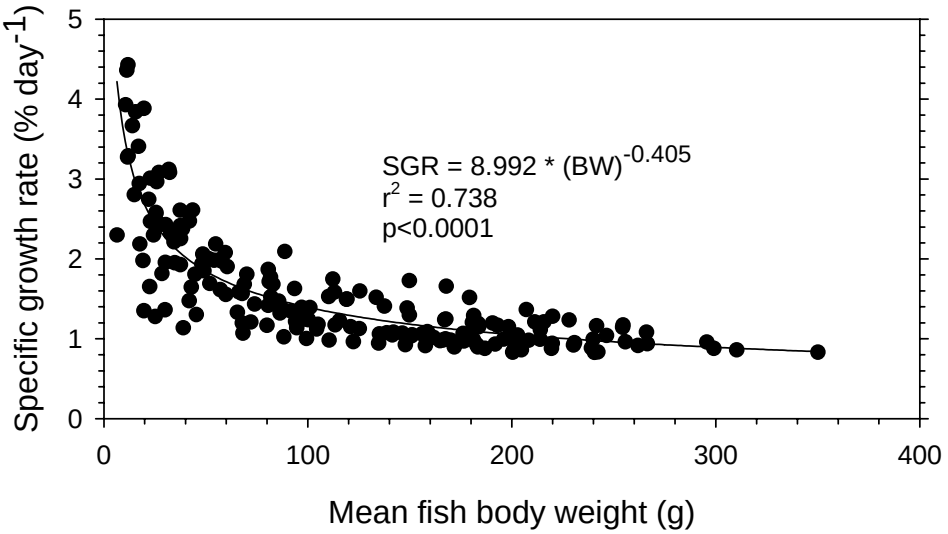


Fig. 2.15. Relationship between mean fish body weight (g) and specific growth rate (% per day) of cage reared chambo in Lake Malawi.

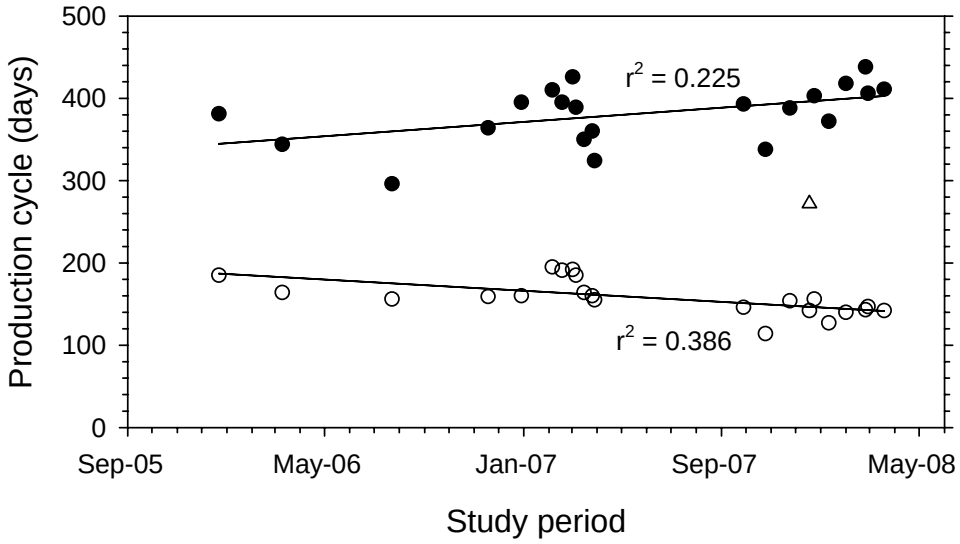


Fig. 2.16. Observed lengths of production cycles (closed circles) compared against predicted lengths of production cycles (open circles) that would have been achieved if recommended feedings rates were followed to yield the same fish body weights.

these ideal rates were compared to the realized feeding and growth rates. First, a model that relates fish body weight (g) to recommended feeding rates (% body weight day^{-1}) for tilapia in grow-out cages was applied that used data from Schmittou (2006). The model is of form:

$$\mathbf{FR = -0.90*LN(}\boldsymbol{BW}\mathbf{) + 7.385} \qquad \textbf{(v)}$$

where FR is feeding rate (% body weight day^{-1}), *BW* is fish body weight (g) and LN is natural logarithm. This model was then used to determine appropriate feeding rates for the reared chambo fish species weighing between 10 and 600 g in cages. The feeding rates were in turn used to determine potential specific growth rate (SGR) values (% day^{-1}) for the fish body weights from Fig. 2.15. After determining potential specific growth rates of the fish at various body weights and corresponding size-dependent feeding rates, the total amount of feed and the number of days that would be required to grow the fish to required market sizes was calculated. Time (*T*, days) was calculated after rearranging the SGR equation below for *T*.

$$\mathbf{SGR = 100*[LN(}W_f\mathbf{) - LN(}W_i\mathbf{)]/}\boldsymbol{T} \qquad \textbf{(vi)}$$

where W_f is final weight of fish, W_i is initial weight of fish and *T* is time in days. Fish consuming recommended feed rations are predicted to grow faster than poorly fed fish (Fig. 2.16). A significant relationship was observed between the realized and predicted (recommended) amounts of feed supplied to the cages (Fig. 2.17). The total amount of feed that would be supplied to the cages if recommended feeding rates were used is smaller by 233,061 kg than the amount that was actually supplied. Addition of 233,061 kg of feed to the cages is equivalent to 85,081 kg C, 8,901 kg N and 3,888 kg P that would be wasted in producing fish of equivalent

size under idealized feeding rates. This extra feed also means extra costs to the production cycle and reduces profit margins.

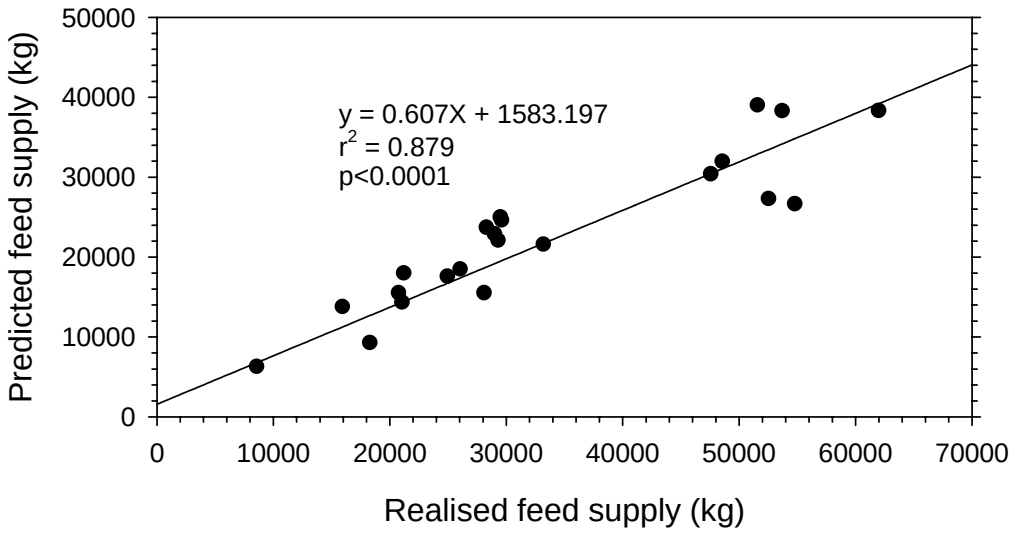


Fig. 2.17. Correlation between realized (observed) feed supply (kg) and predicted (recommended) feed supply (kg) in fish cages in Lake Malawi.

2.5 Discussion

Negative impacts of cage culture have been reported in many parts of the world (Jones, 1990; Pillay, 1992; Wu, 1995; Troell & Berg, 1997; Vista et al., 2006). However, the cage culture established in Lake Malawi is the first in the African great lakes. As such, very little experience-based assessment was available to guide production or to project potential impacts to the surrounding environment and underlying sediments. Aquaculture in Africa has mainly been land-based in earthen ponds, and very few cage culture operations have been carried out at a large scale, the prominent ones being in Lake Kariba in Zimbabwe/Zambia and in Lake Volta in Ghana. Lakes Kariba and Volta are man-made reservoirs built for hydro-electricity and/or irrigation and have only ancillary fisheries with relatively low biodiversity and no endemism. These reservoirs are totally different from the three African Great Lakes in terms of size and functionality so that impacts of cage culture operations observed in these smaller systems may not be applicable to the same extent in the three great lakes. For example, Lake Kariba has a water capacity of 160 km^3 and a short water retention time of 3 years (Karenge & Kolding, 1995) while Lake Malawi has a much larger water volume of 8,500 km^3 and a longer water residence time of about 140 years (Bootsma & Hecky, 1993). Unlike in Lake Malawi, P concentrations in Lake Kariba can be easily predicted using Dillon and Rigler (1974) and Vollenweider (1976) phosphorus budget models (Thornton & Walmsley, 1982). The short water retention time in Lake Kariba implies that eutrophication in the lake can be more quickly remedied than in Lake Malawi, where nutrient build-up may not be easily noticed because of the immense dilution capacity of the lake. This study is the first to assess the potential loss of nutrients from cage culture in the African Great Lakes.

2.5.1 C, N and P losses from the cages

Previous mass balance studies in temperate climate cage fish farms (e.g. Gowen & Bradbury, 1987; Holby & Hall, 1991; Hall et al., 1990, 1992) have shown that only about 24% of C, 31% of N and 31% of P added through feed were removed at harvest while the remainder of added food was lost into the environment. Our analysis of production records from the fish cages in Lake Malawi showed that only 14% of C, 30% of N and 12% of P added through feed were converted into body mass of fish at harvest. Nutrient losses in mortality removed from cages were minimal at 0.70-1.83% (on average) of total C, N and P inputs so that large quantities of the nutrients added through feed were basically discharged into the surrounding environment in dissolved and particulate forms. Feed was the main source of these nutrients while stocked juveniles contributed only <2% to the total C, N and P added to the cages. The loss percentages show that N was more retained than C and P by the ”chambo” (tilapia) fish species reared in Lake Malawi when compared to similar percentages in other water bodies and species. This suggests that the feed was N deficient compared to P which has been confirmed by C:N and N:P molar ratios in feed of 10.2±2.1 and 14.2±4.1 which indicate moderate N deficiency and P sufficiency in the feed, respectively (Chapter 3, see Guildford & Hecky, 2000),

Fish production in cages is directly related to the amount of feed supplied to the reared fish and feed is usually the most significant component of the fish farming budget. As such, better feeding strategies, such as feeding rations and intervals, can significantly reduce feed costs. Most culturists reduce daily feed rations slightly to cut on feed costs without reducing the growth rate of the fish, i.e., so that fish still exhibit optimal growth rates. The use of consistently

lower than recommended feeding rates by Maldeco Aquaculture in their fish cages may have been a strategic step to reduce feed costs. The benefits however may not have been realized because the feeding rates were too low for optimal fish growth. The lower feeding rates, among other factors, are thought to have reduced the growth rates of the reared chambo species so that the fish needed longer production periods and consequently more feed to grow to required market size of ≥300 g. A study by Clark et al (1990) showed similar effects of feeding rates on fish growth rates using Florida red tilapia reared in floating marine cages. They reared red tilapia for 84 days at six feeding rates (demand, ad libitum, 110%, 90%, 70% and 50% of estimated satiation feeding rate, defined as the percent of body weight consumed daily) and they finally observed that specific growth rates of the reared red tilapia were lowest at 50% and highest at 110% and ad libitum feeding rates. An ad libitum feeding rate is when fish are fed by hand to satiation based on observed consumption on a daily basis.

The reared fish species were frequently observed grazing the periphytic algae on fish cage nets, which I interpreted as feed supplementation since the fish were probably underfed, but the periphytic algae may not have been assimilated as suggested by significantly different $\delta^{13}C$ and $\delta^{15}N$ signals of the reared fish and the periphytic algae (Chapter 4).

2.5.2 The difference between realized and recommended feeding rates

I did calculations to estimate the amount of time and feed that would be required to raise the chambo to similar body weights observed in Lake Malawi cages using feeding rates recommended by Schmittou (2006). From the calculations it became clear that fish production

using recommended feeding rates suggested by Schmittou (2006) would have reduced feed usage by between 13-51% (mean of 30%; see Fig. 2.17) and production time by between 47-67% (mean of 58%; see Fig. 2.16) compared to the amount of feed and production time required under the lower actual feeding rates. Dadzie (1992) made the same point, that it is possible for African culturists to grow *Oreochromis niloticus* to market size of 450 g within 150 days under intensive culture in cages. Intensive culture of course does include use of appropriate feeding rates so that caged fish have access to the right quality and quantities of all necessary dietary nutrients at all times during production. Even fast-growing tilapia species such as *O. niloticus* have exhibited reduced growth rates when subjected to poor feeding regimes. Since chambo is naturally a relatively slow growing tilapiine, Maldeco Aquaculture Ltd needs to adopt appropriate feeding strategies to maximize growth in cages and achieve good economic returns. By using lower feeding rates, Maldeco Aquaculture Ltd wasted approximately 233,061 kg of feed which translated into large amounts of C, N and P losses to the environment.

2.5.3 Stocking density

Another important factor that might have contributed to lowering of production performance in grow-out cages in Lake Malawi is stocking density. Cage stocking density refers to the number or weight of fish stocked per unit volume or area of a cage. An overstocked cage may show reduced production performance caused by declining water quality, or fish having inadequate access to feed and space (Schmittou, 2006). It has been shown that water quality and

access to feed lowered by overcrowding limit production performance before restricted space becomes a factor (Schmittou, 2006). To help cage culturists, Schmittou suggested a simple equation for estimating stocking density of fish in grow-out cages as follows: $N_c = W_c/w$ where N_c is the number of fish to stock per m^3 of cage volume, W_c is the expected total weight of fish per m^3 of cage at harvest and w is the desired mean weight of fish also at harvest. For example, to economically achieve a target harvest of 20 tonnes/cage and average fish weight of 300 g, Schmittou's equation predicts a stocking density of 87 fingerlings/m^3 or 66,667 fingerlings/cage only. Similarly, a stocking density of 65 fingerlings/m^3 equivalent to 50,000 fingerlings/cage will be required to achieve a target harvest of 20 tonnes/cage but at an average fish weight of 400 g. In comparison with stocking densities predicted by this equation, the cages in Lake Malawi were probably overstocked with fingerlings. On average, the 22 cages were stocked with between 13,400-116,400 (mean of 79,282) fingerlings/cage or approximately 104±38 fingerlings/m^3. Current reports indicate that stocking density in the cages has even increased to 130,000 fingerlings per cage (M. Mkandawire, farm manager, pers. comm.) which would be excessive and result in inefficient use of expensive juvenile stock.

Because many of the fish cages were overstocked and inadequately fed (i.e., lower feeding rates than recommended by Schmittou, 2006), fish production in the cages required extended grow-out periods to achieve 20 tonnes total harvests and ≥300 g individual fish weights which were production targets for good economic returns for the cage operators. Other workers (e.g., Dadzie, 1992) however support overstocking of cages to maximize production and profits. I did indeed observe a direct and strong positive relationship between stocking density and cage

harvest in the cages in Lake Malawi, but stocking density was also exponentially related to fish mortality in the cages (Figs. 2.6 and 2.7). I also observed a decrease in individual fish body weight at harvest as stocking density of fingerlings increased in cages (Figs. 2.5 & 2.13a) which agrees with observations made by Coche (1977), Guerrerro (1980) and Carro-Anzalotta and McGinty (1986). Consequently, in most cages, both production targets of 20 tonnes fish/cage/yr total harvest and individual fish body weight of ≥300 g at harvest were not achieved. The relationship is that high stocking density negatively affects individual fish growth rates leading to small individual fish at harvest, despite increasing total yield while low density will produce large individual fish but low total yield (Watanabe et al., 1990; Siddiqui et al., 1997). High stocking density also promotes social hierarchy of fish in cages as large fish grow bigger and faster while subordinate small fish grow much slower thereby increasing size variation in harvest (El-Sayed, 2006).

In addition, most of the densely stocked cages were stocked with immature fingerlings (~5 g; Fig. 2.10). The size at stocking has direct effect on tilapia growth and yield (El-Sayed, 2006). According to Huguenin (1997), stocking larger and older fingerlings shortens their grow-out period in cages in addition to lowering fish mortality rates and overall risk of an investment. Since mortality rates in the cages were highest soon after stocking, it is not clear if mortality in the cages was solely due to overstocking or due to both overstocking and fingerling sizes. Problems associated with stocking immature fingerlings have been observed by many workers (e.g. Dadzie, 1992) and include longer grow-out periods even for cages stocked with fast growing fish species. Dadzie (1992) noted that stocking cages with *O. niloticus* fingerlings

weighing 40 g could expedite harvesting by 30 days compared to stocking 20 g fingerlings. This is because fingerlings grow slower in cages than in hatchery ponds/tanks where water temperatures are generally higher and the fingerlings are fed several times a day to satiation as opposed to twice a day in grow-out cages.

2.5.4 Consequences of nutrient losses to the surrounding environment

Among some eighteen elements known to be required by primary producers in aquatic systems, N, P and Si have received the greatest attention as elements likely to limit algal growth (Hecky & Kilham 1988). However, only diatoms require Si to grow. Algal growth in marine ecosystems is generally said to be limited by N while freshwater systems are said to be limited by P (Vollenweider, 1968; Vollenweider et al., 1974; Schindler, 1978). However, concentrations of both N and P in surface waters of the African Great Lakes of Malawi, Tanganyika and Victoria are low (Bootsma & Hecky, 1993), which suggests the potential for limitation by both N and P in these lakes. Several research methods employed by many workers in the last decade have, indeed, confirmed seasonal N and P limitation in Lake Malawi (Hecky et al., 1996; Guildford & Hecky, 2000; Guildford et al., 2000, 2003, 2007; Patterson & Kachinjika, 1993; Gondwe et al., 2008), so that any external source of N and/or P, including cage culture, to the lake's epilimnion may increase the risk of eutrophication (Cornel & Whoriskey, 1993). The mass balance model estimates that mean C, N and P losses from the fish cages in Lake Malawi were at 1000 kg of C, 106 kg of N and 43 kg of P per tonne of fish harvest at C:N molar ratio of 11.0,

C:P molar ratio of 60.0 and N:P molar ratio of 5.5. The low N:P ratio of 5.5 of the cage wastes would favour proliferation of N_2-fixing algal species in the surrounding environment, some of which are potentially toxic to fish and other animals including humans (Kiirikki et al., 2001). Fish feeds are generally low in Si (Mente et al., 2006) so that diatom growth may not increase due to feed losses into Si deficient environments (Parson et al., 1978; Doering et al., 1989). However, upwelling in the southeast arm of Lake Malawi (see Fig. 3.8) replenishes Si levels in the epilimnion at regular basis (Owen & Crossley, 1992) so that N and P losses from fish cages may still increase diatom biomass in the water column. It is important to realize that the amount of C, N and P wastes getting discharged into the surrounding environment from fish cages in Lake Malawi is proportional to the amount of feed supplied (Fig. 2.5) which is in turn proportional to fish biomass in the cages (Fig. 2.11). As a result, the nutrient losses from the cages are expected to increase significantly as the number of cages increases and more feed is supplied to the reared fish to achieve the operators target annual harvests. Table 2.3 show that nutrient losses from the production of 15,000 tonnes fish/yr would be comparable to dissolved nutrient loadings in Linthipe, Dwangwa, Bua and Songwe Rivers but much higher (9-420 times) than levels in smaller rivers draining into the lake. Linthipe, Dwangwa and Songwe Rivers are the most disturbed large river systems in the Malawi catchment area. A study carried out in 1997 by Hecky et al (2003) showed that the three rivers had the highest runoff and annual loads of suspended sediments, suspended nutrients and dissolved nutrients of any of their neighboring rivers of comparable basin area.

The community species composition and biomass of the phytoplankton are regulated primarily by nutrient availability in what is generally called the bottom-up trophic cascade (McQueen et al, 1989). In most cases, algal biomass is altered first before changes in community species composition occur (Cornel & Whoriskey, 1993). However, these changes resulting from input of nutrients from fish cages in Lake Malawi may not be easily noticed in the water column surrounding the cages because of the openness of cage area to wind currents which facilitate rapid dispersion to the surrounding environment (Chapter 3).

2.6 Conclusion

Poor production performance and high nutrient losses from fish cages in Lake Malawi were influenced by suboptimal production strategies, primarily stocking of cages with immature fingerlings, overstocking of the cages and use of lower feeding rates than recommended to grow the fish. The consequences of these strategies were that the caged fish grew slowly and therefore required prolonged production cycles, which were on average 376±42 days, to reach market sizes. Therefore, they used more feed than necessary. In contrast, use of proper feeding and stocking rates predicts that 32% and 42% reductions in feed usage and production time, respectively, would be possible to achieve the same fish body weights observed under current management strategies. This study has therefore shown that in terms of feed costs, it would be cheaper to grow fish following recommended feeding and stocking rates. Since fingerlings grow relatively faster in ponds than in cages, it is important to stock mature fingerlings (~15 g for

chambo) in cages to expedite fish growth and harvesting. Stocking mature fingerlings of approximately the same size also ensures that the fish grow at the same rate, which is necessary to prevent hierarchal patterns from developing amongst the fish in cages. Size hierarchy in cages leads to food monopoly by large and aggressive fish while the smaller subdominant fish stagnate due to lack of access to feed.

This study has revealed the high nutrient losses from the fish cages in the southeast arm of Lake Malawi. Much of the losses are due to substandard production strategies used during the study period. Since this is the first cage culture operation in the African Great Lakes, these nutrient losses are expected to drop on a cage basis as management decisions regarding production strategies improve with time, experience and research. However, if cage culture continues to expand in Lake Malawi, it will become an important new source of nutrient loading to the lake and could lead to degradation of coastal water quality as the low N:P loading rate of these nutrient losses from cages would enhance growth of noxious and potentially toxic cyanobacteria species.

Chapter 3
Physical-chemical measurements in the water column along a transect through a tilapia fish farm in Lake Malawi, Africa

3.1 Abstract

The empirical data presented here show that at the 2007 production capacity of ~200 tonnes fish/yr, impacts of the cage wastes in the water column in the vicinity of the cages were minimal despite the large estimated discharges from the cages. No significant differences were observed in concentrations of dissolved and particulate nutrients (NH_4^+, NO_3^-, SRP, PC, PN and PP), extracted chlorophyll, chlorophyll fluorescence, dissolved oxygen, TSS and PAR extinction coefficient between the cage site and the control stations upstream or downstream of the fish farm. It is suggested that the cage wastes were rapidly and efficiently dispersed by relatively fast water currents which averaged 9.3 cm s^{-1} below the fish cages. Consumption of the cage wastes by large numbers of wild fish species which aggregated around the fish cages and their movement in the vicinity of the cages also enlarged the area of dispersion of the cage wastes through defecation and served to dilute impact on an areal basis.

3.2 Introduction

Lake Malawi is the southern most African great lake located in the western arm of the East African Rift Valley system. Lake Malawi is a critical resource to the riparian countries of Tanzania, Mozambique and Malawi and to the world at large. The lake has the highest number of freshwater fish species in the world estimated between 700 and 1000 species, composed largely (>99%) of endemic cichlids (Snoeks, 2000). The unique biodiversity in the lake is threatened by increasing demographic pressure and high poverty among the riparian populations that have accelerated environmental degradation in the catchment areas through deforestation, biomass burning and cultivation of marginal high slopes and wetlands. In addition, rains in Malawi facilitate considerable runoff due to their high volume over short time periods causing massive loadings of sediments and nutrients into the lake (Hecky et al., 2003). These inputs are changing littoral water conditions, particularly around river mouths, and are beginning to alter algal communities and their sedimentation rates in the more densely populated and shallower southern portion of the lake, particularly the south east arm (Puchniak, 2005). Data presented in chapter 2 (Fig. 2.4 and Table 2.3) indicate that the establishment of cage culture in the lake can add another major source of C, N and P nutrients into the lake's epilimnion. Gowen and Bradbury (1987), Holby and Hall (1991), Hall et al (1990, 1992) and Kaushik (1998) estimated that 70 to 80% of nutrients added to fish cages through feed are lost to the surrounding environment in the form of unconsumed feed, fish feces and metabolic wastes in temperate systems. Similarly, in Lake Malawi, estimates from production records indicate that 71 to 88% of nutrients added through feed are lost to the environment from fish cages (Chapter 2). The amount of nutrients

discharged from the fish cages are directly proportional to the amount of feed used so that as the fish farm expands, the levels of nutrients discharged may exceed levels observed in major rivers draining into the lake (see Table 5.1, Chapter 5). Because Lake Malawi algal growth may be seasonally limited by both N and P (North et al, 2008), eutrophication might occur with loading of whichever nutrient is limiting phytoplankton at that particular time. Because the fish cages have been deployed in already deteriorating nearshore waters of the southern end of the lake (Puchniak, 2005), the cage wastes rich in both N and P may further eutrophy the littoral zone as well as reduce the diversity of the endemic cichlid species in the lake. As mentioned above, the cichlid species in Lake Malawi have generally been described to be very sensitive to exploitation and habitat degradation, such as through eutrophication, because they are trophic specialists, lay fewer eggs than less specialized fishes, invest heavily in parental care and occur in small population sizes in addition to their stenotopic characteristics (Ribbink, 1994). Seehausen et al (1997) demonstrated that the diversity of cichlid species in the African great lakes also depends on their ability to exercise reproductive/sexual isolation between closely related sympatric species by using species-specific nuptial male colorations. However, the differences in nuptial male colorations between sympatric heterospecifics can be masked and susequently lost in turbid eutrophic waters which may lead to the erosion of gene pools through interbreeding between closely related species (Seehausen et al., 1997). In Lake Malawi, water transparency (Secchi depth) ranges between 12 and 17m in clear offshore waters (Guildford et al., 2003) but is lower, ranging between 2 and 8 m, in the more productive southeast arm of the lake (FAO, 1993).

This chapter provides results of the first empirical study, conducted in 2007, aimed at determining effects of the cage wastes on the surrounding waters and underlying sediments compared to control stations away from the cage farm. Measurements included dissolved and particulate nutrient levels, turbidity, dissolved oxygen, chlorophyll concentration, temperature and Secchi disc depth. The control stations were grouped into downstream and upstream stations based on the main current direction through the cages. Water current velocity and direction below the fish cages were measured using an acoustic current meter at 2.5 m above the lake bottom. Aggregation and feeding activities of wild fish around the fish cages were measured qualitatively by snorkeling as well as using an underwater camera. Consequently, the role of water current velocity through the cages and consumption of cage wastes by wild fishes which aggregated around the cages on the dispersion, sedimentation and concentration of nutrients and particulate material from the fish cages into the surrounding environment was investigated.

3.3 Materials and methods

3.3.1 Study site

The farm site has been presented in Chapter 2. Sampling for this study was done along a 10 km south-north transect through the cage area along the main direction of two dominant winds and resulting water currents. The transect included one station (KGC) at the centre of the cage farm and eight control stations – 4 stations south and another 4 stations north of the cage farm. Table 3.1 provides the sampling station depths and distances from the cage area as well as

their locations (latitudes and longitudes) along the study transect (Fig. 3.1). Control stations in this study were located further away from the fish farm (at least 1 km away) than control stations in many similar studies (e.g., Cornel & Whoriskey, 1993). Distant control stations were necessary in this study because no environmental impact assessment (EIA) study was done prior to the establishment of the cages to provide pre-impact baseline conditions. This study was based on the assumption that, prior to the establishment of the cages, the farm area including the bottom sediments had similar environmental conditions to those currently observed at furthest control stations from the cage area. Therefore, statistically significant variation in any environmental parameters between the cage area and the control stations might be attributable to cage operations.

This physical-chemical study was a component of a larger project involving a number of activities to assess environmental impacts of cage operations to the surrounding area and sediments. As such, some physic-chemical parameters and stations were sampled more frequently than others. The transect stations, including the KGC at the center of the fish farm, were routinely sampled three times a month; twice for water sampling for chlorophyll a and nutrient analyses, and once a month for zooplankton, phytoplankton, sediment and benthic invertebrate samples for stable isotope analyses. Phytoplankton and zooplankton samples were collected by net hauls while sediment and benthic invertebrates (bivalves, snails and worms) samples were collected using a Ponar grab. The same sampling was also done around cages.

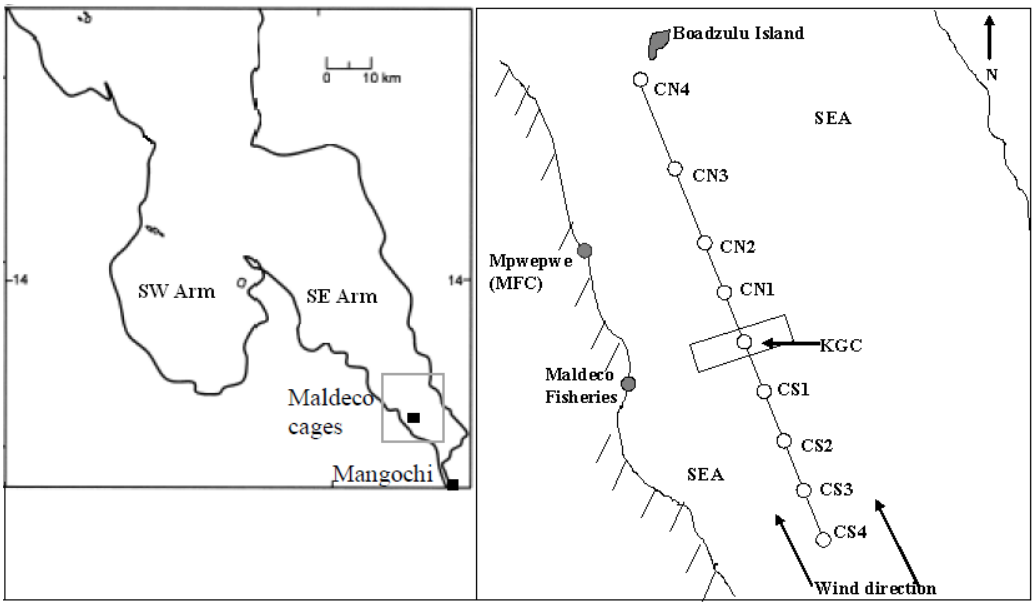


Fig. 3.1. Southern Lake Malawi showing the location of the cage facility at Maldeco and the study transect through the cage fish farm.

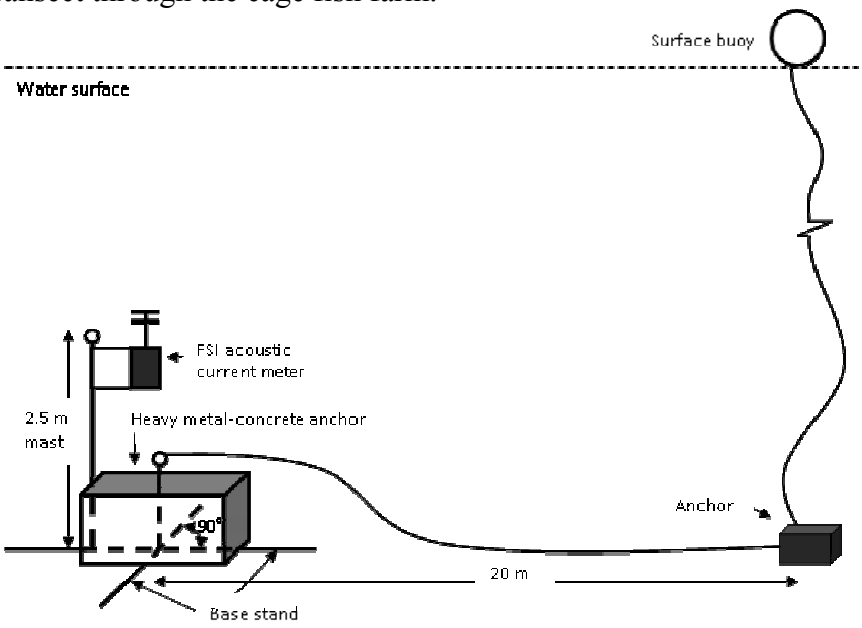


Fig. 3.2. A firm mooring stand used to deploy an FSI acoustic current meter at the fish farm in Lake Malawi. The stand is not drawn to scale. Deployment was done using a 300 HP winch on one of Maldeco Aquaculture boats, also used for harvesting.

3.3.2 Meteorological data

Daily average wind speed, daily maximum air temperature, daily minimum air temperature and rainfall data for the period between August 2006 and March 2008 were gathered from a government operated meteorological station at Mangochi township ($14^{o}28$'S, $035^{o}15$'E) located between Lakes Malawi and Malombe (Figs. 1.1 & 3.1). Mangochi township is located 20 km to the south of the fish farm along the Shire River, which is the lake's outlet at the southern tip of the southeast arm. The meteorological station at Mangochi was the closest to the fish farm. Although the station was located about 1 km inland from the water, the recorded data, except for wind speed due to local sheltering, should represent the general meteorological conditions of the whole southeast arm including the fish farm. Wind direction data were not available. Daily mean air temperature was calculated as the mean of the daily maximum and daily minimum temperatures (i.e., $T_{mean} = [T_{min} + T_{max}]/2$) (Weiss and Hays, 2005).

3.3.3 Water temperature

Water temperature at the cage area was measured at six depths (0, 1, 2, 5, 10 and 13 m) using a chain of HOBO UTBI-001 Tidbit temperature loggers manufactured by Onset Computer Corporation. HOBO temperature loggers are fitted with precision components that eliminate the need for user calibration. The first chain of tidbit temperature loggers was deployed and uploaded three times between 14 March and 05 July 2007. The tidbit chain got lost in stormy weather at the lake during a fourth deployment which started on 06 July 2007. Replacement

loggers were deployed on 9 November and these collected data until the end of the project (29 December, 2007). The Tidbits were left with the cage operators who are still collecting water temperature data at the cage farm for their cage production purposes and for my continued monitoring of the water conditions.

Temperature depth profiles were also collected using a SBE-19 SEACAT (CTD) profiler as described below under the "water column profiling" section.

3.3.4 Water current data

A 2D-ACM Falmouth Scientific Inc (FSI) acoustic water current meter was deployed in bottom waters (2.5m above the bottom) between May and December 2007 to measure current velocity (cm s^{-1}), current direction (degrees) and water temperature (^{o}C) at the cage area. The acoustic current meter was attached to a steady mast as shown in Fig. 3.2. The instrument measures horizontal current velocities (accuracy of ±2% of reading) in all four directions; positive values are recorded in the north and east directions only while south and west are recorded as negative. The instrument also measured the earth's magnetic field using a 3-axis solid state compass, the tilt of the instrument using a 2-axis solid state accelerometer and water temperature using a solid state temperature sensor. Both tilt and magnetic field measurements were used to calculate vector averaged north/south (+/-) and east/west (+/-) components of the current velocities. Average resultant current velocity was calculated as the square root of the sum of the squares of the vector averaged north and east current velocities. Average current direction

(0 to 360^{o} clockwise from true north, accuracy of ±2^{o}) was calculated from the arctangent of the vector averaged north/south (+/-) and east/west (+/-) current velocities. Using the built-in real-time clock, we were able to program the instrument to periodically switch itself on and take data and then shut itself when not taking data. Our "on time" was 1 minute, for all deployments, beginning at the start of specified interval (the Interval Time) which was either 5, 10 or 20 minutes depending on the length of the deployment period sampled. This arrangement helped to conserve battery power and internal flash memory (1MB) of the instrument so that we did not unexpectedly run out of battery power or storage memory during any of our deployments.

3.3.5 Water column profiling

Vertical attenuation of photosynthetically active radiation (PAR, 400 – 700nm) in the water column at each station was measured with a LI-COR (LI-1000) datalogger fitted with a LI-192 flat plate underwater quantum sensor which has a resolution of 0.01 μmol m^{-2} s^{-1}. PAR irradiance was measured above and below the air-water interface and every 1 m thereafter to 15 m water depth. PAR vertical extinction coefficient was calculated as the slope of a linear regression through log transformed PAR values and depth data for each profile made. Secchi depth, which also indicates water transparency for image forming light, was also routinely measured at all stations with a white and black 20 cm diameter Wildco Secchi disk.

Water column vertical profiles of temperature (^{o}C), oxygen (mg L^{-1}), conductivity (mS cm^{-1}) and in situ chlorophyll fluorescence (μg L^{-1}) were measured with a SBE-19 SEACAT (CTD) profiler.

The CTD profiler measured chlorophyll fluorescence using a fitted WETStar fluorometer (model WS3S) from WETlabs (application range of 0.06-150 μg L^{-1} chlorophyll fluorescence and sensitivity of ≥0.03 μg L^{-1}). A pH sensor (SBE 18) on the CTD malfunctioned and therefore the pH data were not used. Also, on some occasion I determined the algal classes, their vertical distribution and their contribution to total chlorophyll fluorescence in the water column using a bbe Moldaenke Fluoroprobe. This was done in February, March and during November and December, 2007.

3.3.6 Water sampling

Discrete water samples for nutrient and chlorophyll analysis were collected from 2 and 10 m depths using 5 L Hydro-Bios Kiel water samplers into clean acid-washed 10 L collapsible sample bottles. The water samples were quickly taken back to the laboratory at the Malawi College of Fisheries (MCF) at Mpwepwe about 5km from the cage farm, where subsamples were filtered (within 6 h) through non- or pre-combusted GF/F filters for various analyses. The filters were stored frozen in aluminum foil or petri dishes for chlorophyll a, total suspended solids (TSS), particulate phosphorus (PP), particulate nitrogen (PN) and particulate carbon (PC) analyses. The filtrates passed through non-combusted GFF filters were stored frozen in clean acid-washed 500 mL sample bottles for ammonium (NH_4^+) (Holmes et al., 1999), nitrate-nitrite (NO_3^- - NO_2^-) and soluble reactive phosphorus (SRP) analyses (Stainton et al., 1977). Particulate carbon (PC) and nitrogen (PN) on pre-combusted GFF filters were analyzed using an Exeter

Analytical Inc. CE-440 Elemental Analyzer while particulate phosphorus (PP) was analyzed using the molybdate-ascorbic acid method after digestion with potassium persulphate (Stainton et al., 1977). All nutrient analyses were done following standard methods at the Aquatic Ecology Group Analytical Laboratory at the University of Waterloo. Chlorophyll pigments on the filters were extracted in 10 mL of methanol-acetone-water solvent (68%, 27% and 5% V/V respectively) in the dark at 4^{o}C for 24 hrs. Extracted chlorophyll pigments and pheopigments were measured with a Turner Designs Series 10 Fluorometer before and after acidification with 1N hydrochloric acid. Chlorophyll a concentrations were later corrected for pheopigment absorption which, on average, accounted for about half of chlorophyll absorption.

3.3.7 Particle sedimentation

Weighted cylindrical sediment traps made of opaque poly-vinyl chloride (PVC) tubing (0.80m long and 0.84m internal diameter; aspect ratio of 9.5) were deployed under the cages and at control stations to determine the effect of the cages on local sedimentation rates of particulate matter (PM), carbon (PC), nitrogen (PN) and phosphorus (PP). Sets of sediment traps were deployed 13 times under the cages and 6 times at control stations between March and June and between October and December, 2007. The length of the deployments was between 4 and 12 day and no preservative was used in the traps. The cages sampled for sedimentation rates differed in their total fish biomass (from 2612 to 23875 kg), mean individual body weights of the caged fish (28 – 261 kg) as well as in the daily feed ration (6.2-21.6 g feed/kg fish/day) that was provided to

the fish during the sampling period. Also, because the cages were able to drift significantly from side to side with winds, base-weighted sediment traps under the cages were actually suspended below the cages by ropes tied to adjacent cages and/or buoys to keep the traps in constant reception of the sedimenting particles as the cages drifted. At control stations, traps were attached to a tight rope stretched between an anchor at the bottom and a buoy just below the water surface.GPS positions of sediment trap deployments were always recorded. Sediment traps at control stations were located at almost the same depth as traps under the cages (~7 m below surface). Out of six trials when sediment trap deployments were made at control stations, only three were successful, because sediments traps used during the other three trials were moved by artisanal fishing vessels that trawled around the study area fishing for "Utaka" (*Copadichromis* spp.). The water samples (~4.5L) in sediment traps were carefully transferred into clean 5L jars and transported in insulated boxes to the laboratory at the Malawi College of Fisheries about 5 km from Maldeco Fisheries docking area. At the laboratory the water samples in the 5 L jars were agitated and subsamples were collected and filtered through pre-weighed, non- and pre-combusted GFF filters for analyses of TSS, PC, PN and PP as described above. Sedimentation rates of PM, PC, PN and PP into the sediment traps were calculated according to White (1990) as mass/m^2 of trap opening/day (i.e., g $m^{-2}d^{-1}$). The flux of particulate material into the sediment traps below the cage bottom was assumed uniform.

3.3.8 Underwater observations

Fish activities and distribution around the fish cages, along a transect from the cages, and at control stations were observed using a Deep Blue Underwater Video Camera (Marine Video Splashcam) and through snorkelling. The Deep Blue underwater camera came with a 30 m long high-strength umbilical cable which allowed the camera to be deployed from a boat. The deployment from a boat, compared to diving with the camera, provided less interference to fish activities which were monitored in real time on a 7" LCD monitor in the boat. Since the camera's umbilical cable was long enough, camera deployments around the cages were also made from the cage platforms. To estimate the distribution of wild fish along a transect from the cages the camera was attached to a 5 m long staff which was firmly fixed to a slowly moving research boat. All videos were recorded on VHS for later viewing and analysis.

3.3.9 Data analysis

Student t-tests (paired and non paired) was used to test the difference at alpha of 0.05 between data due to sampling depth (2 and 10m), distance from the fish farm, season (windy mixing vs calm seasons) and current direction through the fish farm as illustrated and discussed in **Appendix 1: A flow chart of statistical analysis of data** on page 181. Stations to the north and south of the fish farm were, respectively, upstream and downstream of the fish farm during the calm season between January and April and October and December. Because water current direction changed from southerly to northerly during the windy mixing period between May and

September, the north stations became downstream of the fish farm while south stations became upstream. Because all the statistical tests indicated no significant differences ($p>0.05$), data were plotted as means (per given sampling day) for south stations, KGC and north stations. On any given sampling day, data from 2 and 10 m depths and from stations within a location (south, KGC or north) were treated as replicates and a mean was calculated.

3.4 Results

3.4.1 Meteorological data

Air temperature, wind speed and rainfall data gathered from a meteorological station at Mangochi township covered the period between August 2006 and March 2008. The temporal patterns in Fig. 3.3 show that the daily mean air temperatures at Mangochi township (and in the southeast arm) were lower during the austral winter between May and September (minimum temperature was 17.4^{o}C recorded on 03 July 2007) and higher during the rest of the year. Wind speed on the other hand was higher during the austral winter (May to September) and during the dry summer (October to November) and lower during the wet summer between December and April. According to Eccles (1974) the warm and wet season between December and April tends to be dominated by moderate northerly winds while the strong and persistent southerly winds become predominant between May and September. Fig. 3.3 shows similar patterns at Mangochi where high air temperatures between November and April corresponded to low wind speeds while the cool austral winter season between May and September was dominated by high wind

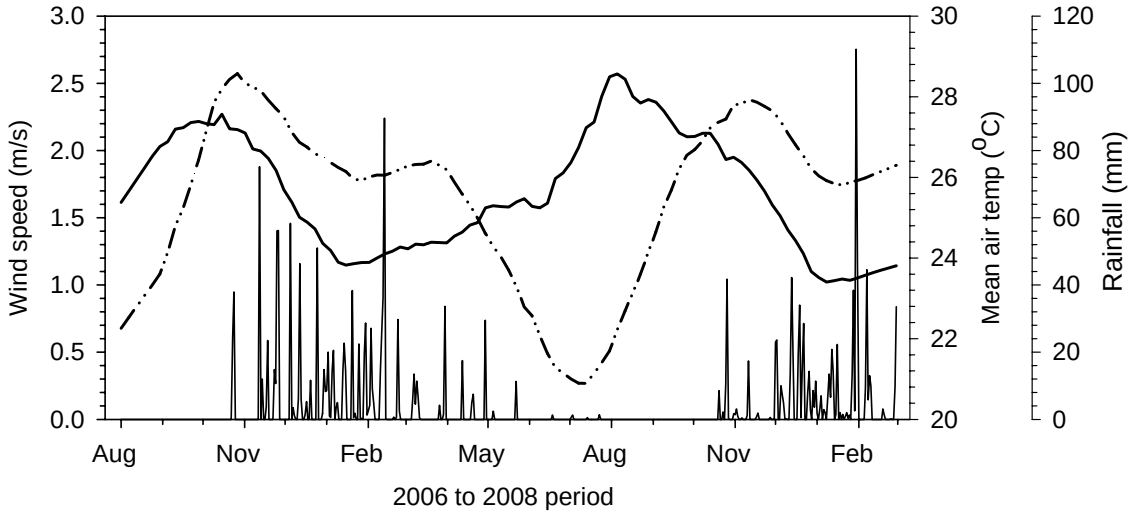


Fig. 3.3. Seasonal patterns in daily average wind speed (m s^{-1}; solid line), mean air temperature (^{o}C; dash-dot-dot line) and rainfall (mm; vertical lines) observed at Mangochi meteorological station between 1st August 2006 and 29th February 2008. The station is situated approximately 1 km inland from the Shire River at the southern tip of Lake Malawi and 20 km from the fish farm. Source: Department of Climate Change and Meteorological Services, Malawi.

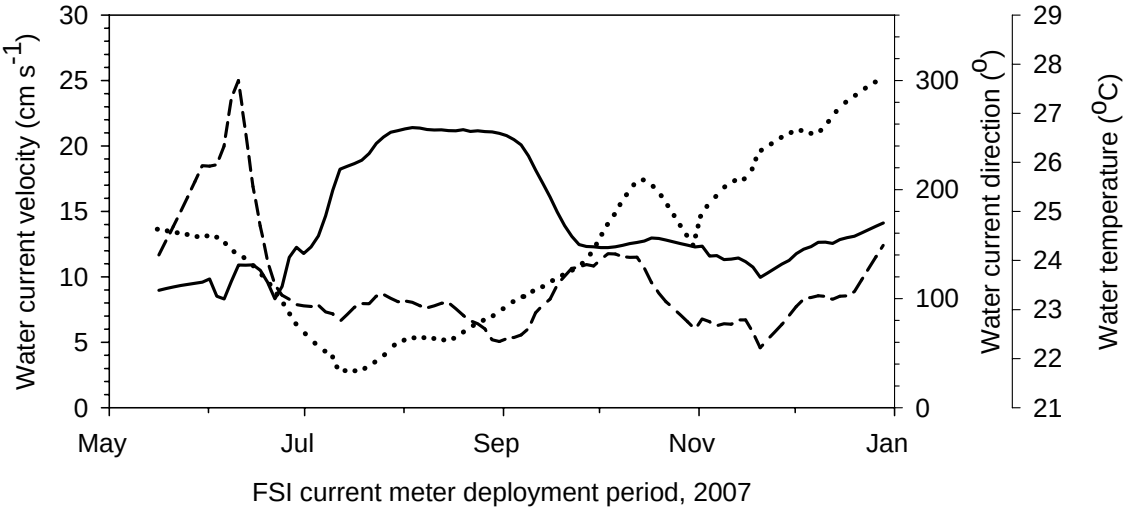


Fig. 3.4. A FSI acoustic water current meter (2D-ACM) data showing smoothed average current velocity (cm s^{-1}) (dashed line) and smoothed current direction (degrees) (solid line) and water temperature (^{o}C) (dotted line) of bottom water at the cage farm in Lake Malawi.

speeds. Rainfall was generally higher at the start of the rainy season between November and December although storm events were also observed in February of both 2006-2007 and 2007-2008 rainy seasons (Fig. 3.3). Total precipitation over the Mangochi-southeast arm area for the 2006-2007 rainy season was 1176.4 mm, slightly lower than the estimated precipitation of 1350 mm/yr over the lake's surface reported by Spigel and Coulter (1996). While air temperature measured at Mangochi township closely estimated air temperature at the cage area, wind speed data probably underestimated wind speed at the cage area due to local sheltering since the met station was located about 1 km inland from the Shire River. To scrutinize the data from Mangochi for local sheltering, I compared average air temperature and wind speed data recorded in 1999 at Mangochi and at Senga Bay beach separated by 80 km of distance over water. The now discontinued Senga Bay met station was run by the SADC/GEF Lake Malawi Biodiversity Conservation Project between 1996 and 2001. The comparison clearly showed that while daily average air temperatures recorded by the two met stations were relatively similar (correlation of 0.90), wind speed measurements were less well correlated (0.51). Wind speed measured on the beach at Senga Bay was, on average, 75% higher than speed measured at Mangochi although seasonal trends in the data sets were congruent (figure not presented). A study by Hamblin et al (2003) reported similar observations with regard to wind parameters over Lake Malawi. They observed that due to local sheltering, a met station at Likoma Island underestimated wind speed compared to another met station on a research vesicle on the lake. The 2007 wind speed data at Mangochi met station were not corrected for sheltering using 1999 correlations because the level of sheltering at Mangochi in 1999 and 2007 might be different as degradation continues in the

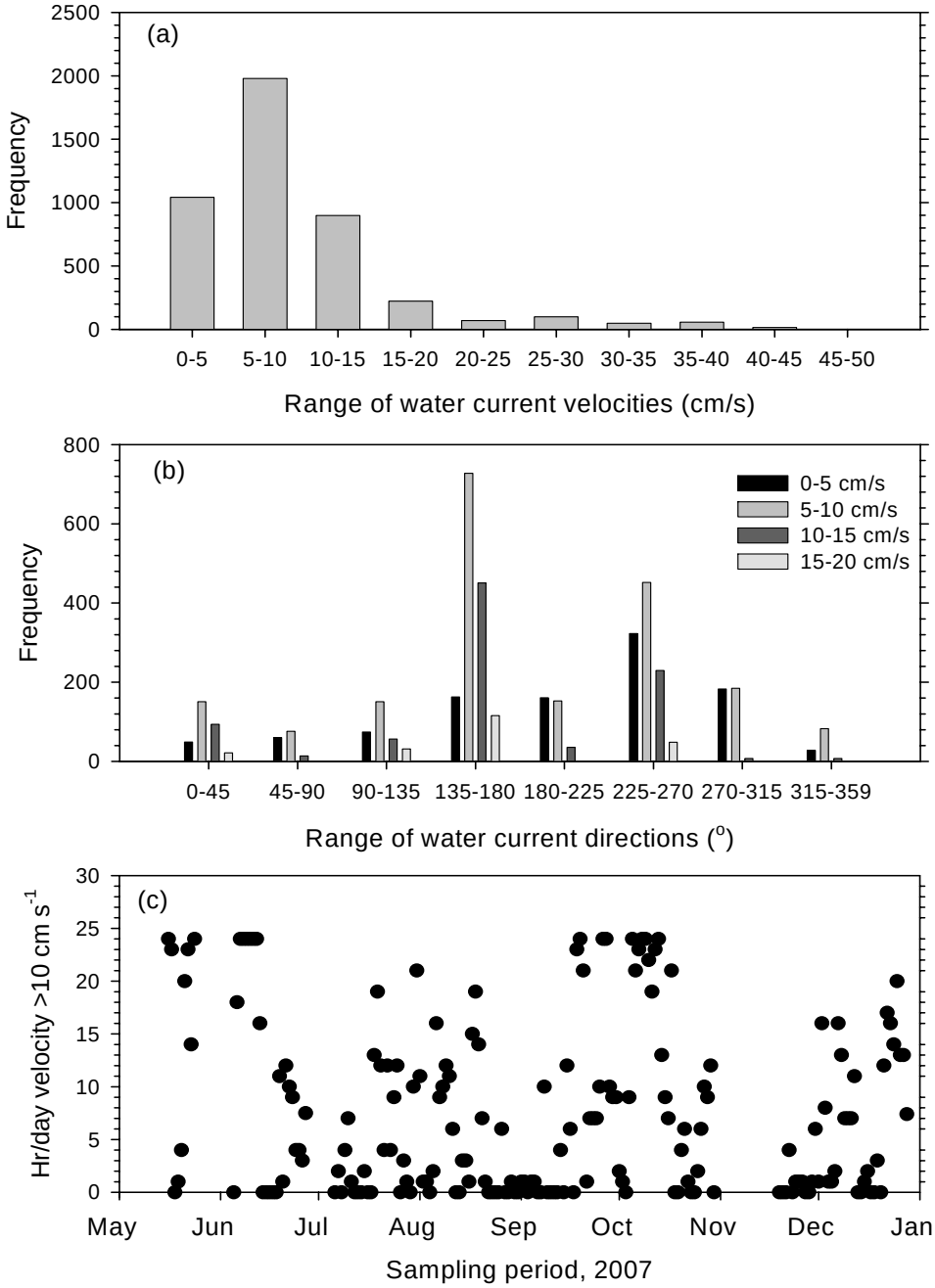


Fig. 3.5. FSI water current meter data measured at the cage area in the southeast arm of Lake Malawi showing frequency of hourly current velocity (cm s^{-1}) (a), frequency of current direction (^{o}) of current velocity ranges (b), and number of hrs/day that had current velocity > 10 cm s^{-1} during the study period (c).

lake's catchment area. The data should therefore be used cautiously because wind speed at the fish farm might be higher than reported at Mangochi.

3.4.2 Water current

Water currents are important in the dispersal of wastes from the fish farm. Currents through fish cages determine the distance and direction cage wastes would spread and finally be deposited. Water current velocity measured 2.5 m off the bottom below the fish farm in Lake Malawi fluctuated between 0.4 and 47.2 cm s^{-1} and the overall mean velocity for the duration of the deployment was 9.3 cm s^{-1} (Fig. 3.4) which is higher than mean current velocities observed in many large lakes of the world (Saylor & Miller, 1987). During the study period, mean current velocity was highest in June and lowest in November. The frequency distribution analysis showed that over 88% of the current records had velocities between 0.4 and 15.0 cm s^{-1}, approximately 50% of observations in the 0.4-15.0 cm s^{-1} range had velocities ranging between 5.0 and 10.0 cm s^{-1} (Fig. 3.5a). Water current direction (Fig. 3.4) in the water below the fish farm cages was consistent and mostly controlled by wind stress on the lake surface. The dominant current direction at the fish farm was southward (mean of 150^{o}) for most of the year. The direction changed to northerly between June and September when strong and persistent southeasterly winds predominated at the lake (Fig. 3.4). These current directions conform to directions of persistent seasonal winds at the lake as described by Eccles (1974). According to Eccles (1974), the period between June and September (when mean current direction was 250^{o})

is dominated by southerly winds while the period between October to April (when mean current direction was 150^{o}) is generally dominated by northerly winds. Fig. 3.5b shows the two most frequently measured water current directions at the cage area, between 135^{o} and 180^{o} and between 225^{o} and 270^{o}. The dominant 5-10 cm s^{-1} speed currents were mostly flowing through the fish farm to the southeast between 135^{o} and 180^{o}. Fig. 3.5c shows the occurrence of persistent of current speeds which were higher than 10 cm s^{-1} over a 24 hr period. Only about 25% of the approximately 200 days of data acquisition with the current meter had 0 h per day of >10.0 cm s^{-1} current speed indicating vigourous currents occurred nearly every day.

3.4.3 Water temperature

Water temperature measured in the surface 5m with a CTD along the 10 km study transect varied between 22.83 and 28.90^{o}C (Fig. 3.6 & 3.7) or > 6^{o}C over the annual cycle. The lowest temperature recorded with a CTD was 22.4^{o}C measured on 25 June below control station CS2, south of the fish farm (Fig. 3.6) when tilted isotherms suggest that upwelling may have been occurring. The temperatures recorded by the continuously deployed benthic current meter were even lower at approximately 22^{o}C in early July (Fig. 3.4). This continuously recording thermistor demonstrates the seasonal range in temperatures near the bottom is also on the order of 6^{o}C as observed for surface waters. While spatial temperature differences along the transect were negligible, Fig. 3.6 & 3.7 show a clear temporal transition

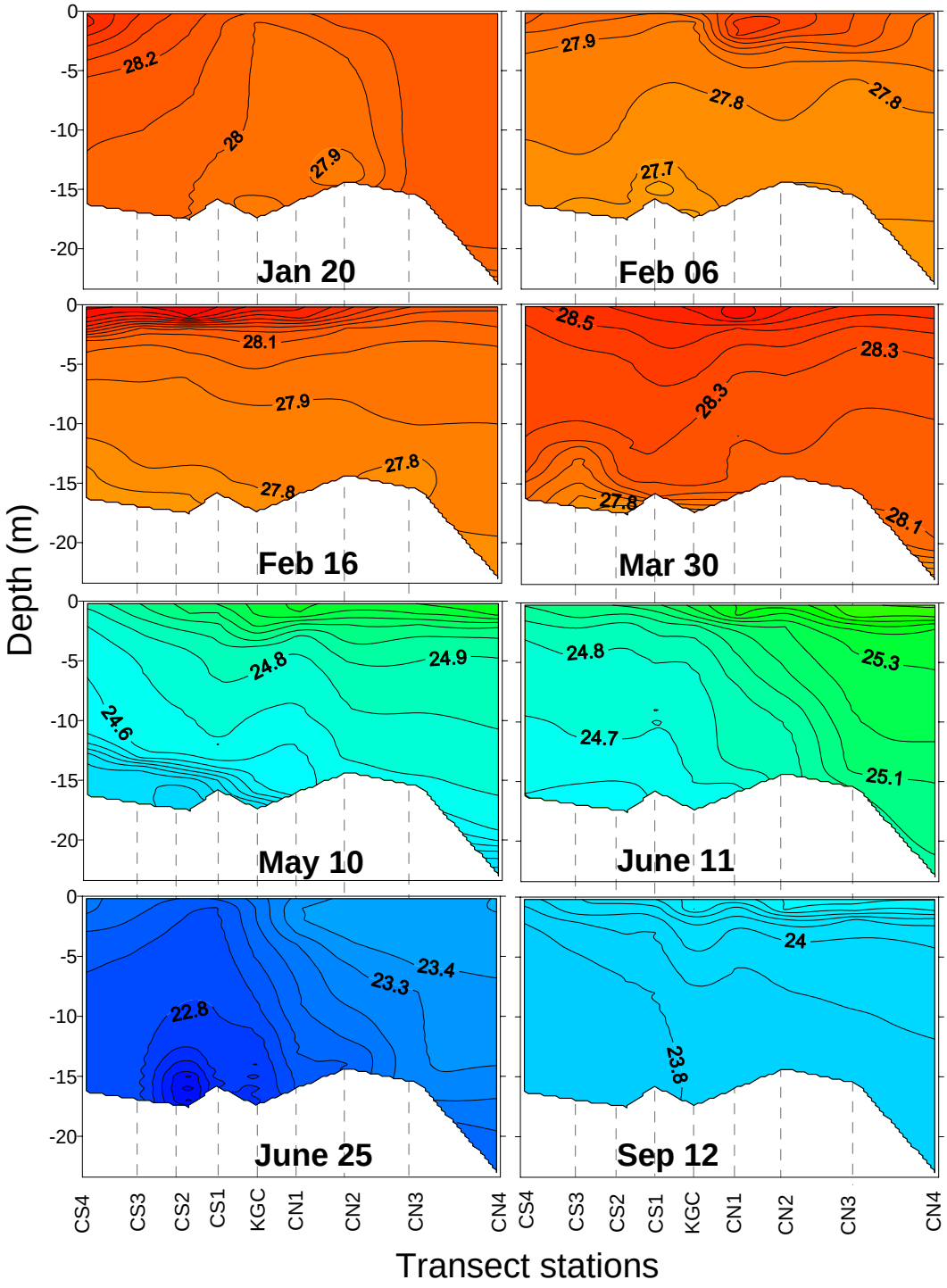


Fig. 3.6. Spatial maps of vertical temperature profiles (^{o}C) along the study transect through the cage area measured with Sea-Bird CTD profiler between January and September 2007. For contour levels see Fig. 3.7.

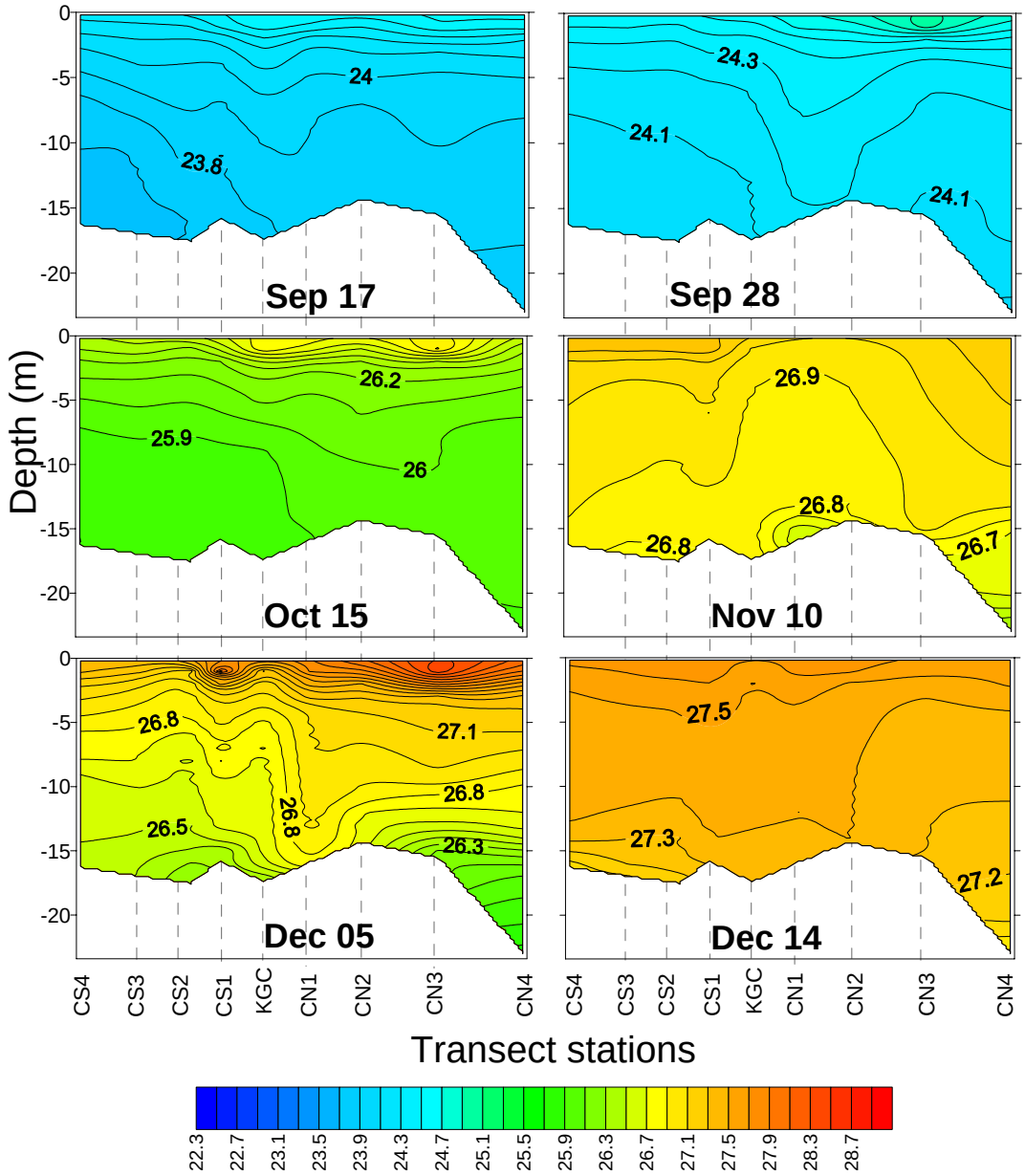


Fig. 3.7. Spatial maps of vertical temperature profiles (^{o}C) along the study transect through the cage area measured with Sea-Bird CTD profiler between September and December 2007.

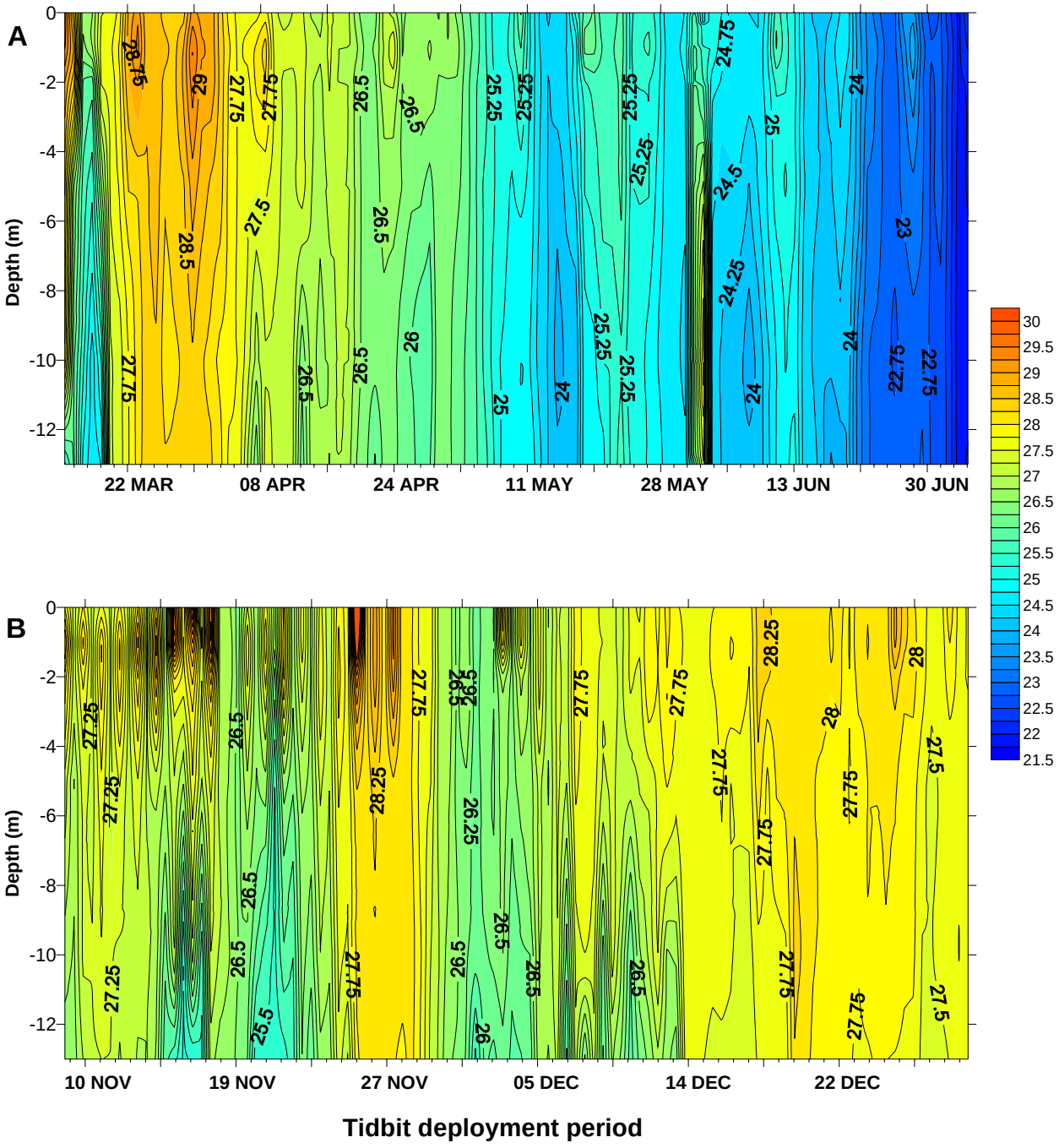


Fig. 3.8. Temporal trend of hourly vertical temperature profiles (^{o}C) measured with a series of HOBO tidbit temperature loggers deployed at 0, 1, 2, 5, 10 and 13 m depths at the fish farm between 14 March and 5 July (A) and between 9 November and 29 December, 2007 (B).

from warm rainy weather conditions between December and April to cool and windy mixing conditions between May and September and then back to slightly warmer post-mixing period between October and December 2007. Vertical isotherms observed during most of the sampling days show that the water mixes reasonably well (very weak thermal stratification) vertically although near-surface diel stratification was frequently observed. These were probably temporary due to diurnal heating. Also, the occurrence of relatively strong tilt to the isotherms near or at the cages in Fig. 3.6 & 3.7 suggests that the fish cages may be deflecting currents up or down (e.g., June 25, Nov 10 and Dec 5) as water moves through the cages or that larger scale internal vertical water motions are affecting the cage environment which would also increase efficacy of mixing.

While CTD casts were repeated after several days, HOBO UTBI-001 tidbit temperature loggers continuously measured water temperature (^{o}C) at 5-20 minute intervals at 0, 1, 2, 5, 10 and 13 m depths at the fish farm (Figs. 3.8A & B). There were no data collected between July and October because we lost the tidbit loggers in July when a stationary boat, which housed security officers on duty at the cages, drifted and got tangled in the tidbit chain during stormy weather at the lake. The lowest water temperature recorded with tidbit temperature loggers was 21.5^{o}C which coincided with the lowest air temperature of 17.4^{o}C recorded at Mangochi on 03 July 2007 and with a minimum in bottom temperatures logged by the current meter (Fig. 3.4). Fig. 3.8A shows a clear decline in temperature between the end of a warm rainy season (March) and the peak of a windy mixing season (July) on Lake Malawi. The figure also shows loss of stratification in the water column as weather changed from warm and rainy to dry and windy.

Vertical isotherms, which indicate strong mixing, appeared at the start of the windy season in May and remained prevalent until July. As a result, water current velocity measured at 2.5 m off the bottom below the fish farm may be similar to currents over the whole water column during the mixing period (Fig. 3.4). Fig. 3.8B shows strong, possibly diurnal, thermal stratification in November and December particularly in the surface 5 m depth of water. Hamblin (1999) observed similar strong stratification in the water column in November and December in the southern part of the lake. In the current study, the stratification broke down between 14 and 29th December probably because of the input of cooler rain water. Rains are generally heavier and more frequent during the beginning of the wet season (Fig. 3.3). During the study period a number of upwelling events were observed at the cage area (Fig 3.8) which confirms previous observations (Hamblin et al., 1999) that upwelling events in the southern part of Lake Malawi appear to occur through much of the year. These upwelling events lasted 1 to 8 days at a time. Daily water temperatures, especially surface temperatures, were lowest around 04:00-05:00 hours and highest around 14:00 hours (figure not presented). Between March and July, the water column was almost isothermal between 0400 and 0500 hrs when the mean temperature difference between surface and bottom waters was 0.25±0.53^{o}C. The temperature difference rose to 0.46±0.57^{o}C between November and December 2007. Highest water column temperatures were observed around 1400 hrs at which time the change in temperature between surface and bottom waters was 0.96±0.74^{o}C between March and July and 1.35±1.07^{o}C between November and December, 2007.

3.4.4 Oxygen levels

Dissolved oxygen has been used as the primary indicator of localized pollution of cage culture operations in many water bodies. Dissolved oxygen concentration in the surface 5m along the study transect, measured by an oxygen probe attached to a SBE-19 SEACAT (CTD) profiler, varied between 4.35 and 7.68 mg L^{-1} and % saturation varied between 65.9 and 96.4%. Dissolved oxygen concentrations in the water column were lower during the stratified period between January and early May, and higher in the windy-mixing season between May and September (Fig. 3.9 & 3.10). Patches of low dissolved oxygen concentration were observed in bottom waters mainly during the two stratified periods, January to early May and November to December, 2007. These dissolved oxygen concentrations in bottom waters were lowest (3 mg L^{-1} in 23^{o}C waters) at stations south of the fish farm on May 10, just immediately before the onset of the wind mixing period. The period between June and September, 2007 showed similar or marginally changing oxygen levels from the lake surface to the bottom. One of the impacts of organic particle sedimentation below fish cages is that it causes deoxygenation of the sediment-water interface as the organic matter decomposes. While most of the low oxygen zones observed in this study (e.g., 10th May 2007) were possibly due to large scale water movements within the lake, some low oxygen zones (e.g., 20th January and 30th March 2007) may be attributed to cage culture operations. The thermal structure on May 10 suggests upwelling of cooler and less oxygenated deeper water could have occurred while the thermal structure on 30th March suggests strong stratification, particularly at depth, which might have reduced oxygen exchange with the water layer immediately above it (Fig. 3.6).

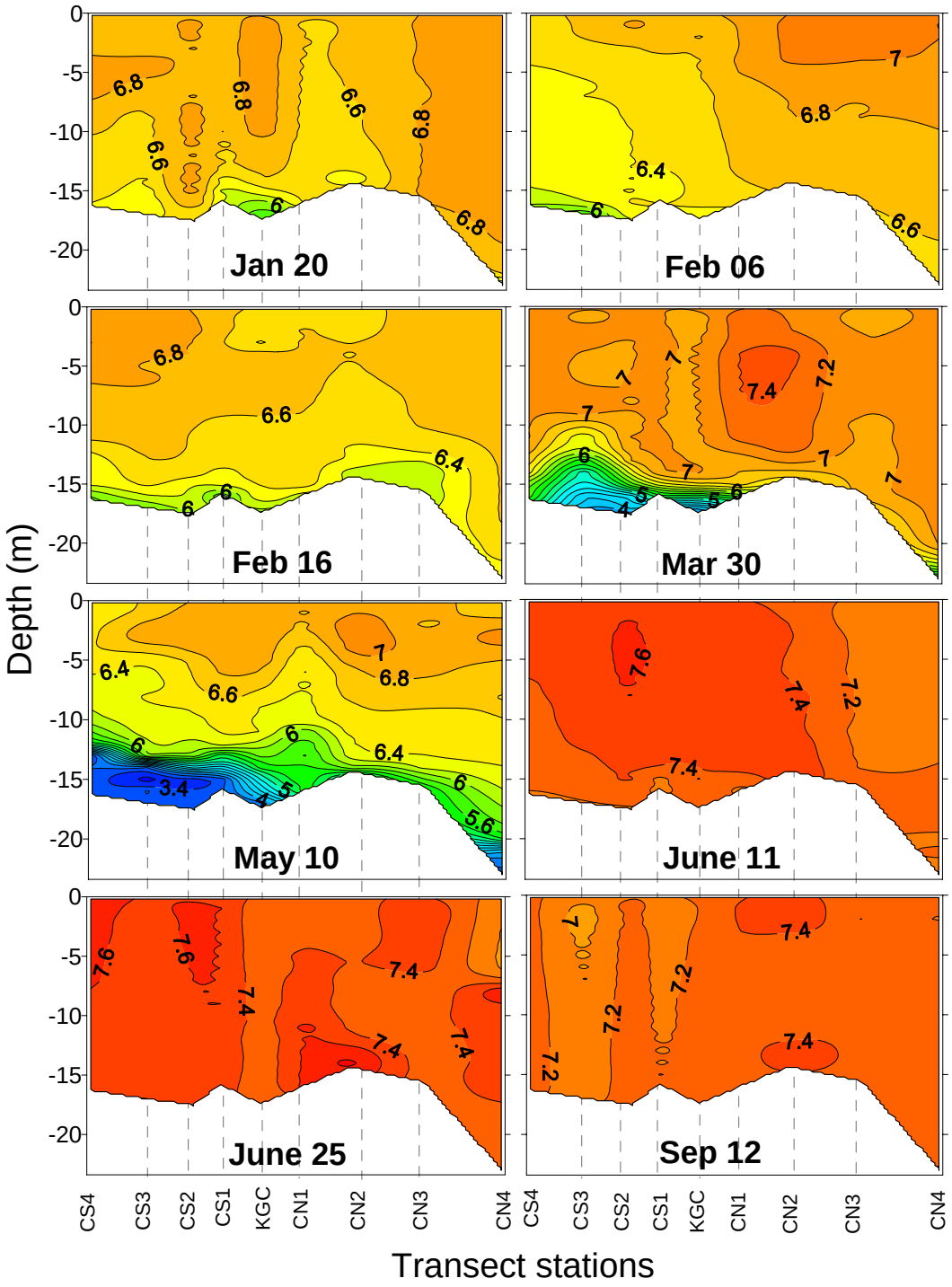


Fig. 3.9. Spatial distribution of vertical oxygen profiles (mg L^{-1}) along the study transect through the cage area measured with an oxygen probe attached to a Sea-Bird CTD profiler between January and September 2007. For contour levels see Fig. 3.10.

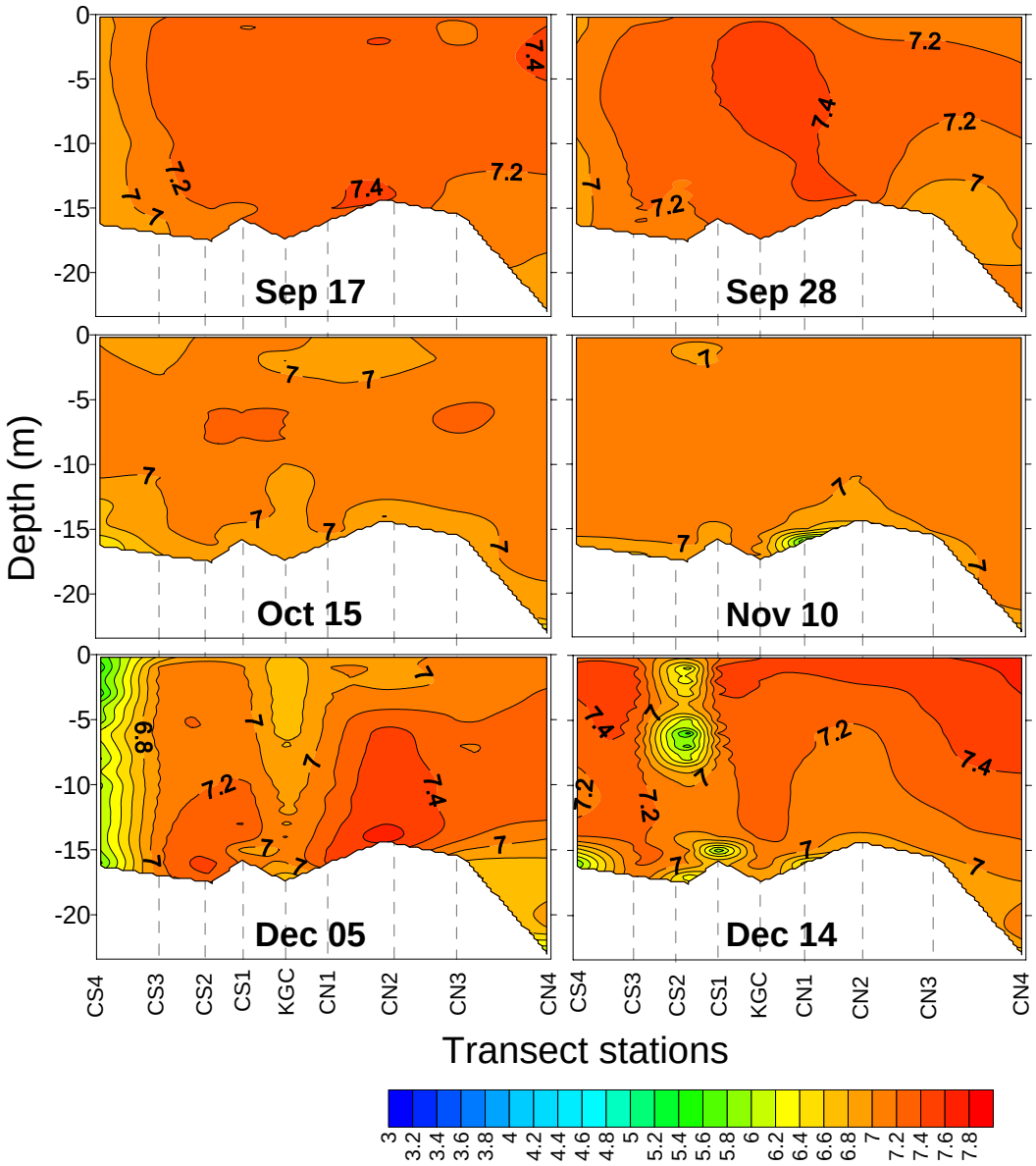


Fig. 3.10. Spatial maps of vertical oxygen profiles (mg L^{-1}) along the study transect through the cage area measured with an oxygen probe attached to a Sea-Bird CTD profiler between September and December 2007.

3.4.5 TSS and water transparency indicators

TSS values (Fig. 3.11a; Table 3.1) ranged between 0.35 and 3.70 mg L^{-1} at north stations, 0.05 and 2.05 mg L^{-1} at the KGC station and between 0.05 and 4.60 mg L^{-1} at south stations. The mean TSS concentration at the KGC station (1.67±1.17 mg L^{-1}) was higher than the means at the north and south stations which were 1.38±0.86 and 1.22±0.53 mg L^{-1} respectively. However, Student t-test indicated there were no significant differences in TSS between stations along the study transect ($p>0.05$). The high variability in the data at the cage station (KGC) could be due to sampling time which could be before or after feeding. TSS was more strongly correlated to both Secchi depth ($r^2=0.60$) and PAR extinction coefficient ($r^2=0.78$) values at KGC than at the other stations. No clear temporal patterns were observed in TSS data although levels tended to decline between September and December before rising again at the end of December probably due to tributary inputs. Secchi depths fluctuated between 2.9 and 9.0 m at the KGC station, 2.9-9.6 m at north station and 2.5-10.1 m at the south stations (Fig. 3.11b, Table 3.1). Secchi depth at the south stations tended to be slightly higher during the wet season (December-March) than during the dry season (April-October) while at the north stations Secchi depth tended to be higher during dry than wet season. At the KGC station, Secchi depth was similar during the two seasons. Wind induced mixing at the lake, however, did not affect Secchi depth at any station along the study transect. Mean Secchi depth was lowest at the fish farm (5.7±1.6 m) but increased in both south (6.1±1.9 m) as well as north (5.9±1.8 m) directions along the study transect.

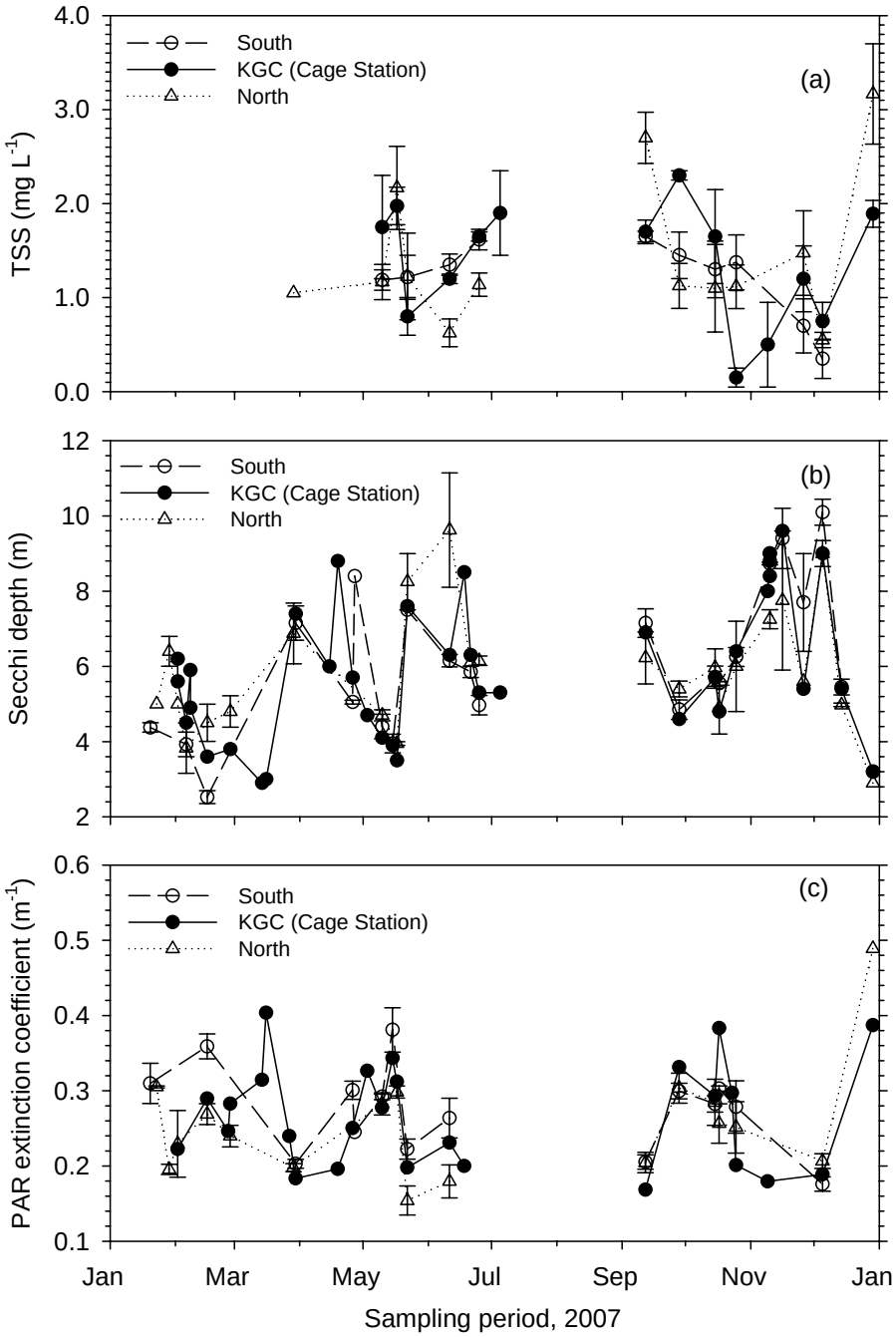


Fig. 3.11. Temporal trends of total suspended solids (TSS) (a), Secchi depth (b) and PAR extinction coefficient (c) data at south control stations, KGC station and north control stations along the study transect. The data markers represent means of pooled data per sampling day because no significant differences were observed with depth at each station and between stations.

PAR extinction coefficient varied between 0.14 and 0.49 m^{-1} at the north stations, 0.17-0.40 m^{-1} at the KGC station and 0.17-0.41 m^{-1} at the south stations (Fig. 3.11c, Table 3.1). Mean PAR extinction coefficient increased from north (0.25±0.05 m^{-1}) to KGC (0.27±0.07 m^{-1}) to south (0.28±0.06 m^{-1}) stations respectively. Similar PAR extinction coefficient values for the cage area were reported by Gondwe et al (2008) who sampled an inshore-offshore transect ~2 km from the cage farm in 2002. Mean PAR extinction coefficient at the southern stations was significantly higher than the mean at the northern stations ($p<0.02$) but both means were not significantly different from that at the cage station (KGC). With respect to individual stations along the study transect, extinction coefficient increased from north to south but only the furthest stations on both sides of the transect (CN4 and CS4) had significantly ($p<0.03$) different mean coefficients from that at the cage station (KGC). No clear seasonal patterns were observed in the data (Fig. 3.11c).

3.4.6 Chemistry data

NH_4^+ concentrations (Fig. 3.12a) in the water column along the study transect ranged between 4.43 and 12.1 μg L^{-1} at the KGC station, 5.5 and 13.1 μg L^{-1} at the south stations and between 4.1 and 16.7 μg L^{-1} at the north stations. The mean NH_4^+ concentrations were 8.0 μg L^{-1} at the KGC station, 8.3 μg L^{-1} at the south stations and 8.1 μg L^{-1} at the north stations. Student t-test showed that there were no significant differences in mean NH_4^+

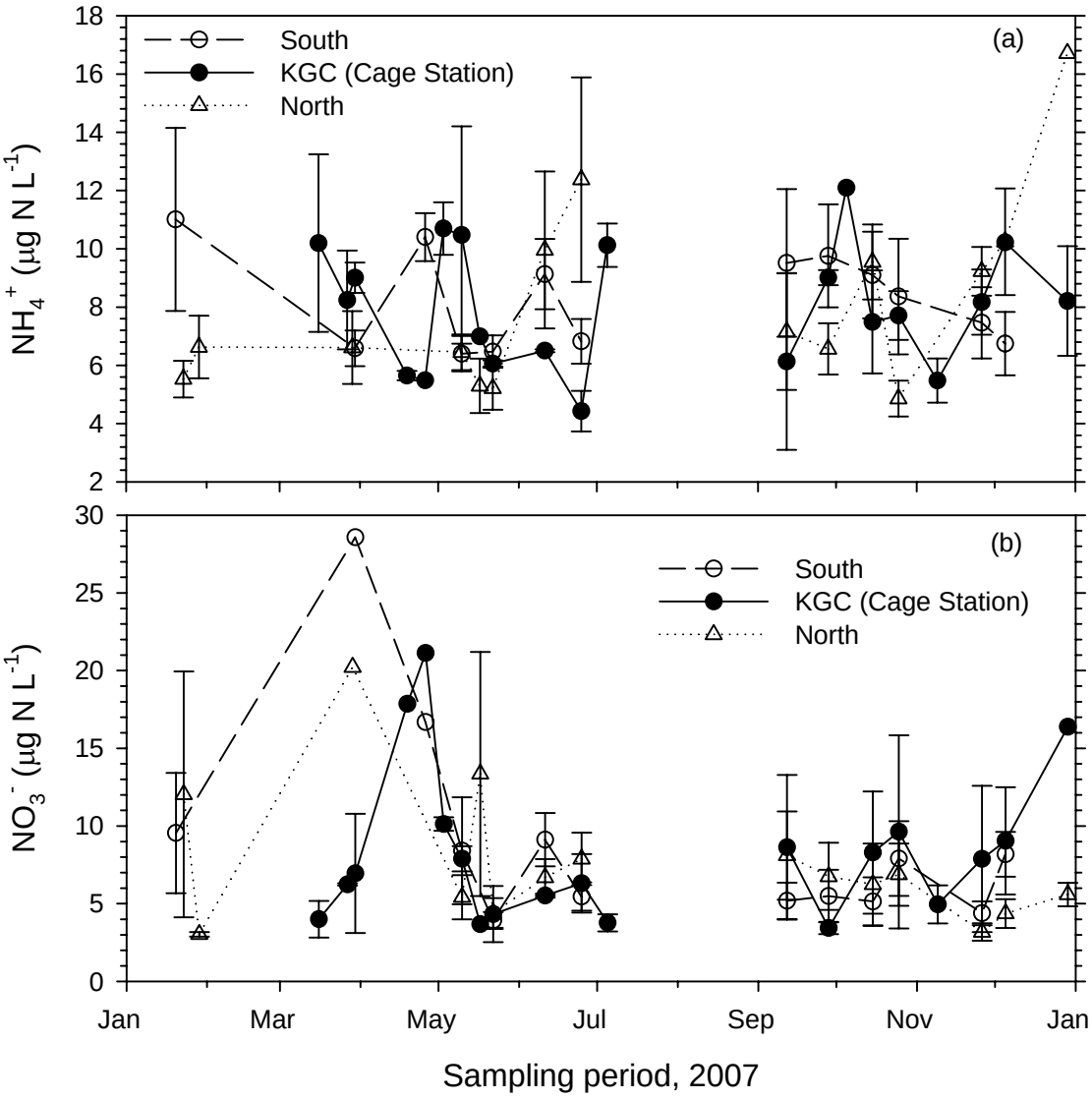


Fig. 3.12. Temporal trends for ammonium (NH_4^+, µg N L^{-1}) (a) and nitrate (NO_3^-, µg N L^{-1}) (b) at south control stations, KGC station and north control stations along the study transect. The data markers represent means of pooled data per sampling day because no significant differences were observed with depth at each station and between stations.

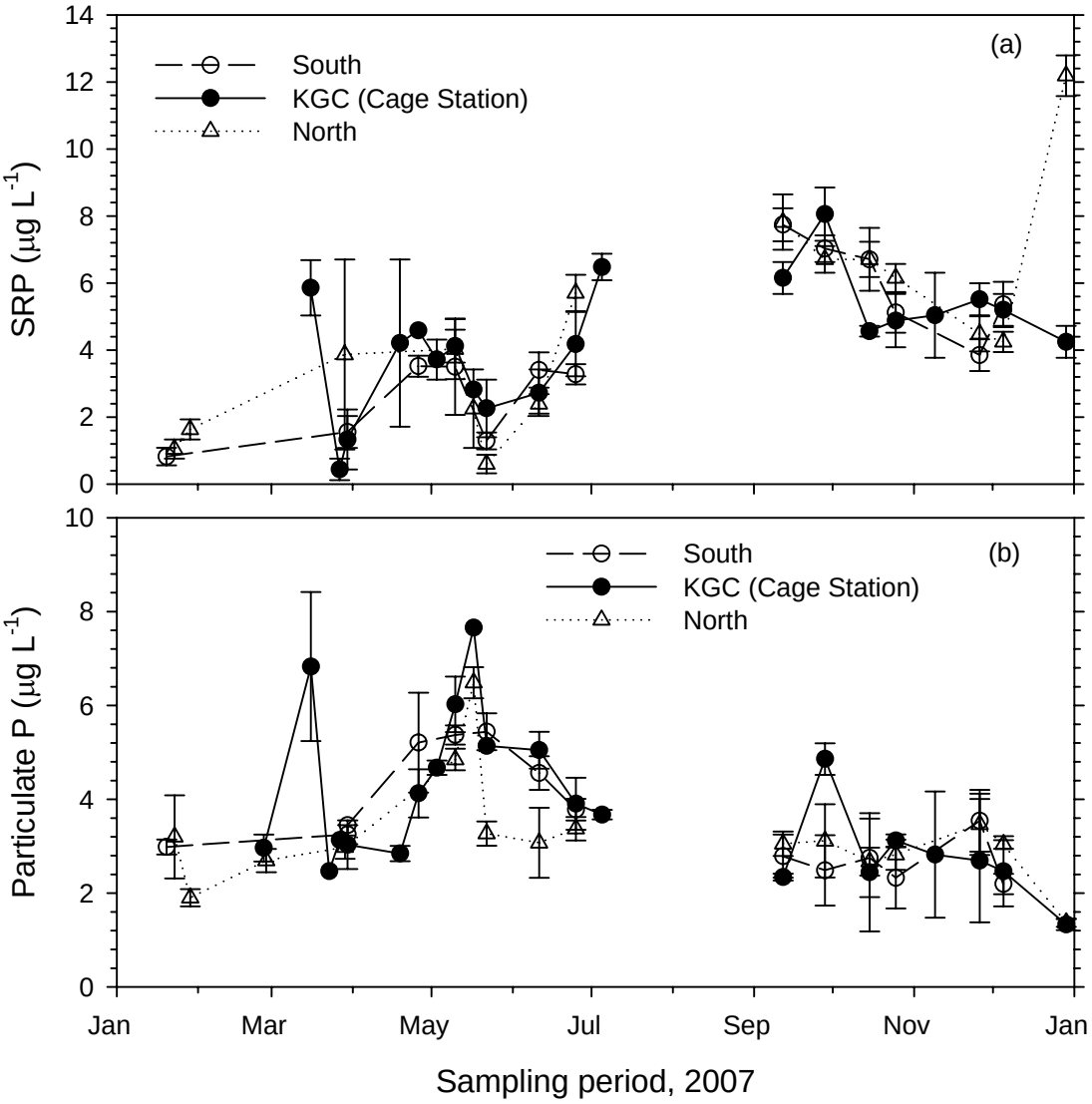


Fig. 3.13. Temporal trends for soluble reactive phosphorus (SRP, µg L^{-1}) (a) and particulate phosphorus (PP, µg L^{-1}) (b) at south control stations, KGC station, and north control stations along the study transect. The data markers represent means of pooled data per sampling day because no significant differences were observed with depth at each station and between stations.

concentrations between the sampled stations (KGC and 4 control stations) along the study transect ($p>0.05$). No clear temporal patterns were observed in the data (Fig. 3.12a).

Nitrate (NO_3^-) concentrations (Fig. 3.12b) in the water column along the study transect ranged between 3.4 and 21.1 μg N L^{-1} at the KGC station, 3.1 and 28.6 μg N L^{-1} at the south stations and between 2.5 and 21.2 μg L^{-1} at the north stations. Although Student t-test showed that there were no significant differences in mean NO_3^- concentrations between the sampled stations ($p>0.05$), the mean NO_3^- concentration at the KGC station tended to be slightly higher than mean concentrations at the south and north stations along the study transect, particularly during the stratified period (Fig. 3.12b). No clear temporal patterns were observed in the data.

Soluble reactive phosphorus (SRP) concentrations (Fig. 3.13a) ranged between 0.4 and 8.1 μg L^{-1} at the KGC station, 0.4 and 8.1 μg L^{-1} at the south stations and between 0.3 and 12.2 μg L^{-1} at the north stations. The mean SRP concentrations at these stations were 4.32 μg L^{-1} at the KGC station, 4.2 μg L^{-1} at the south stations and 4.7 μg L^{-1} at the north stations. Student t-test analysis showed that there were no significant differences in mean SRP concentrations between stations (KGC and 4 control stations) along the study transect ($p>0.05$). SRP in the water column tended to increase during the mixing season but gradually declined during the calm season between September and December 2007 (Fig. 3.13a).

Particulate phosphorus (PP) concentrations (Fig. 3.13b) in the water column ranged between 1.3 and 7.7 μg L^{-1} at the KGC station, 1.45 and 6.3 μg L^{-1} at the south stations and between 1.4 and 7.0 μg L^{-1} at the north stations. The mean PP concentration at the KGC

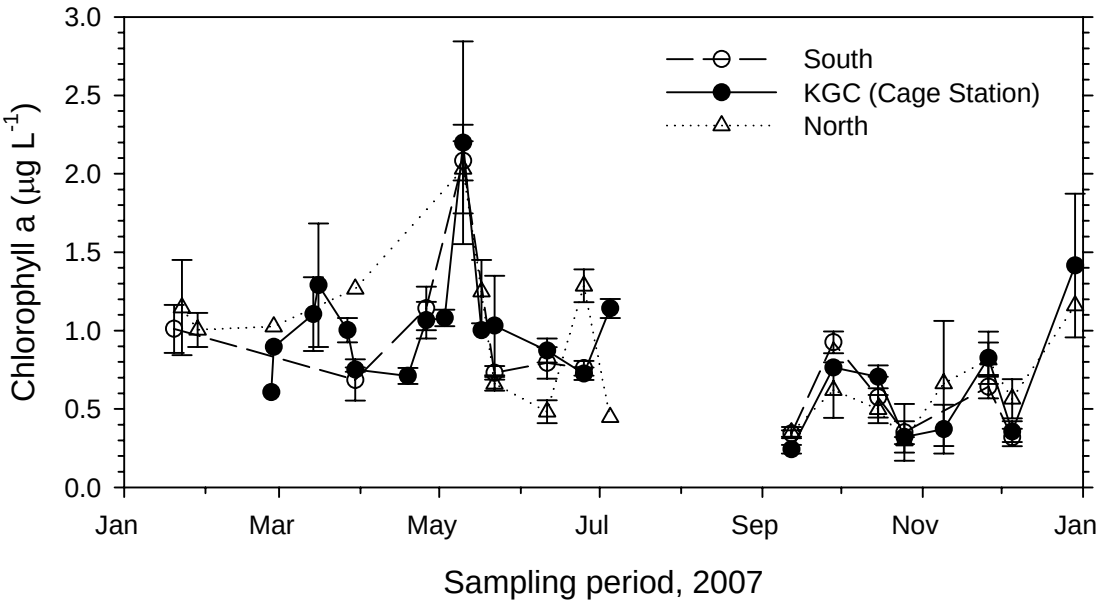


Fig. 3.14. Temporal patterns of chlorophyll concentrations (µg L^{-1}) in the water column at the south control stations, cage station (KGC) and at north control stations along the study transect. The data markers represent means of pooled data per sampling day because no significant differences were observed with depth at each station and between stations.

station (3.8 μg L^{-1}) was not significantly higher than the mean PP concentrations at the south (3.6 μg L^{-1}) and north (3.4 μg L^{-1}) stations. Student t-test analysis showed that there were no significant differences in mean PP concentrations between stations (KGC and 4 control stations) along the study transect ($p>0.05$). PP was higher during the mixing season but was relatively stable and similar along the study transect during the calm season between September and December 2007 (Fig. 3.13b).

Due to technical problems with the Exeter Analytical Inc. CE-440 Elemental Analyzer , only samples collected between January and July 2007 were able to be analyzed for particulate carbon (PC) and nitrogen (PN). PC varied between 192.3 and 496.5 μg L^{-1} (mean of 291.1 μg L^{-1}) at south stations (n=24), between 184.9 and 427.9 μg L^{-1} (mean of 279.1 μg L^{-1}) at the cage station (KGC, n=21) and between 156.1 and 505.4 μg L^{-1} (mean of 294.4 μg L^{-1}) at the north stations (n=22). PN varied between 24.2 and 72.7 μg L^{-1} (mean of 38.4 μg L^{-1}) at south stations, between 26.9 and 47.5 μg L^{-1} (mean of 37.4 μg L^{-1}) at the cage station (KGC) and between 22.4 and 65.9 μg L^{-1} (mean of 40.3 μg L^{-1}) at the north stations (Table 3.2). As with NH_4^+, NO_3^-, SRP and PP concentrations, both PC and PN concentrations at the cage station (KGC) were not significantly different from concentrations at south and north control stations ($p>0.05$). Table 3.3 presents molar ratios of particulate C to particulate N (C:N), particulate C to particulate P (C:P) and particulate N to particulate P (N:P) in water samples along the study transect through fish cages, in feed and in sedimenting particles collected in sediment traps deployed under the cages and at control stations. These ratios represent the calm and rainy season between January and April and the windy mixing season between May and July 2007.

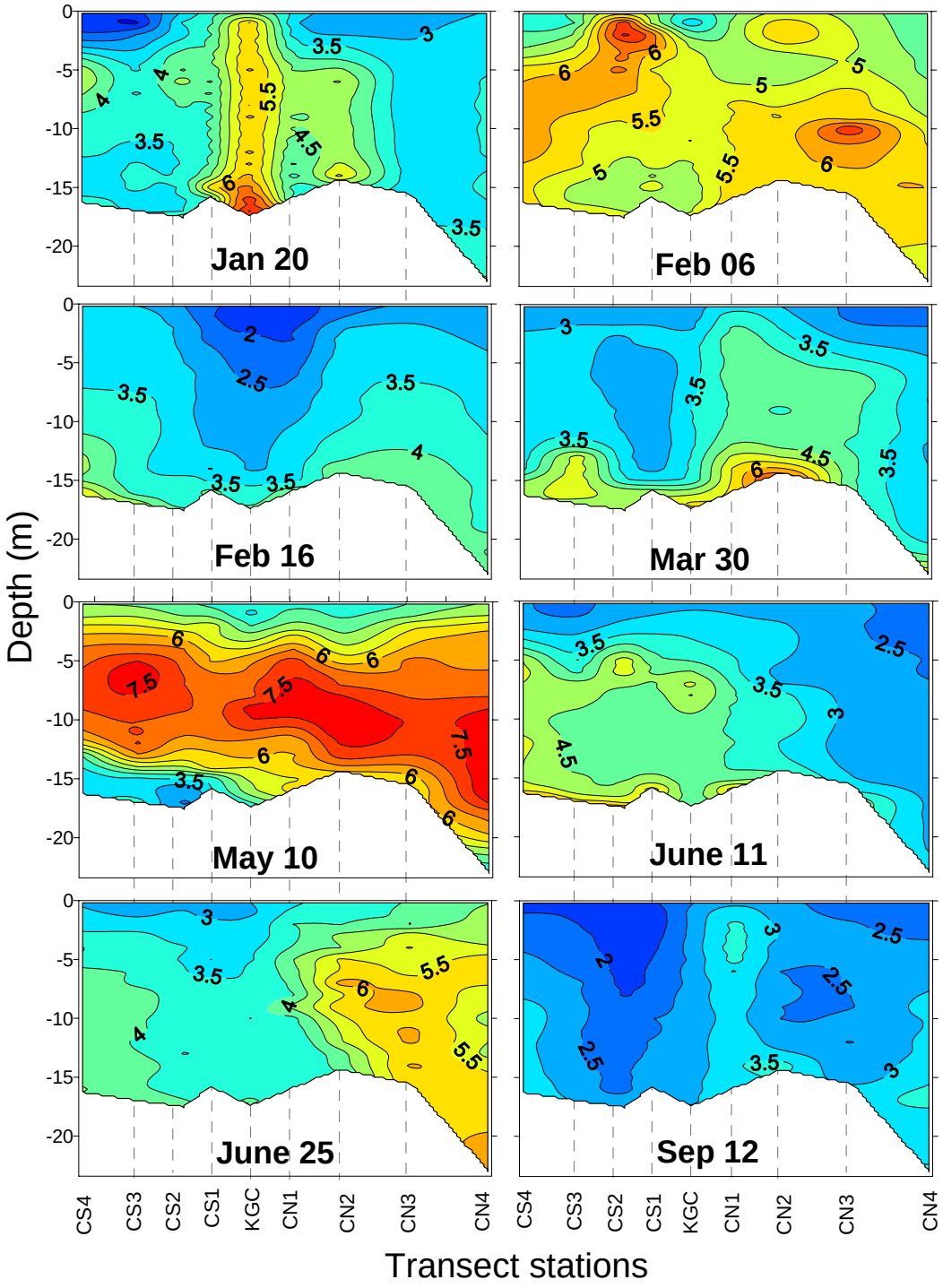


Fig. 3.15. Spatial maps of *in vivo* vertical chlorophyll fluorescence profiles (µg L^{-1}) along the study transect through the cage area measured with a Wet-Star fluorometer attached to Sea-Bird CTD profiler between September and December 2007. For contour levels see Fig. 3.16.

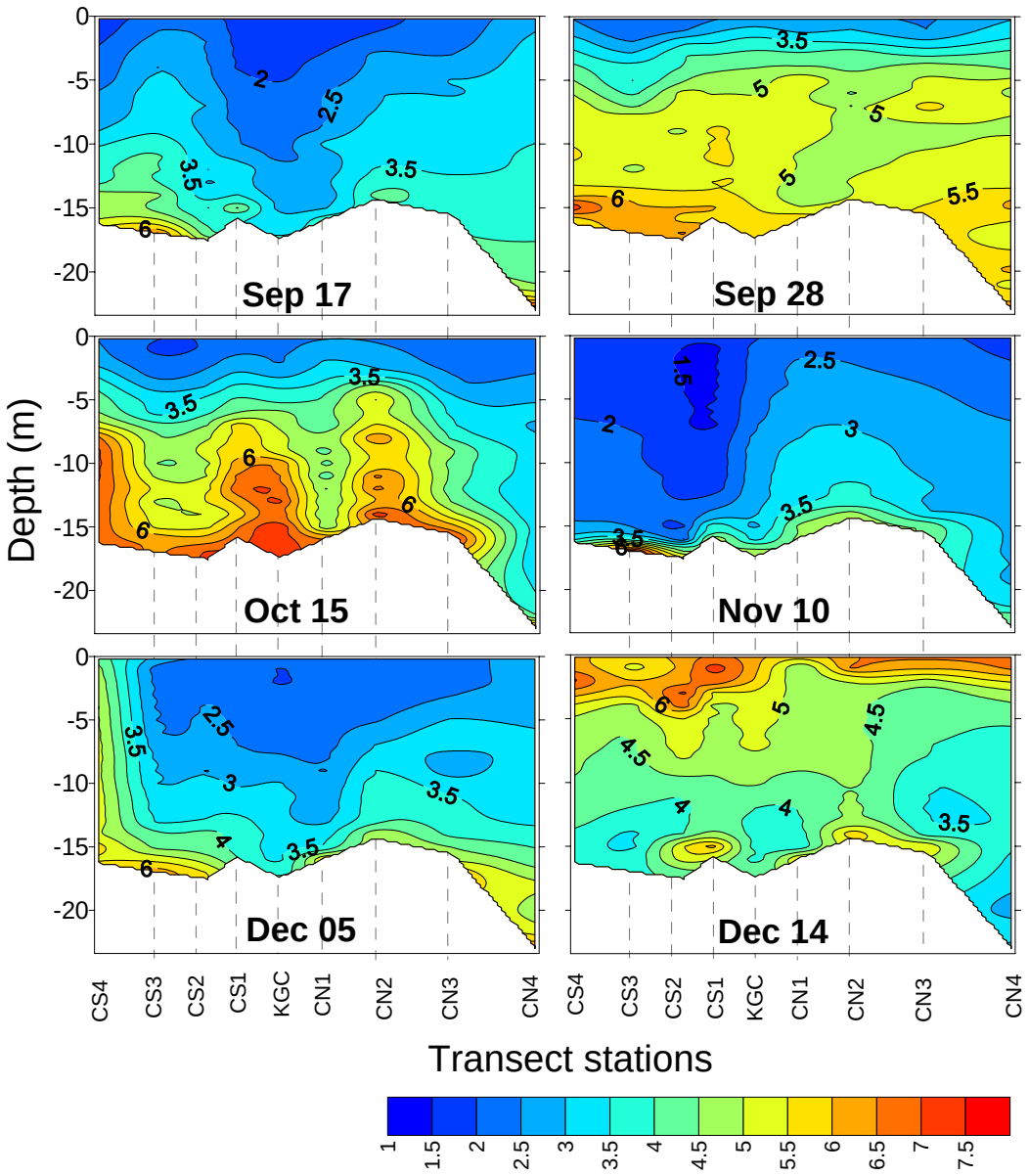


Fig. 3.16. Spatial maps of *in vivo* vertical chlorophyll fluorescence profiles (µg L^{-1}) along the study transect through the cage area measured with a Wet-Star fluorometer attached to Sea-Bird CTD profiler between September and December 2007.

The C:N, C:P and N:P ratios generally decline from north stations, through the cages to the south stations (Table 3.3) although water current direction changed from southerly between January and April to northerly between May and July. According to Guildford and Hecky (2000; *Table 1*), these ratios show moderate deficiency for both N and P in the water column along the study transect. In contrast, the C:N and N:P ratios for feed show, respectively, that the feed is moderately deficient in N but high in P which agrees with P losses estimated using the mass balance model in Chapter 2. While the C:N ratio in cage wastes collected in sediment traps remains relatively similar to the ratio in feed, C:P and N:P in cage wastes are much higher than in feed (Table 3.3). P in cage wastes seems to be quickly lost into the environment as a result the cage particulate wastes become moderately to severely P deficient (Table 3.3).

3.4.7 Particle sedimentation

As mentioned above, sedimentation rates of PM, PC, PN and PP below the cages and at control stations were estimated using sediment traps according to White (1990) as mass m^{-2} of trap opening per day (i.e., g $m^{-2}d^{-1}$). The flux of particulate material into the sediment traps below the cage bottom was assumed uniform for the area of the cages.

The sedimentation rates for particulate material (PM), carbon (PC), nitrogen (PN) and phosphorus (PP) were significantly higher ($p<0.005$) under the cages than at control stations (Fig. 3.19; Table 3.2). The sedimentation rates under the cages ranged between 11-105g $m^{-2}d^{-1}$ for PM, 2-31 g m^{-2} d^{-1} for PC, 0.2-3.1 g m^{-2} d^{-1} for PN and 0.02-0.09 g m^{-2} d^{-1} for PP while at

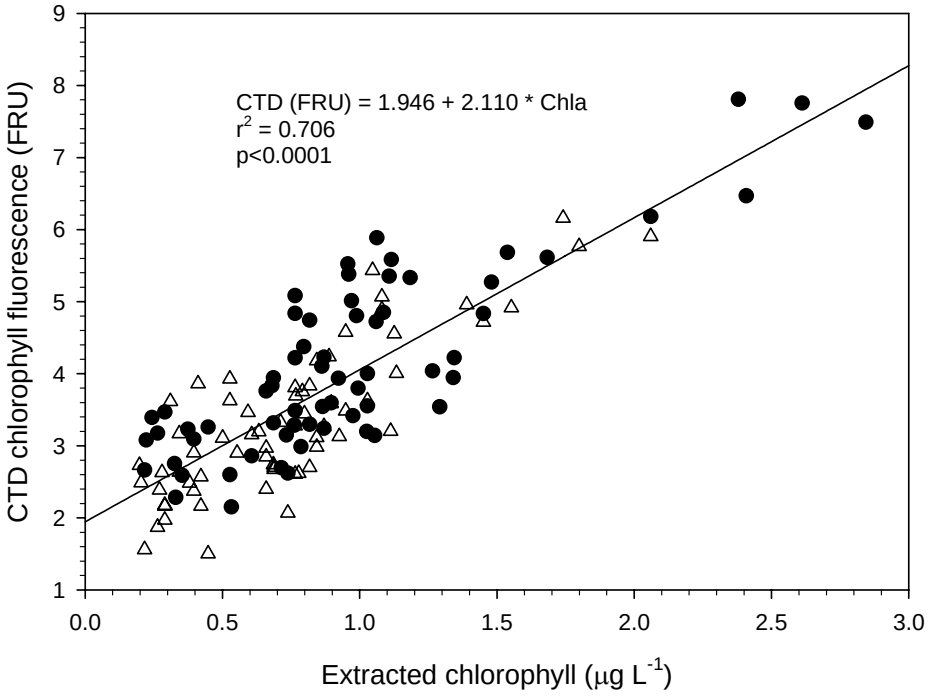


Fig. 3.17. Relationship between extracted chlorophyll concentrations (μg L^{-1}) and CTD fluorescence values (FRU) at 2 m depth (open triangles, n = 66) and 10 m depth (closed circles, n = 65) along the study transect through fish cages in the southeast arm of Lake Malawi.

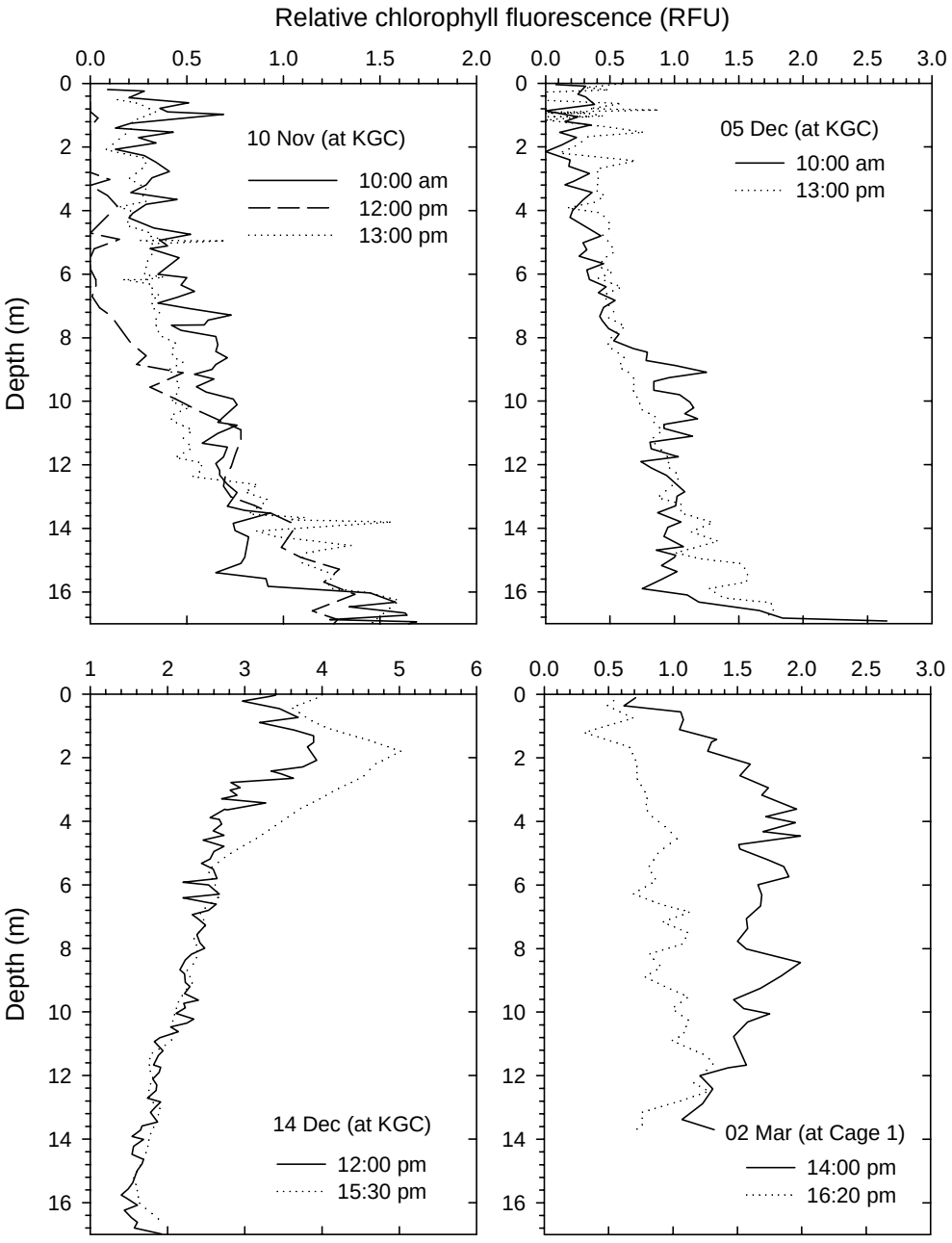


Fig. 3.18. Repeated relative chlorophyll fluorescence profiles taken with a bbe Moldanke Fluoroprobe at KGC and Cage #1 stations. Note the changing scale in the x-axis.

control stations the ranges were 2.7-4.7, 0.5-0.7, 0.06-0.09 and 0.004-0.008 g m^{-2} d^{-1} for PM, PC, PN and PP respectively. The mean sedimentation rates for PM, PC, PN and PP were, respectively, on average 13, 21, 18 and 8 times higher under the cages than at control stations (Table 3.2). The high variability of sedimentation rates under the cages could be due to the fact that the cages studied differed in size of the fish contained (28-262 g per fish), fish biomasses in the cages (2,612-23,875 kg) and daily feed rations (6.2-21.6 g feed/kg fish/day). Further analysis revealed that sedimentation rates were generally higher under cages containing large fish (181-262 g/fish) consuming "finisher" feed type than under cages stocked with medium fish (138-154 g/fish) consuming "grower" feed type or small fish (28-80g/fish) consuming "starter" feed (Fig. 3.20). These differences could be related to the different sedimentation velocities of the fecal material produced by these fish groups. Generally, fecal pellets from bigger fish have higher sedimentation velocities than fecal pellets from smaller fishes. The mean percent composition of C, N and P in particulate material falling under the cages ranged between 11.9-45.0% C, 1.3-5.5% N and 0.05-0.23% P while at control stations the percent compositions ranged between 13.4-22.6% C, 1.6-2.8% N and 0.12-0.17% P. In the feed, percent compositions ranged between 40.2 and 49.7% for C, 4.0 and 7.5% for N and between 0.7 and 1.2% for P.

Because C, N and P wastes estimated from mass balance are proportional to feed supply (Chapter 2), further analyses were done to determine any relationships between sedimentation rates of PM, PC, PN and PP under the cages and daily feed rations and fish biomass in cages (Figs. 3.21, 3.22 and 3.23). This is important because if a relationship exists, then particle sedimentation rate can be estimated from feed supply and/or total fish biomass in a cage. As

expected from Chapter 2, a strong correlation was observed between daily feed ration added to cages (kg feed $m^{-2}d^{-1}$) and total fish biomass (kg fish) in the cages (Fig. 3.21a). However, we observed weak ($0.060 \leq r^2 \leq 0.141$) and non-significant ($0.206 \leq p \leq 0.419$) correlations between total fish biomass in each cage and sedimentation rates of PM, PP, PN and PC below the cages (Figs. 3.21b, c, d & e). Similar weak correlations were observed between daily feed ration (kg feed m^{-2} d^{-1}) and the sedimentation rates of PM, PC, PN and PP under the cages (Fig. 3.22a, b, c and d). However, strong correlations ($p<0.001$) were observed between sedimentation rates (g m^{-2} d^{-1}) of PM and those of PC, PN and PP in Fig. 3.23a, b and c respectively.

3.4.8 Extracted chlorophyll *a* and chlorophyll fluorescence

Extracted chlorophyll concentration, commonly used to estimate phytoplankton standing biomass in aquatic systems (Dillon & Rigler, 1974), varied between 0.22 and 2.84 μg L^{-1} at the KGC station, between 0.14 and 2.41 μg L^{-1} at the south stations and 0.17 and 2.61 μg L^{-1} at the north stations (Fig. 3.14). A mean chlorophyll concentration of 0.90 μg L^{-1} at the KGC station was slightly higher than mean concentrations of 0.81 μg L^{-1} and 0.83 μg L^{-1} at the south and north stations, respectively. Chlorophyll concentrations along the transect were generally highest in May at the beginning of the mixing season probably due to algal (especially diatoms) resuspension but declined immediately after the peak.

Chlorophyll fluorescence has often been used to estimate extractable chlorophyll concentration in aquatic systems (Yentsch & Menzel, 1963; Lorenzen, 1966). In situ chlorophyll

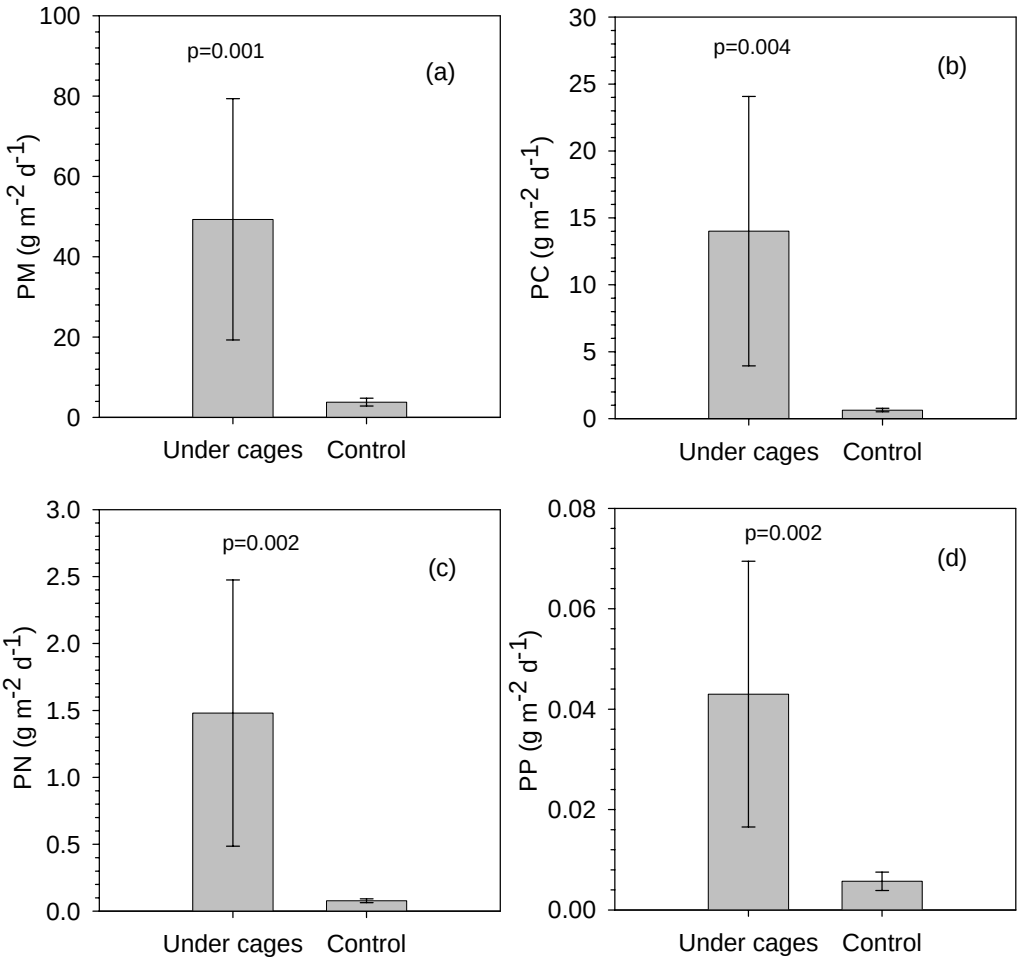


Fig. 3.19. Sedimentation rates (g m^{-2} d^{-1}) of particulate material (PM) (a), particulate carbon (PC) (b), particulate nitrogen (PN) (c) and particulate phosphorus (PP) (d) under the cages and at control stations. The difference between sedimentation rates under the cages and at control stations is indicated by the p value in each figure.

fluorescence measured along the study transect with CTD WETStar fluorometer varied between 1.0 and 7.5 μg L^{-1} (Fig. 3.15 & 3.16). A linear regression between extracted chlorophyll at 2 and 10 m depths against CTD chlorophyll fluorescence recorded at the same depths indicated a significant relationship (Fig. 3.17; $r^2 = 0.702$; $p<0.001$). The regression coefficients between chlorophyll fluorescence and extracted chlorophyll at 2 m (2.17) and at 10 m (1.96) were not significantly different at alpha of 0.05 which suggests that the algal assemblages at 2 and 10 m depths were the same and light had little effect on the fluorescence measurements at the two depths. In addition, alga cells require 2 seconds only for dark adaptation before excitation (Buetler et al., 2002) which is possible to get with the pumped CTD[2] that was used in the study so that regardless of previous light regime, algal cells at both depths would probably still emit optimal fluorescence in a pumped CTD. Fluoroprobe (bbe) casts in February, March and between November and December, 2007 indicated that the algal community in the water column along the study transect was generally dominated by diatoms and dianoflagellates. Fluoroprobe casts done at the cage station (KGC) in February and March, and between 10 November and 05 December, 2007 indicated that 50.2±28.7% of total chlorophyll fluorescence originated from golden-brown algae while green algae, cyanobacteria and phycoerythrin-containing algae contributed 25.9±29.5, 4.6±4.7 and 19.2±18.8%, respectively. Fluoroprobe (bbe) casts done on 11 and 14 December, 2007 show an increased abundance of cyanobacteria (17.4±5.7%) and cryptophyta (35.5±6.6%) and a decreased abundance for brown (41.3±17.2%) and green algae (5.8±6.4%).

[2] A pumped CTD uses a submersible electric pump to move water through tubing to sensors on the CTD for testing.

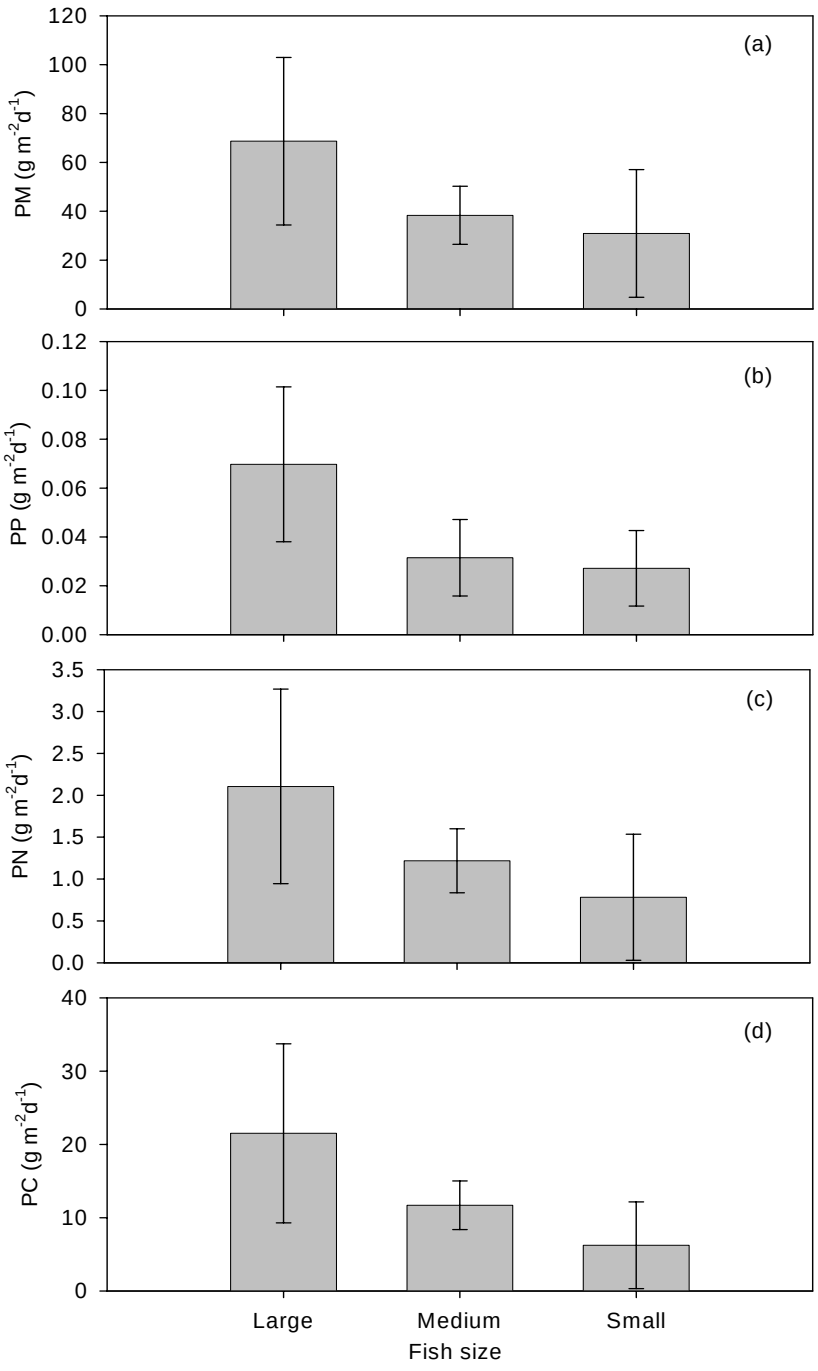


Fig. 3.20. Sedimentation rates (g m^{-2} d^{-1}) of particulate material (PM) (a), particulate carbon (PC) (b), particulate nitrogen (PN) (c) and particulate phosphorus (PP) (d) under cages stocked with large, medium and small fish.

Buoyant cyanobacteria were possibly responsible for the high surface chlorophyll fluorescence observed in CTD casts done along the study transect on 14 December 2007 (Fig.3.16). Fig. 3.18 shows the dynamic nature of algal abundance in the water column as indicated by *in situ* chlorophyll fluorescence readings recorded in Fluoroprobe (bbe) casts done within hours of each other. The multiple CTD casts clearly show that the chlorophyll fluorescence profiles were different from cast to cast over the space of a few hours (Fig. 3.18).

3.4.9 Underwater observations

Large numbers of wild fishes were observed around the cages. From personal observation using underwater camera and snorkeling, the fish density was much higher in surface waters (~3m) around the cages than deeper in the water column. Density also decreased rapidly with distance from the cages. No similar numbers of fish were observed at control stations. In surface waters (<2m) immediately around fish cages (<0.5m), fish density tended to be higher on the leeward side during feeding as feed was thrown into the cages from the windward side of the cages. Similarly, blind caged fish, which could not visually feed, tended to aggregate in surface waters at the leeward side of the cages collecting feed dust on the water surface which probably suggests that a significant amount of waste feed was spreading horizontally with current. While the surface waters were dominated by larger and more mature fish (relative to other wild fishes), the area under the cages were sparsely populated and seemed to be dominated by small, probably immature individuals (personal observations).

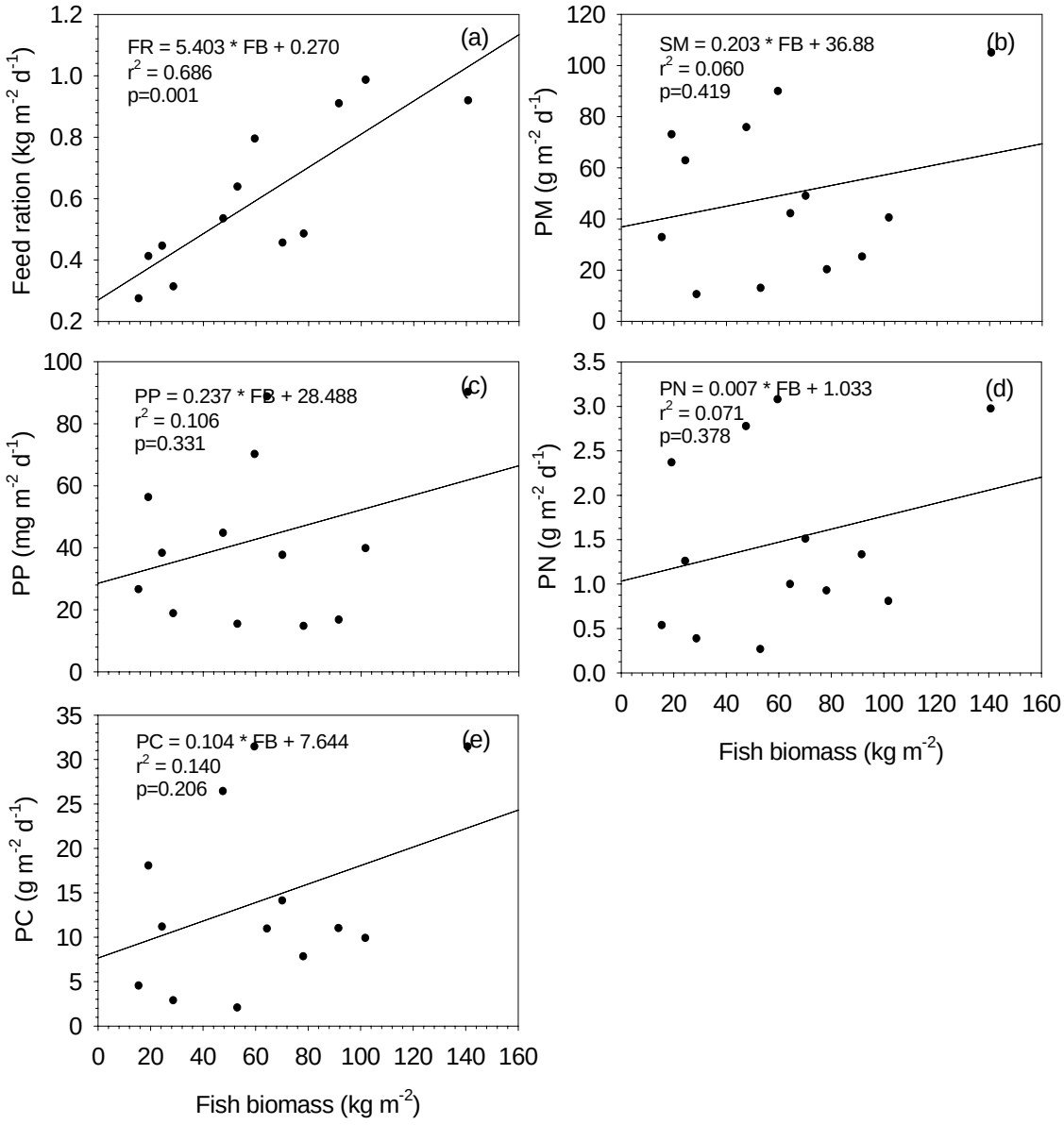


Fig. 3.21. Correlations between fish biomass (kg fish m^{-2} cage area) and (a) feed ration (kg m^{-2} d^{-1}), (b) particulate material (PM, g m^{-2} d^{-1}), (c) particulate phosphorus (PP, mg m^{-2} d^{-1}), (d) particulate nitrogen (PN, g m^{-2} d^{-1}) and (e) particulate carbon (PC, g m^{-2} d^{-1}).

3.5 Discussion

3.5.1 Meteorology and water currents through the fish cages

Malawi's climate has been described as having a warm and dry summer (October–November), a warm and wet austral summer (December–April) and finally a cool and dry winter between May and September (Torrance 1972). These climate seasons are evident in the meteorological data presented here from Mangochi township (Fig. 3.3). Air temperature and wind speed and direction changed with the seasons. According to Eccles (1974) the winds over Lake Malawi tend to be northerly during the rainy season (November to April) while strong and persistent southeasterly winds predominate between May and September. During the mixing period between May and September, strong and persistent southeasterly winds force epilimnetic water to pile up at the northern end of the lake which tilts the thermocline so that cooler nutrient-rich metalimnion water is brought closer to the surface at the shallow southern tip of the lake (Eccles, 1974; Owen & Crossley, 1992; Hamblin et al., 2003). Frequent upwelling of nutrient-rich waters has enabled the southern part of Lake Malawi to become the most productive part of the lake in terms of algal (Hecky & Kling, 1987; Bootsma, 1993), zooplankton (Irvine & Waya, 1999) and fisheries (Turner, 1977a & b) productions. Easterly winds briefly blow across the lake, particularly in the morning in October. The northerly winds are generally weaker and less persistent than the southeasterly, as depicted by wind speed and air temperature data from Mangochi township for these periods presented in Fig. 3.3. However the long fetch length (>500 km) of the northerly winds produces high southerly surface waves and bottom currents through

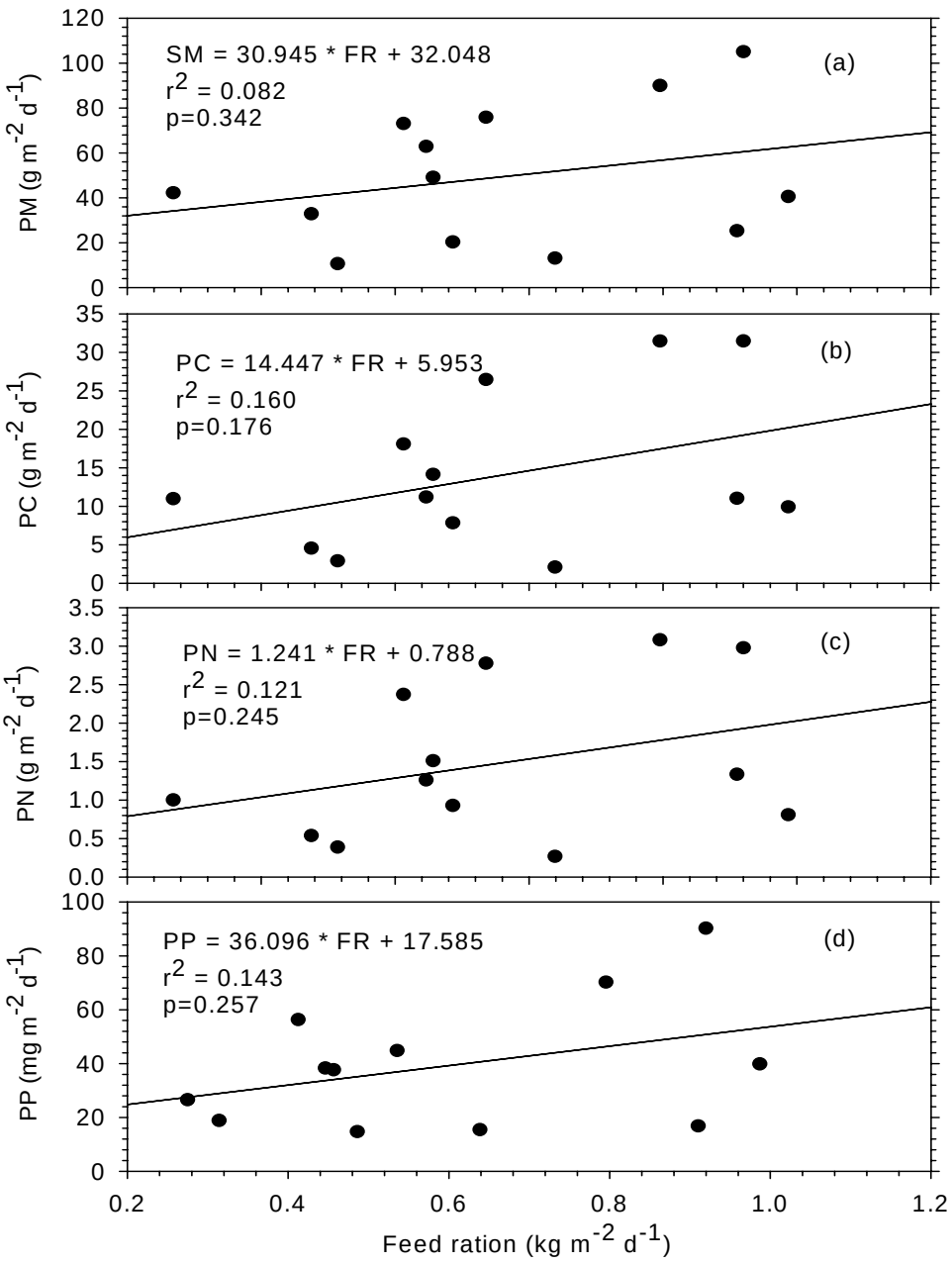


Fig. 3.22. Correlations between areal feeding ration (kg feed m^{-2} d^{-1}) and sedimenting particulate material (PM, g m^{-2} d^{-1}) (a), particulate carbon (PC, g m^{-2} d^{-1}) (b), particulate nitrogen (PN, g m^{-2} d^{-1}) (c) and particulate phosphorus (PP, mg m^{-2} d^{-1}) (d). Note the different units for PP.

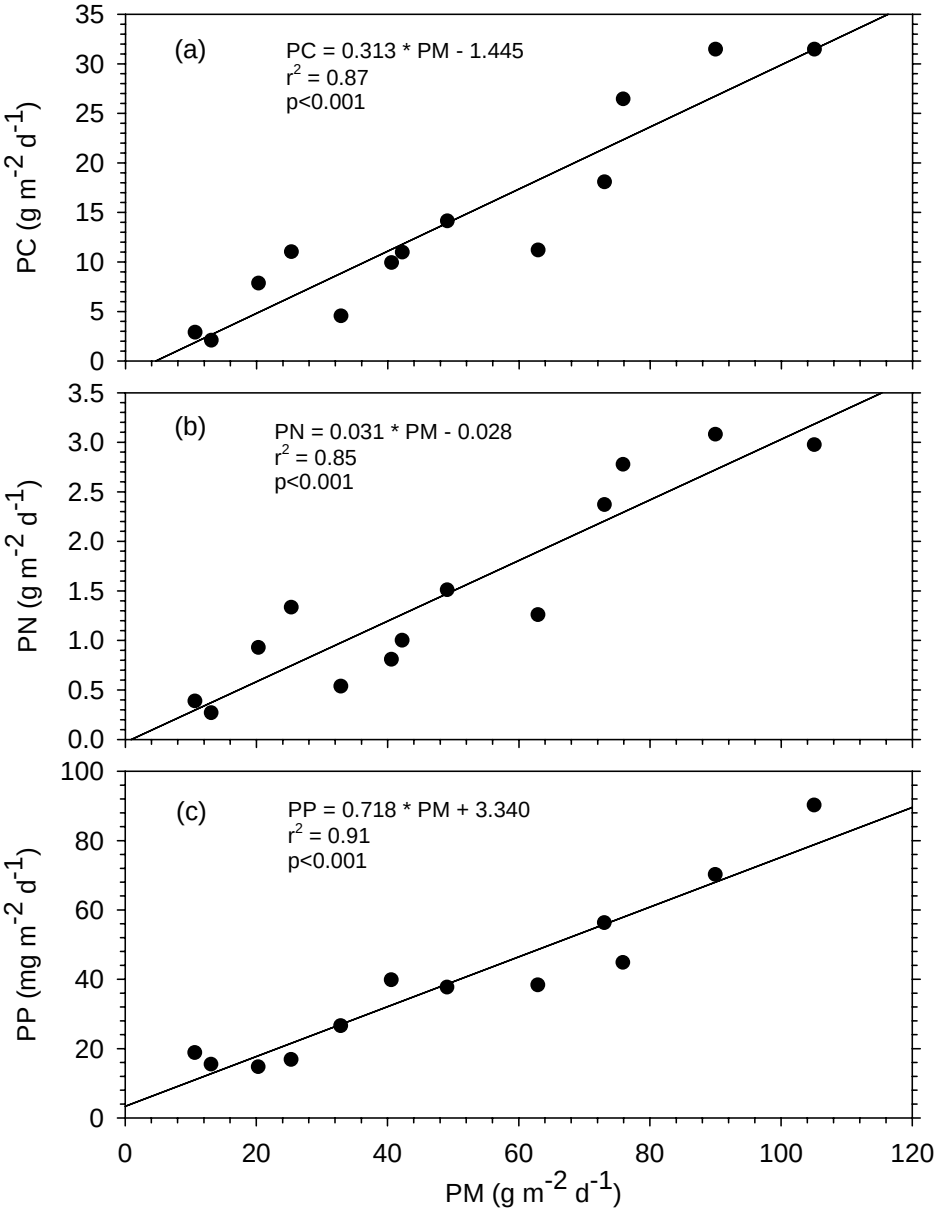


Fig. 3.23. Correlations between sedimenting particulate material (PM, g $m^{-2}d^{-1}$) and particulate phosphorus (PP, mg $m^{-2}d^{-1}$) (a), particulate nitrogen (PN, g $m^{-2}d^{-1}$) (b) and particulate carbon (PC, g $m^{-2}d^{-1}$) (c) under the cages.

the fish farm which were on average about 9.3 cm s^{-1} (see below).

Many previous studies have shown that water temperature and water current velocity and direction in lakes are usually controlled by atmospheric forcing. For example, water currents below the fish farm followed wind directions on the lake's surface during the study period except for the easterly winds during which, because of the closeness of the study site (fish farm) to the western shore, bottom currents moved eastward instead of westward with the easterly winds. Consequently atmospheric forcing also affects dispersion of wastes from fish farms and other point sources in lakes. Water temperatures observed along the 10 km transect through the fish farm in Lake Malawi ranged between 21.5-30.0 ^{o}C (Fig. 3.8) which could facilitate rapid decomposition of organic wastes as they disperse from the fish farm. Vertical stratification of the water column was observed between November and December and may have been present in March (when we were unable to sample) which could enhance the accumulation of cage wastes in the lower water layers and on the bottom. Because the main source of oxygen to deeper water layers is through vertical mixing with surface layers, vertical stratification of the water column may lead to de-oxygenation of bottom waters at the sediment-water interface. The low oxygen patches in bottom waters observed on 20th January and 30th March 2007 (Fig. 3.9) might be attributed to lack of vertical exchange of oxygen with top water layers caused by vertical stratification of the water column. The thermal structure on 10th May suggests that upwelling of cooler and less oxygenated deeper water could have occurred. But even then observed occurrences of such conditions were rare and total anoxia or even persistent hypoxia was not observed in the vicinity of the cages.

Table 3.1. Secchi depths, PAR extinction coefficients (*k*), extracted chlorophyll (Chla) concentrations, and total suspended solid (TSS) levels measured at stations along the study transect. The values are mean±SD. Included are the station locations, depths (m) and distances (km) to fish farm (FF). Note CS and CN stations are to the south and north respectively of the cage area. KGC is at the centre of the cage area.

Station	Depth (m)	Distance to FF (km)	Secchi (m)	Ext. coef (k) (m-1)	Chla (µg L^{-1})	TSS (mg L^{-1})	Station Locations	
							Latitude (S)	Longitude (E)
CS4	15.8	4.4	5.9±1.7	0.29±0.07	0.84±0.52	1.31±0.40	14° 21.356’	035 ° 11.440’
CS3	16.7	2.9	5.8±1.9	0.28±0.06			14 ° 20.749’	035 ° 10.838’
CS2	17.5	1.7	6.1±2.2	0.27±0.06	0.83±0.54	1.13±0.60	14 ° 20.252’	035 ° 10.410’
CS1	15.3	1.0	6.0±2.2	0.27±0.04			14 ° 20.046’	035 ° 09.990’
KGC	17.1	0.0	5.9±2.0	0.27±0.07	0.91±0.49	1.45±0.89	14 ° 19.549’	035 ° 09.844’
CN1	16.1	1.1	5.5±1.5	0.26±0.03			14 ° 19.084’	035 ° 09.452’
CN2	14.6	2.3	5.3±1.5	0.28±0.08	0.78±0.51	1.62±0.81	14 ° 18.637’	035 ° 09.017’
CN3	15.5	4.1	6.5±2.1	0.23±0.06			14 ° 17.564’	035 ° 08.872’
CN4	23.1	6.2	6.8±2.1	0.21±0.05	0.83±0.48	1.24±0.98	14 ° 16.644’	035 ° 08.081’

Table 3.2. Mean (± SD) concentrations (µg L^{-1}) and sedimentation rates (g $m^{-2}d^{-1}$) of particulate material (PM), carbon (PC), nitrogen (PN) and phosphorus (PP) at stations along the study transect and under the cages. Sedimentation rates were estimated from sediment trap samples collected under the cages (UC) as well as at control stations (CS). Deployment of sediment traps were repeated 13 times under the cages (UC) and 3 times at control stations (CS). The cages studied for sedimentation rates were stocked with small, medium and large fish and their diets were therefore also different (starter, grower and finisher respectively).

	CS4	CS2	KGC µg L^{-1}	CN2	CN4	CS g $m^{-2}d^{-1}$	UC g $m^{-2}d^{-1}$	UC/CS (ratio)
PM	-	-	-	-	-	3.78±0.97	49.30±28.87	13
PC	303.7±75.9	278.4±76.7	279.2±55.1	343.3±191.0	297.0±100.4	0.64±0.13	14.01±10.07	21
PN	40.75±11.52	36.05±7.61	37.42±6.32	40.61±12.83	39.97±11.28	0.08±0.01	1.48±1.00	18
PP	3.79±1.34	3.37±1.31	3.80±1.59	3.39±1.21	3.32±1.32	0.01±0.00	0.05±0.03	8

Sedimentation rates of PM, PC, PN and PP (g $m^{-2}d^{-1}$) were measured using sediment traps deployed under the cages and at control stations. The top area of the circular cages is 170 m^2. PC, PN and PP (µg L^{-1}) were measured in water samples collected with a van Dorn bottle.

Table 3.3. Mean (±SD) molar ratios of particulate C to particulate N (C:N), particulate C to particulate P (C:P) and particulate N to particulate P (N:P) in water samples along the study transect, in fish feed and in sedimenting particles collected in sediment traps deployed under the cages and at control stations. Water samples for particulate C, N and P analysis were collected between January and July 2007.

Sample type	Station	C:N	C:P	N:P
Fish feed		10.2±2.1	136.7±20.8	14.2±4.1
Sediment trap samples				
	Under cages	10.8±1.9	796.9±477.2	75.3±45.6
	Control stations	9.6±0.5	309.1±99.7	32.5±10.7
Water samples				
	CS4	8.9±1.7	170.2±37.8	20.6±3.8
	CS2	9.0±1.6	169.5±39.6	18.8±3.0
	KGC	8.8±1.4	188.3±49.9	23.2±9.9
	CN2	9.5±2.5	190.0±35.8	22.1±4.0
	CN4	8.6±1.3	229.6±47.0	27.1±6.7
	South stations	9.0±1.6	169.8±37.8	19.7±3.5
	North stations	9.1±2.0	210.8±45.7	24.6±6.0

*Interpretation of the ratios in the table is according to Guildford and Hecky (2000, *Table 1*)

Table 3.4. List of wild fish species which aggregated around the fish cages in Lake Malawi. The dominant fish species were the zooplanktivorous *Copadichromis* spp. Source: Trawling survey by the Malawi Department of Fisheries (unpubl. data) and from hand net samples caught in surface waters around the cages. Diets and preferred habitats of the species are according to Turner (1995) and references therein. Trophic levels are according to Froese and Pauly (2009).

Species	Habitat	Diet	Trophic Level
Astatotilapia calliptera	Vegetated areas in shallow waters, rivers & streams	Benthivore	3.0
Buccochromis atritaeniatus		Piscivore	4.5
Buccochromis heterotaenia	Areas of mixed rocks & sand and vegetated areas	Piscivore	4.5
Buccochromis nototaenia	Sandy-	Piscivore	3.0
Champsochromis caeruleus	Inhabits open waters over rocky and sandy areas	Piscivore	4.5
Copadichromis chrysonotus	Inshore waters <20m	Zooplanktivore	3.4
Copadichromis eucinostomus	Open waters	Zooplanktivore	3.0
Copadichromis virginalis	Demersal, sand dwelling species	Zooplanktivore	3.1
Ctenopharynx intermedius	Intermediate	Zoobenthivore	3.4
Ctenopharynx pictus	Intermediate	Zoobenthivore	3.0
Diplotaxodon limnothrissa	Offshore pelagic, absent in sallow inshore waters	Zooplanktivore	4.4
Engraulicypris sardella	Pelagic	Zooplanktivore	3.1
Fossorochromis rostratus	Shallow sandy areas	Zoobenthivore	3.4
Hemitilapia oxyrhynchus	Shallow sediment rich bays	Algavore	2.0
Lethrinops macrochir	Sandy -shallow water over sandy bottoms	Benthivore	3.0
Mylochromis anaphyrmus	Sandy (>10 m)	Molluscivore	
Mylochromis lateristriga	Shallow waters, mainly in southern Lake Malawi	Arthropods	
Nimbochromis livingstoni	Shallow sediment rich bays	Piscivore	4.5
Nimbochromis polystigma	Areas of mixed rocks and sand, vegetated areas	Piscivore	4.5
Nyassachromis argyrosoma	Dominant in shallow waters < 20 m	Zoobenthivore	
Opsaridium microlepis	Pelagic waters	Piscivore	4.0
Oreochromis karongae	Sandy, shallow vegetated bays, rocky biotopes etc	Planktivore	2.0
Placidochromis subocularis	Sandy - shallow waters over sandy areas	Zoobenthivore	3.4
Protomelas similis	Common in shallow weedy areas	Herbivore	2.0
Protomelas triaenodon	Common over sandy bottoms at depth 10-18 m		3.0
Pseudotropheus elegans	Sandy - common in open sandy areas <10 m	Zoobenthivore	3.3
Rhamphochromis spp	Pelagic	Piscivore	4.5
Stigmatochromis woodi	Sandy	Piscivore	4.5
Trematocranus microstoma	Sandy - shallow inshore waters including vegetated areas	Zoobenthivore	3.4
Trematocranus placodon	Sandy (shallow waters)	Molluscivore	3.5

Water currents are important in lakes and oceans because they influence the transport pathways of nutrients and suspended particles (Beletsky et al., 1999; Simons & Schertzer, 1985; Hamblin et al., 2003) and can result in nutrient distributions that are spatially variable (horizontally and vertically) and declining away from major nutrient sources in most large lakes (Simons & Schertzer, 1985) including Lake Malawi (Bootsma, 1993; Patterson & Kachinjika, 1995; Hamblin et al., 2003; Gondwe et al., 2008). At high current velocities, rapid water renewal can make nutrient gradients difficult to measure unless a point source is very strong. In water bodies where cage culture is operated, adequate water currents are necessary to disperse cage wastes. Faster water currents minimize impacts of fish farms on the surrounding environment by dispersing nutrient and particulate wastes over a larger area than is usually the case under low water current regimes. Gowen and Bradbury (1987) suggested that cage wastes can only be dispersed to 1.5-2 times the area of the farm itself, but recent studies employing stable isotope analyses have shown that with high water currents, wastes from cages can be dispersed over much larger areas and distances than previously suggested. Sarà et al (2004) studied waste dispersion from a small fish farm of 4 cages located in a high current area in the Mediterranean Sea. The study detected cage wastes over many hectares of water and sediments particularly along the directions of the main water currents through the fish cages. However, because current direction can be random, wastes may be carried and deposited in all directions around fish farms as Holmer et al (2007) showed in the Mediterranean Sea. In lakes, water currents have generally been described as less persistent due to more rapid frictional dampening of currents and so they depend strongly on short-term atmospheric forcing due to their size compared to oceans where currents/gyres are more stable (Beletsky et al., 1999). In Lake Malawi, water current strength and

direction were seasonal (Fig. 3.4) generally controlled by meteorological forcing over the lake. In contrast to currents in many lakes, water currents through the cages in Lake Malawi were high enough to disperse the nutrients and particles generated in the cages so that no significant differences were found between control areas and the cage site. At a mean current speed of 9.3 cm s^{-1} and a range of 0.4-47.2 cm s^{-1}, the currents through the fish farm were higher than water currents observed in many large lakes including the North American great lakes. In the North American great lakes, average summer currents range between 1 and 2.4 cm s^{-1} while winter currents range between 1.6 and 2.8 cm s^{-1} giving an annual mean of only <2 cm s^{-1} (Beletsky et al., 1999). Below the fish farm in Lake Malawi, only 15 days out of ~200 days of data acquisition with FSI 2D-ACM acoustic current meter registered full 24 hrs of <10 cm s^{-1} current speed. This shows that higher currents through the fish farm were frequent enough to significantly disperse sedimenting and even settled cage wastes, to distant locations.

The difference in current speeds between the North American great lakes and Lake Malawi could most likely be due to the morphology of the lakes combined with the different meteorological conditions – tropical versus temperate conditions. Circulation patterns in Lakes Huron, Michigan and Superior tend to be cyclonic because of their larger surfaces areas (Beletsky et al., 1999) while narrow and elongated Lakes Malawi and Tanganyika have uniform wind fields along their axes and vertical circulation patterns which produce upwelling at the southern parts of the lakes as opposed to developing persistent circulation gyres. Also wind forcing in temperate great lakes responds to well developed frontal systems that are characterized by rapidly varying wind intensities and directions during their passage. In contrast, the wind

patterns on Lake Malawi are characterized by persistent directions reversing only with the seasons. These dominant seasonal winds tend to orient along the rift valleys allowing very persistent current movements which are not disrupted by changing wind directions.
While water currents at the bottom of the fish farm (mean of 9.3 cm s^{-1}) were higher than currents in most large lacustrine systems, no measurements were made in surface waters but are likely to be higher due to direct wind stress on the water surface. Surface currents of at least 27.8 cm s^{-1} (1 km h^{-1}) have been reported in the southern part of the lake by Eccles (1974).

3.5.2 Physical-chemical data in the water column along the study transect

Using different methods, researchers have shown that fish farms discharge large amounts of orthophosphate and nitrogenous nutrients into the surrounding waters (Hall et al., 1990; Holby & Hall, 1991; Hall et al., 1992). In some cases, particularly in sheltered and semi-enclosed areas, high nutrient concentrations have been reported in surface waters adjacent to fish cages (e.g. Neofitou & Klaoudatos, 2008). Most studies in high current zones fail to show significant differences in nutrient concentrations or algal abundances between cages and control sites (e.g., Guildford, 1993; Pitta et al., 1999). In the fish farm in Lake Malawi, a mass balance (input = output) method using recorded feed inputs, juvenile fish stocked, fish mortalities and fish harvests revealed that large amounts of organic and nutrient wastes (85% of C, 76% of N and 88% of P added to the cages through feed) were discharged into the water column in dissolved and particulate forms (Fig. 2.4, Chapter 2). Although nutrient losses can depend on the species

being cultured, these nutrient losses from the fish farm in Lake Malawi are in general similar to losses observed in many cage culture operations around the world. Hall et al (1990; 1992) and Holby and Hall (1991) reported losses of 80% of C, 76% of N and 82% of P from rainbow trout cages in coastal marine waters of the Gullmar Fjord, western Sweden. The losses from the fish cages in Lake Malawi were consistent with the high feed conversion ratios which varied between 2.1 and 3.9 (mean of 2.7) estimated for the tilapia (chambo) species cultured in the lake (Chapter 2). Holmer et al (2007) reported similar large losses from cages stocked with sea bass and sea bream species which had relatively lower feed conversion ratios (1.6 to 2.4) in the open oligotrophic Mediterranean Sea. Despite the large estimated discharges of orthophosphate and nitrogenous nutrients, no significant differences were observed between the fish farm station (KGC) and control stations along the study transect in NH_4^+, NO_3^-, SRP, PC, PN, and PP concentrations. Nevertheless, mean TSS, NO_3^- and PP concentrations were relatively higher at the fish farm station (KGC) than at control stations along the study transect indicating that cage operations did have some minimal impacts on the surrounding environment.

Increased oxygen demand caused by enrichment of the water with cage wastes may reduce DO in the water column, particularly bottom waters (Mente et al, 2006). In this study, however, no significant correlations could be established between cage waste loading and dissolved oxygen (DO) and chlorophyll concentrations in the water column. Low oxygen patches were, however, observed below the fish cages which could partly be attributed to cage operations which increased biological oxygen demand in the vicinity of the cages as well as obstruction of water current flows by the cage structures.

The molar C:N, C:P and N:P ratios presented in Table 3.3 suggest that suspended particulate organic matter along the 10 km study transect in the fish cages in Lake Malawi was moderately N and P deficient between January and July 2007. These results are in agreement with many studies which have generally described Lake Malawi as seasonally P and N limited (Hecky et al., 1996; Guildford & Hecky, 2000; Guildford et al, 2000, 2003, 2007; North et al., 2008; Gondwe et al., 2008). Consequently, high discharges of both N and P into the lake should, in principle, increase algal biomass commonly measured as chlorophyll concentration in the water column. Cage culture therefore presents a potential point-source of these N and P nutrients to the lake's water column. The molar N:P ratio for the feed of 14.2:1 suggests that the feed has high levels of P relative to N so that feed losses into the environment may favor the proliferation of N_2-fixing algal species such as the *Anabaena* cf. *maxima* and *A. discoidea* Schmidle previously observed in the lake (Gondwe et al., 2008). Some of the N_2-fixing algal species are potentially toxic to fish and other animals (Kiirikki et al., 2001).

As with some nutrients and DO, no significant differences in chlorophyll concentrations and chlorophyll fluorescence values between cage and control stations were observed in the southeast arm of Lake Malawi. The mean chlorophyll concentration of 0.90 μg L^{-1} at the fish farm station (KGC) was only slightly higher than concentrations at the south (0.81 μg L^{-1}) and north (0.83 μg L^{-1}) control stations which suggests that at the 2007 fish production capacity of ~200 tonnes, the fish farm in Lake Malawi had little effect on the surrounding environment.

These findings are consistent with observations reported by many researchers which show that the severity of impacts of fish farms depends, among other factors, on the hydrology of

the area holding the fish farms (Wu et al., 1994; Gillibrand & Turell, 1997; Dudley et al., 2000; Alongi et al., 2003). As discussed above, high water current speeds such as those currents (mean of 9.3 cm s^{-1}) observed at the fish farm in Lake Malawi quickly disperse cage wastes before significant build-ups can occur (Karakassis et al., 2005). The location of the KGC station at the center of the fish farm rather than immediately adjacent to the cages also affected our ability to observe high nutrient discharges from the cages. However the KGC station was intended to represent the whole fish farm, which was necessary for the determination of collective nutrient discharges from the fish farm. Also, the 4 columns x 12 rows cage fish farm was constructed perpendicular to the main directions of the two prevailing winds, southeasterly and northerly, which blow along the axis of the lake. This orientation of the cages meant that cage wastes could less easily be measured at the KGC station because cage wastes were dispersing perpendicular to the KGC station almost throughout the year except, briefly when easterly winds blew across the lake to the west and bottom water moved in a counter direction to the east. Finally, the nutrient concentrations might also be affected by the high consumption of the wastes by large populations of opportunistic wild fishes around the cages (Figs 2.1 & 2.2) which increased dispersion rates of the wastes and the area of deposition around the cages as discussed above.

While most of the low oxygen patches observed in bottom waters could be attributed to large scale water movements, some localized instances of low oxygen levels, such as on January 20 and March 30 (Fig. 3.9), could partially be attributed to cage culture operations in the lake. Strong stratification in the affected bottom waters on March 30 might have hindered oxygen exchange with the surrounding water masses (Fig. 3.6).

Similar to nutrients, dissolved oxygen and chlorophyll concentrations discussed above, no significant differences were observed for TSS, Secchi depth and PAR extinction coefficient between cage and control stations along the study transect. These findings are also in agreement with observations by Beveridge et al (1994) and Wu et al (1994) who described the changes in TSS, PAR extinction coefficient and chlorophyll around their study fish farms as insignificant or localized.

Sedimentation studies in tilapia cages in tropical lake systems are rare but studies in salmonid cages have shown that a significant portion of the total wastes from cages are in particulate forms; 29 to 71% of C (Hall et al., 1990); 59 to 66% of P (Holby & Hall, 1991); and 49 to 51% of N (Hall et al., 1992). This suggests that higher sedimentation rates are expected under the cages than at stations far from the cages dominated by seston material only. As discussed above, the sedimentation rates of cage wastes may be modified by strong currents through the cages and consumption of cage wastes by wild fishes so that estimation of actual losses becomes difficult under these conditions. For example, the mass balance model in Fig. 2.4 (Chapter 2) indicates that about 1.5 kg P was discharged per day (534 kg P/yr) from the cages in Lake Malawi but only 0.6% of the discharged P (8.5 g P/day; Table 3.2; i.e., multiply 0.05 g m^{-2} d^{-1} by 170 m^2 cage area) was determined in particulate material collected in sediment traps under the cages. Consequently, it was not possible to estimate the proportions of particulate and soluble C, N and P in the wastes discharged from the cages in Lake Malawi as was done in salmonid cages. A reasonable estimate of losses can only be obtained if sediment traps were deployed under guarded cages (with nets) during calm days to eliminate the influence of wild fish

consumption and current dispersion. Nevertheless, the sedimentation rate of particulate material (PM) in Lake Malawi was 13 times higher under the cages than at control stations. Similar results have been reported by Holmer et al (2007) around fish cages in the Mediterranean Sea where sedimentation rate of particulate material was 8-25 times higher under the cages than at control stations. Higher sedimentation rates under the cages have been attributed to the fact that waste feed and feces from the cages are larger and denser than seston material so that they quickly sink underneath or close to the cages. However, at high current velocities the cage wastes have been shown to spread further away from the cages than at low current velocities. Holmer et al (2007) determined cage wastes in all directions around a fish farm by sampling along transects perpendicular to a dominant current direction through the fish farm. This observation by Holmer et al (2007) may help to partly explain the lack of significant differences for almost all parameters (particularly, in this regard, TSS) measured between downstream and upstream stations along the dominant current direction when dispersion by water current generally predicts that downstream stations should be more affected by the cage wastes than the upstream stations. As with particulate material (PM), sedimentation rates of PC, PN and PP observed in this study in Lake Malawi were on average 21, 18 and 8 times higher under the cages than at control stations which suggest that fish cages can act as important points sources of organic matter and nutrients to the sediments beneath and the water column surrounding the cages. In the study by Holmer et al (2007) in the Mediterranean Sea, sedimentation rates for PC, PN and PP were respectively 4-27, 2-10 and 20-1200 times higher under the cages than at control stations. Because of rapid dispersion of cage wastes by strong water currents and

consumption by wild fish, sedimentation of PM, PC, PN and PP from the fish cages could not be quantitatively related to fish biomass in cages as well as feed rations supplied to the cages.

3.5.3 The role of wild fishes in waste dispersion

The capacity of floating structures, including fish cages, to attract and aggregate wild fish species has been demonstrated in many water bodies (Carss, 1990; Bjordal & Skar, 1992; Deudero et al., 1999; Freon & Dagorn, 2000; Castro et al., 2002; Holmer et al., 2007). In stocked fish farms, the main factor attracting and retaining wild fishes to the cages is the persistent fallout of unused feed and feces from the cages as a result of the daily feeding of caged fish rather than just the floating structures (Tuya et al., 2006). The presence of fish cages in an area could therefore affect the presence, abundance, diet and residence times of wild fishes in that area (Carss, 1990). During the period of this study, large numbers of wild fishes were observed around the fish cages in Lake Malawi (Fig. 2.2 & 2.3) which agrees with observations around many fish cages around the world. The number of wild fishes around the cages was, however, not estimated in this study but could be comparable to counts of 2,000-86,000 reported by Dempster et al (2004) around fish cages in the south-western Mediterranean Sea. The fish species composition around the cages in Lake Malawi was also high at >31 fish species as determined in a survey conducted by the Fisheries Department in January 2007 (Table 3.4). The wild fish species around the cages in Lake Malawi were dominated by zooplanktivorous *Copadichromis* spp. The high abundance and diversity of the wild fish species around the cages

in Lake Malawi may be attributed to the proximity of the fish farm to the shore (~1 km) as at the three fish farms in the Mediterranean Sea studied by Dempster et al (2005). In addition, the fish abundance is generally higher in the southeast arm compared to the rest of the lake (Turner, 1995) attributed to higher planktonic and benthic primary production (Bootsma et al., 1996) encouraged by frequent upwelling of nutrient-rich deep waters in the area (Eccles, 1974; Hamblin et al., 2003).

As discussed above, the fish density was much higher in surface waters around the cages, particularly during feeding, but the density decreased with depth as well as with distance from the cages (personal observations). During feeding, wild fish were easily caught with a hand net in surface waters around the cages (Fig. 2.2 & 2.3). No appreciable numbers of fish were observed at control stations. Such rapid drops in fish abundance with distance from the cages have previously been reported at many cage fish farms (e.g., Dempster et al., 2002). In the immediate surface waters around fish cages in Lake Malawi, fish density tended to be higher on the leeward side during feeding as feed was thrown into the cages from the windward side of the cages. While the surface waters were dominated by larger and mature fish, the area under the cages were less populated and seemed to be dominated by small, probably immature individuals (personal observations). Dempster et al (2005) studied fish aggregation patterns around fish farms in the Mediterranean Sea and reported highest wild fish abundance and biomass in surface waters around one farm while an opposite pattern (highest abundance and biomass on the bottom beneath the cages) was observed around the other although both farms were dominated by similar planktivorous species. The high abundance of wild fishes in surface waters around the

cages suggests that most of the feeding by the wild fish occurred in surface waters as compared to the area below the cages. For feeding to occur in surface waters, it requires that most of the feed particles disperse horizontally through the side net rather than vertically through the bottom net. Due to the high density of caged fish, which were vertically distributed inside the cages, the amount of feed particles dispersed through the side net could be larger than through the bottom net. In other words, the concentrations of waste feed particles in near surface waters around the cages were likely higher than beneath the cages. It is however not clear if the aggregated fish in surface waters around the cages also migrated to areas beneath the cages after feeding periods or not. Some fish were observed feeding on the sediments below the cages. Nocturnal feeding fishes such as catfishes may also be involved in reducing waste accumulation below the cages. A nocturnal *Anguilla* sp. was once found and caught while hiding in a hollow metal deployed near fish cage #1 to anchor a chain of tidbit temperature loggers in 14 m deep waters. The fish feeding on the sediments below the cages help to reduce the impact of fish farms on the lake bed by mixing, oxygenating and resuspending sediments and enhancing waste dispersal (Dempster et al., 2005).

Since Lake Malawi is characterized by low food abundance due to its oligotrophic status at algal biomass of <1 µg Chl*a* L^{-1}, high dependence of wild fishes and other organisms on cage wastes may be expected. Sandy shore fish species in the lake have been described from gut content and stable isotope analyses as opportunistic feeders which, despite their feeding specializations, utilize food resources that are readily available at any given time period (Duponchelle et al., 2005). This ability to switch diets occurs in Lake Malawi fish species

because they have evolved an astonishing diversity of feeding adaptations which enable them to exploit a wide array of food resources in the lake (McKaye & Marsh, 1983). Under such circumstances a relatively high proportion of the cage wastes would be consumed by the wild fishes and other organisms around the cages as already shown in fish farms in various water bodies. Mazzola and Sarà (2001) studied the utilization of cage-derived particulate wastes by mussels (*Mytilus galloprovincialis*) and clams (*Tapes* sp) cultivated around an intensive cage culture of European seabass (*Dicentrarchus labrax*) and Gilthead seabream (*Sparus aurata*) in the Gulf of Gaeta, Mediterranean Sea. The study showed, using stable isotopic analysis, that cage-derived organic matter was the dominant and constant source of organic C for both bivalves, accounting for about 80% of their diet. Grey et al. (2004) studied a trout farm in Esthwaite Water (UK) and calculated that cage-derived organic C constituted 45-50% of copepod diet, 62-68% of *Daphnia* diet, 73-89% of roach diet and 54-57% of chironomid diet. In the Mediterranean Sea, Vita et al. (2004) showed that 80% of the particulate organic matter that sank below fish cages was consumed by wild fish within the first 4 m while Phillips et al. (1985) reported gut contents of native fishes around a salmonid cage farm that consisted of 98% fish food. Waste consumption by wild fish in Lake Malawi has been confirmed using stable isotopes in Chapter 4 and may be significant as indicated by the disproportionately low concentrations of POM and particulate nutrients sedimenting below fish cages compared to discharge estimates from mass balance models (Chapter 2). As indicated earlier, only 0.6% of the P estimated by the mass balance model to have been discharged into the environment was empirically measured in particulate material collected in sediment traps deployed under fish cages. Part of the loss may

also be attributed to dispersion by the strong water currents through the cages as discussed above.

The consumption of cage wastes by wild fishes and other organisms around the cages significantly modifies the dispersion of wastes beneath and around the cages by increasing their area of deposition as wild fish move and defecate around the fish farm (Sarà et al., 2004). Consumption also reduces the effects of the organic wastes by decreasing through digestion the nutritional quality of the organic wastes before they reach the lake bottom (Vita et al., 2004). The area of deposition could be much wider in Lake Malawi as a significant proportion of the fishes observed around the cages seem not to permanently reside around the cages as indicated by movement of the wild fishes to fish cages during feeding hours, particularly in the morning (pers. observation). Both consumption and swimming activities around the cages slow down the settling speed of the waste particles (Sarà et al., 2004).

It has been argued that it requires the incorporation of ecological perspectives into management policies to establish viable and long-term solutions to problems in cage culture (Goldberg & Naylor, 2005). These ecological perspectives include the overlooked role of wild fishes around the fish farms in reducing impacts of cage operations. Since not much is known yet about these wild fishes around the cages in Lake Malawi, I suggest that studies should be done to understand the fish aggregations and their interactions with the cages so that they can be successfully maintained around the cages for the benefit of the environment. Some of the management approaches needed to maintain a high abundance of wild fish species around the cages include protecting fish farms from intense artisanal and commercial fishing activities and

deploying artificial reefs under the cages to attract and retain demersal fishes. While pelagic wild fish attracted by floating cages clean the water column around the cages, carefully sited artificial reefs could be deployed under the fish farm to increase the abundance of demersal fishes to remove the feed and fecal particles settling on the lake bed from the cages. A similar tactic has been used under fish farms in the coast of Israel where artificial reefs have been used to increase the abundance and role of demersal fish species to reduce the benthic impact of the fish farms (Angel et al., 2002).

3.6 Conclusion

Data presented in this chapter show that impacts of cage culture on the water column in the vicinity of the cages in Lake Malawi in 2007 were minimal. The observed minimal impacts of the cage wastes in 2007 could be due to (1) low total fish production in the cages of only ~200 tonnes, (2) rapid dispersion of cage wastes by persistent strong water currents which averaged 9.3 cm s^{-1} through the cages, and (3) high consumption of cage wastes by wild fishes which aggregated around the cages. These results support an earlier suggestion in chapter 2 that mass balance techniques are better than the empirical measurements used in this chapter for analyzing nutrient losses from fish cages into the environment in Lake Malawi. Production data recording and management by cage operators should, therefore, be improved and enforced to enable accurate estimation of nutrient losses from fish cages in the lake. As cage production continues and expands, waste generation in fish cages may increase and significantly pollute the surrounding environment as well as the reared fish. It is the responsibility of the Malawi

Department of Fisheries, fish farm operators and other stakeholders to put together a simple monitoring program for water quality in the vicinity of the fish cages as production expands. Water quality parameters of special importance to reared fish include dissolved oxygen and algal biomass, as chlorophyll concentration or chlorophyll fluorescence, which can be easily and cheaply monitored using a Sea Bird CTD already available at the Fisheries Research Institute at Monkey Bay. A significant linear relationship between extracted chlorophyll concentration and chlorophyll fluorescence has been determined in this study (Fig. 3.17) and can be used for the monitoring program. Because the N:P molar ratio of the fish feed supplied is low at 14.2, it would also be useful to monitor algal composition changes in the vicinity of the cages. Large losses of the fish feed would make the environment N deficient (relative to P availability) and would favor proliferation of N_2-fixing algal species under calm weather conditions. N_2-fixing algal species usually form blooms which can deplete dissolved oxygen during senescence.

It is important that cage operators should improve the quality and management of the fish feed supplied to the reared fish in order to reduce waste generation and discharge into the surrounding environment. High feed quality will increase feed assimilation by the caged fish while feeding management will reduce feed losses from the cages which will eventually reduce environmental impacts of cage operations.

It is clear that wild fish species played a significant role in reducing environmental impacts of cage wastes in Lake Malawi. More research should therefore be done to clearly understand these aggregations so that more wild fish can be attracted and retained at the fish farm to reduce potential environmental impacts of the cage wastes.

Chapter 4

Tracing the flux of cage culture organic wastes in the southeast arm (SE Arm) of Lake Malawi using carbon and nitrogen stable isotopes.

4.1 Abstract

The use of $\delta^{13}C$ and $\delta^{15}N$ analysis showed that cage wastes were incorporated in the food web that supported the wild fishes in the vicinity of the cages as expected from visual observations and studies elsewhere. This was indicated by comparing $\delta^{13}C$ and $\delta^{15}N$ signals of caged and wild fish caught in 2007 in the vicinity of the fish farm and signals of control fish samples caught between 1995 and 1997 before the fish farm was started in 2004. Accumulation of cage wastes in sediments below the fish cages were also confirmed using the $\delta^{13}C$ and $\delta^{15}N$ ratios. The accumulation of the cage wastes in the sediments was, however, minimal as indicated by the small differences in the ratios in sediments and some sedimentary organisms (bivalves, snails and worms) below the cages relative to ratios at control stations. This was expected from the rapid and efficient dispersion of the cage wastes facilitated by water currents through the fish farm which averaged 9.3 cm s^{-1} and consumption and dispersion of cage wastes by the large numbers of wild fishes which aggregated around the cages.

4.2 Introduction

Lake Malawi is an extremely important source of clean water for the people in the three riparian countries of Malawi, Tanzania and Mozambique. The high quality water supports an estimated 700-1,000 species of fish, whose diversity, breeding and feeding behavior fascinates the scientific community. As a result, Lake Malawi is one of the world's most important hotspots of biodiversity. The capture fishery has however declined in the lake during the last two decades due to overfishing and habitat degradation (Banda et al., 2005). In an effort to enhance fish production in the lake, Maldeco Aquaculture Ltd introduced cage culture in 2004 of indigenous *Oreochromis* spp. in the southeast arm of the lake. The cage introduction has raised two fundamental questions about the health of the Lake Malawi ecosystem as follows: 1) How will cage culture and associated waste products impact water quality in the southeast arm of Lake Malawi? and 2) How will cage culture and associated waste products affect endemic fish populations?

These questions were based on research findings done in other water bodies that showed that between 70 and 80% of nutrients added to fish cages through feed are lost to the surrounding environment (water column and underlying sediments) in the form of unconsumed feed, fish feces and metabolic wastes (Gowen & Bradbury, 1987; Folke & Kautsky, 1989; Holby & Hall, 1991; Hall et al., 1990, 1992; Troell, 1996; Kaushik, 1998). The discharged wastes, rich in nitrogen (N) and phosphorus (P), have the potential to pollute the surrounding waters and underlying sediments, and may cause problems including eutrophication, toxic algae outbreaks, increased turbidity, decreased oxygen, and loss of biodiversity.

Apart from undermining the quality of the water and sediments close to the cages, the particulate organic wastes discharged from fish cages can be directly utilized by animal consumers in the receiving ecosystem (Roditi et al., 2000). The uptake of the cage-derived organic matter may lead to the modification of planktonic and benthic food webs in the receiving ecosystem (Grey et al., 2004). Mazzola and Sarà (2001) studied cultivated mussels (*Mytilus galloprovincialis*) and clams (*Tapes* sp.) around an intensive fish farm of *Dicentrarchus labrax* and *Sparus aurata* in the Gulf of Gaeta, Mediterranean Sea. They showed that the cage-derived organic matter was the dominant and constant source of organic C for both mussels and clams, accounting for about 80% of the clam diet. Grey et al (2004) studied a trout farm in Esthwaite Water (UK) and calculated that cage-derived organic C constituted 45-50% of copepod diet, 62-68% of *Daphnia* diet, 73-89% of roach diet and 54-57% of chironomid diet. In the Mediterranean Sea, Vita et al. (2004) showed that 80% of the particulate organic matter that sank below fish cages was consumed by wild fish within the first 4 m while Phillips et al. (1985) reported that gut contents of native fishes around a salmonid cage farm consisted of 98% fish food. The consumption by native fishes of the cage wastes may also help to mitigate environmental impacts of the cage operations as discussed in Chapter 2, but may lead to alteration of fish distributions and habitat relationships.

In Lake Malawi, fish species have evolved an astonishing diversity of feeding adaptations which enable them to exploit a wide array of foods including phytoplankton, zooplankton, detritus, epilithic and epiphytic algae, macrophytes, mollusks, insects, benthic invertebrates, whole fish, fish scales, fish eggs and fish larvae (McKaye & Marsh, 1983). Duponchelle et al.

(2005) studied feeding behavior of sandy shore fish species in the lake using gut content and stable isotope analyses of fish tissues and observed large overlaps of diet regardless of species feeding specializations. The overlaps in diets suggest that the fish species on sandy bottoms were utilizing rather similar mixed diets, which were changing opportunistically with the availability of the food sources. Since the abundance of autochthonous food resources in the lake is low at <2 μg L^{-1} chlorophyll concentration (Chapter 2; Guildford et al., 2000, 2007; Gondwe et al., 2008), the opportunistic fish would be expected to rapidly change their diet to take advantage of the organic wastes discharged from fish cages.

The modifications in food webs resulting from the utilization of the cage-derived organic matter may be traced by the analyses of stable isotopes of C and N in the food webs and in the organic wastes, especially if the fish feed has a different isotopic composition from the autochthonous prey organisms. It has been shown that the stable isotopic signatures ($\delta^{13}C$ and $\delta^{15}N$) of food organisms are conveyed up through food webs in a reasonably predictable manner, such that the signatures of an organism, including fish, closely reflect the stable isotopic composition of their diets with enrichment of the isotopic signatures with each transfer (Post, 2002). The $\delta^{13}C$ and $\delta^{15}N$ signatures are slightly (0-1‰) and significantly enriched (3-5‰, mean of 3.4‰) respectively in an organism compared to its diet. Determination of isotopic $\delta^{13}C$ and $\delta^{15}N$ signatures of a consumer and its potential food sources should, therefore, allow trophic links and the basal resources commonly consumed and assimilated by a consumer to be identified (Hecky & Hesslein, 1995; Grey et al., 2001). It is also possible to quantify the fractional contribution of elemental mass from each food source to a consumer's diet by using

stable isotopic values in linear mass balance mixing models (Phillips & Koch, 2002; Campbell et al., 2005). Partitioning of food sources in a consumer's diet is possible because the C and N isotopic values in a consumer are weighted mixtures which are proportional to the amounts of the food sources assimilated assuming different food sources have similar assimilation efficiencies. Similar approaches can be used to partition organic matter sources in the sediments. Partitioning of two food sources using a mass balance mixing model would require determination of isotopic signatures of only one element (e.g., $\delta^{13}C$ or $\delta^{15}N$) while partitioning of three major food sources requires simultaneous use of isotopic values of two elements (e.g., $\delta^{13}C$ and $\delta^{15}N$). Food web studies were previously done by conventional gut content analysis but recent developments in equipment and procedures for analysis of stable isotopes have increased the applications of stable isotope analysis in food web studies (Hobson & Welch, 1992). Stable isotope analysis offers a number of advantages over the conventional gut content analysis because 1) it analyzes stable isotope compositions of assimilated diets as opposed to ingested materials (Kling et al., 1992), 2) it incorporates dietary information over longer time periods as opposed to momentary snapshots (Fry & Arnold, 1982), 3) it may identify sources that may not be detected by inspection of ingested materials (Grey et al., 2001) and 4) it is easier (less time consuming) than gut content analysis.

In this study, $\delta^{13}C$ and $\delta^{15}N$ isotopic ratios were utilized to trace the flux of cage-derived organic wastes into the planktonic and benthic food webs in the southeast arm of Lake Malawi. This is important to define the potential impact of cage culture to food webs in an aquatic ecosystem. Stable isotopes have been previously used in Lake Malawi to define trophic

interrelationships among fishes and their food resources (Bootsma et al 1996; Hecky & Hesslein 1995; Kidd et al 2003). These previous studies provide useful baseline data on expected trophic relationships in Lake Malawi for determining possible changes imposed by cage culture operations.

4.3 Materials and Methods

4.3.1 Study site.

The study site, including sampling stations on the study transect through the fish farm, has been described in details in Chapters 2 and 3 (Figs. 1.1 and 3.1 respectively). In brief, the fish farm under study was established in 2004 by Maldeco Aquaculture Ltd to raise indigenous *Oreochromis karongae* (Chambo) in the southeast arm of Lake Malawi. Later, indigenous *Oreochromis shiranus* (Makumba) was also stocked in mixed cages with *O. karongae* or in separate cages. During the study period, the number of deployed fish cages in the lake increased from 16 to 48 but due to lack of enough fingerlings only about half of the cages were stocked. Fish were fed three pelleted diets, starter, grower and finisher, in that order from the day the fingerlings were stocked to harvest. Starter diet was supplied until the fish weighed 50-70 g, grower was supplied until the fish weighed 150-200g while finisher was supplied until the fish reached market size of ≥300g (Maldeco Aquaculture Ltd, pers. comm.). Each cage was targeted to produce about 20 tonnes (20,000 kg) of fresh fish per year. The fish farm is located about 1

km from shore in 14-24 m of water at an inshore-offshore orientation perpendicular to the main wind and water current directions.

4.3.2 Sampling.

Sample collection for this study was done between January and June and also between September and December 2007. Samples included formulated fish feed diets (starter, grower and finisher), cage wastes (from sediment traps), fish feces, caged fish, wild fish from the vicinity of the cages, periphytic algae growing on cage nets, phytoplankton, zooplankton, sediments, benthic worms, bivalves and snails.

The three feed diets were obtained from Maldeco Aquaculture Ltd in January, June, October and December 2007. Feces produced by caged fish consuming starter, grower and finisher feed diets were collected from the bottom of a container after a 4 hr nutrient regeneration experiment (which will be reported elsewhere) involving caged fish. Periphytic algae were sampled by hand from the cage nets with forceps. Sampling was done between October and December 2007 from 8 cage nets after I observed that caged fish were frequently grazing the periphytic algae.

Sampling for phytoplankton, zooplankton, bivalve species, benthic worms, snails and sediments was done at the fish farm (at KGC station and around fish cages) as well as at control stations along the study transect. Control stations that were sampled were CS2 and CS4 located 1.7 and 4.4 km respectively south of the cages and CN2 and CN4 located north of the cages at 2.3 and 6.2 km, respectively (Table 3.1). I collected vertical net hauls from 15-0 m at KGC, CS2

and CN4 and 13-0 m at CS4 and CN2 using a 10 µm Nitex Wiscosin net for phytoplankton and a 63 µm Nitex Wisconsin net for zooplankton. Hauls made around cages were either from 13-0 m or 15-0 m depending on the water depth at that particular cage. Both nets were 130 mm in mouth diameter. At each station, two and three hauls were combined for phytoplankton and zooplankton respectively. Samples from phytoplankton nets did not contain appreciable numbers of zooplankton because the hauls were collected slowly to avoid clogging the net and also to allow zooplankton in the path of the net to escape. In contrast zooplankton hauls were done faster and contained significant amount of filamentous phytoplankton which were probably *Aulacoseira* which dominate the algal community in the southeast arm (Hecky & Kling, 1987; FAO, 1993). The phytoplankton filaments were removed from zooplankton samples by filtering through 100, 120, 163 and 250 µm sieves. Most zooplankton were observed between 100 and 163 µm sieves. Filtered zooplankton samples were recombined and then filtered onto Whatman QMA filter papers for $\delta^{13}C$ and $\delta^{15}N$ analyses.

Bivalves, snails and benthic worms were sieved from sediment samples collected with a ponar grab sampler (19.9cm x 19.4cm). Similar organisms at each station were pooled together and left in lake water over night to allow them to evacuate their guts after which they were frozen in plastic bags. Benthic worms were filtered onto Whatman QMA filter papers before freezing for analysis at a later date. Sediment samples for stable isotope analyses in most studies (e.g. Duponchelle et al., 2000) have been taken from the upper layer of grab samples. In this study subsamples were collected after resuspending the ponar grab samples in lake water because the grab samples were already mixed since they contained more water than the sandy

sediments at all stations except at CN2 where sediments were consistently finer grained. After resuspension, coarse material was allowed to settle out in approximately 1 minute before subsamples were collected. The assumption was that the organic material, including the waste feed and fish feces, was lighter so that it remained in suspension longer than the non-organic coarse material. The subsample was allowed to settle in a container over several hours before the supernatant water was decanted by siphoning with a plastic syringe. Where turbidity after resuspension was low, subsamples were filtered immediately onto Whatman QMA filter papers for stable isotope analysis. At the fish farm, sediment samples were collected at the center of the farm (KGC station) as well as around the floating cages. However, since the cages floated from side to side with wind, the vicinities sampled could reflect material falling out of the cages depending on when sampling was done.

Particulate wastes falling from the cages were collected in base-weighted sediment traps suspended below the cages with ropes tied at adjacent cages and/or buoys to keep the traps in constant reception of the falling particles as the cages drifted from side to side with wind. At control stations (where no cages were available) sediment traps were attached to a tight rope stretched between an anchor at the bottom and a buoy just below the water surface and their GPS positions were always recorded. Sediment traps at control stations were located at almost the same depth as traps under the cages (about 7m below surface). Subsamples of the sediment trap samples were filtered through Whatman QMA filter papers for $\delta^{13}C$ and $\delta^{15}N$ analyses. Data presented in Chapter 3 (Table 3.3) show that the mean sedimentation rate for particulate material was on average 13 times higher under the cages than at control stations. Sedimentation rates

under the cages were variable, but generally higher under cages containing large fish than under cages stocked with small and medium-sized fish.

Samples (n = 27) of caged fish ranging from 14.3 to 27.8 cm in total length (TL), weighing 57.4-408.9 g (Table 4.3) were donated by Maldeco Aquaculture Ltd, while wild fish were caught around the fish cages with hand nets by the feeding crew, bought from fishermen fishing around the fish farm, and caught by trawling around the fish farm and between cage rows (≤ 100 m apart) using the *R.V. Ndunduma* during a January 2007 survey conducted by the Malawi Department of Fisheries Research Institute at Monkey Bay. The survey was done to assess the abundance and diversity of wild fish species at the fish farm. All the caged fish samples analyzed were either on grower or finisher feed diets. All fish samples collected from the cages, except three, were >100g. The dorsal white muscle was dissected for analysis from one to five fishes of each species while small fish species such as *Engraulicypris sardella* were processed whole.

All samples were dried to constant weight at 50^{o}C oven temperature for 48 h. While phytoplankton, zooplankton and benthic invertebrate samples were analyzed for $\delta^{13}C$ and $\delta^{15}N$ on Whatman QMA filters, fish feeds, fish dorsal muscles, bivalve tissues, snail tissues and sediments were pulverized to a fine powder first using a Retsch MM 2000 ball mill grinder and sealed (4 mg sediments under the cages, 6 mg sediments away from cages and between 0.25-0.30 mg for feed, fish, fish feces, snail and bivalve samples) in tin cups for analysis at the Environmental Isotope Laboratory at the University of Waterloo, Canada. The carbon and nitrogen isotope ratios were analyzed on a Delta Plus Continuous Flow Stable Isotope Ratio

Mass Spectrometer (Thermo Finnigan / Bremen-Germany) coupled to a Carlo Erba Elemental Analyzer (CHNS-O EA1108-Italy).

Stable isotope results of samples are expressed in delta (δ) notation in per mil (‰) deviations from concentrations in international standards according to the following equation:

$$X = [(R_{sample} - R_{standard})/R_{standard}] \times 1000$$

where X is $\delta^{13}C$ or $\delta^{15}N$, R_{sample} is the $^{13}C/^{12}C$ or $^{15}N/^{14}N$ ratio of sample and $R_{standard}$ is the $^{13}C/^{12}C$ or $^{15}N/^{14}N$ ratio of standard, Peedee Belemnite formation for C and atmospheric nitrogen for N. More positive values are isotopically enriched meaning that they contain proportionally higher concentrations of the heavier ^{13}C or ^{15}N isotope (Vander Zanden et al., 2006).

Kidd et al (2003) sampled 40 fish species between 1996 and 1997 for their mercury (Hg) bioaccumulation study in Lake Malawi, 7 species of which were also collected around the fish farm in 2007 for the current study. Because I did not have fish species from control stations to compare $\delta^{13}C$ and $\delta^{15}N$ values with the fish species which aggregated at the fish farm, I used $\delta^{13}C$ and $\delta^{15}N$ values of 6 fish species studied by Hecky and Hesslein (1995) and 7 fish species studied by Kidd et al (2003) as control data. These fish species were collected from Lake Malawi between 1995 and 1997, which is 7-9 years before the establishment of the fish cages in 2004. This comparison allowed me to determine whether the fish species which were caught around the fish farm were feeding on the cage wastes or not.

Statistical analysis of the data was done using Student's t-test for paired and unpaired values. The significance level was set at $p<0.05$.

4.4 Results

The mean $\delta^{13}C$ values of starter, grower and finisher diets were -25.15, -22.90 and -21.87‰ respectively (Table 4.1) which indicated significant ($p \leq 0.001$) depletion for starter diet compared to the other two diets. The mean $\delta^{15}N$ values for the three diets, 2.70‰, 3.07‰ and 2.83‰ for starter, grower and finisher, respectively (Table 4.1), were not, however, significantly different from each other ($p > 0.05$). Because the fish samples were collected from grower and finisher cages only, mean (±SE) $\delta^{13}C$ and $\delta^{15}N$ of feed of -22.39±0.35 and 2.95±0.14‰ were calculated from grower and finisher diets only. In addition, starter diet was replaced by grower and later finisher when fingerlings reached between 50 and 70 g body wet weight so that tissue $\delta^{13}C$ and $\delta^{15}N$ reflecting the starter diet were quickly diluted as fish biomass increased to market size. The starter diet was only 7-32% (mean of 20%) of the total feed weight used for each production cycle.

A total of 27 caged fish were sampled from grower and finisher cages and analyzed for C and N isotopic ratios. The caged fish were significantly enriched in both $\delta^{13}C$ ($p=0.04$) and $\delta^{15}N$ ($p<<0.0001$) compared to the feed materials, with differences of 0.85‰ in $\delta^{13}C$ and 3.43‰ in $\delta^{15}N$ (Fig. 4.1; Table 4.1). The observed trophic shift of 0.85‰ in caged fish is within the expected range of 0-1‰ for $\delta^{13}C$ between diet and consumer. In contrast, periphytic algae available to caged fishes on cage nets were very enriched in $\delta^{13}C$ at -15.88‰ but significantly depleted in $\delta^{15}N$ at 1.51‰ compared to feed materials and caged fish $\delta^{13}C$ and $\delta^{15}N$ values (Fig. 4.1; Table 4.1; $p<0.0001$).

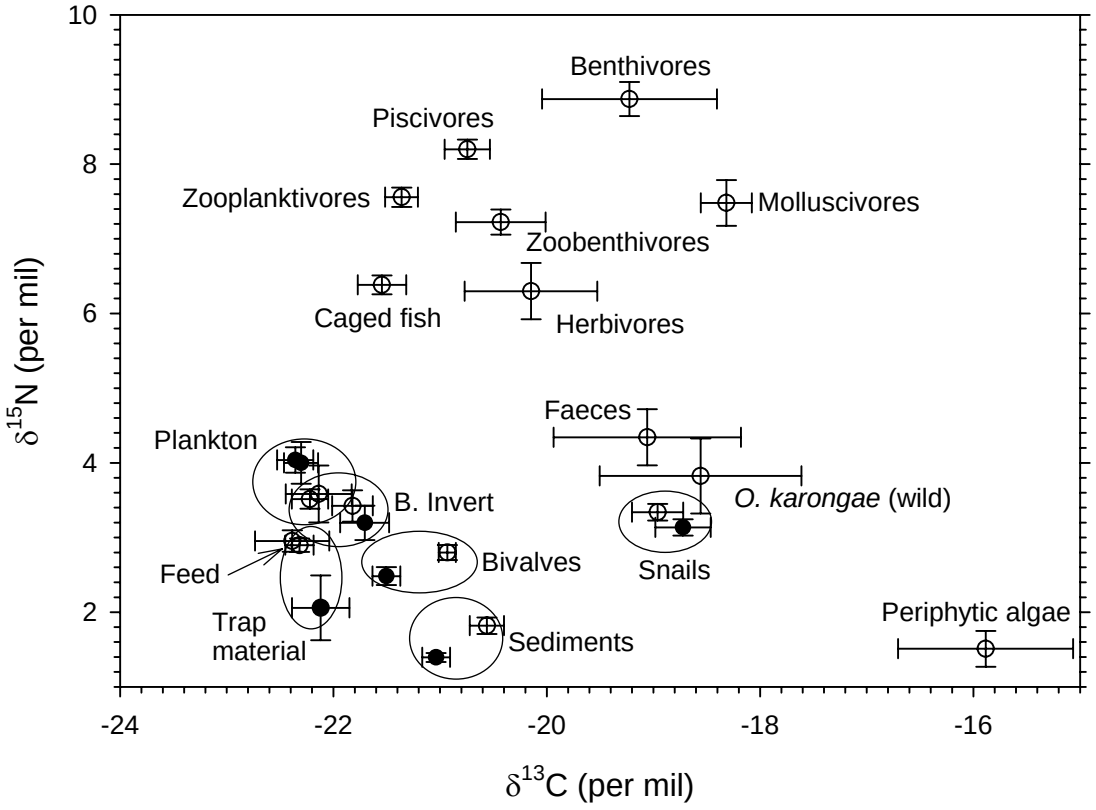


Fig. 4.1. Relationship between mean $\delta^{13}C$ and $\delta^{15}N$ values (‰) for fish feed, cage wastes (trap material), plankton material (zooplankton and phytoplankton), sediments and sedimentary organisms (benthic invertebrates, bivalves and snails), fouling algae, caged fish and their feces, and different trophic guilds of wild fishes caught in the vicinity of the fish farm. Samples were collected at the fish farm (open symbols) and at control sites (closed symbols). Larger circles indicate similar faunal elements that were sampled at the cage site and at control stations away from the cage site. Bars in figure represent standard errors (SE).

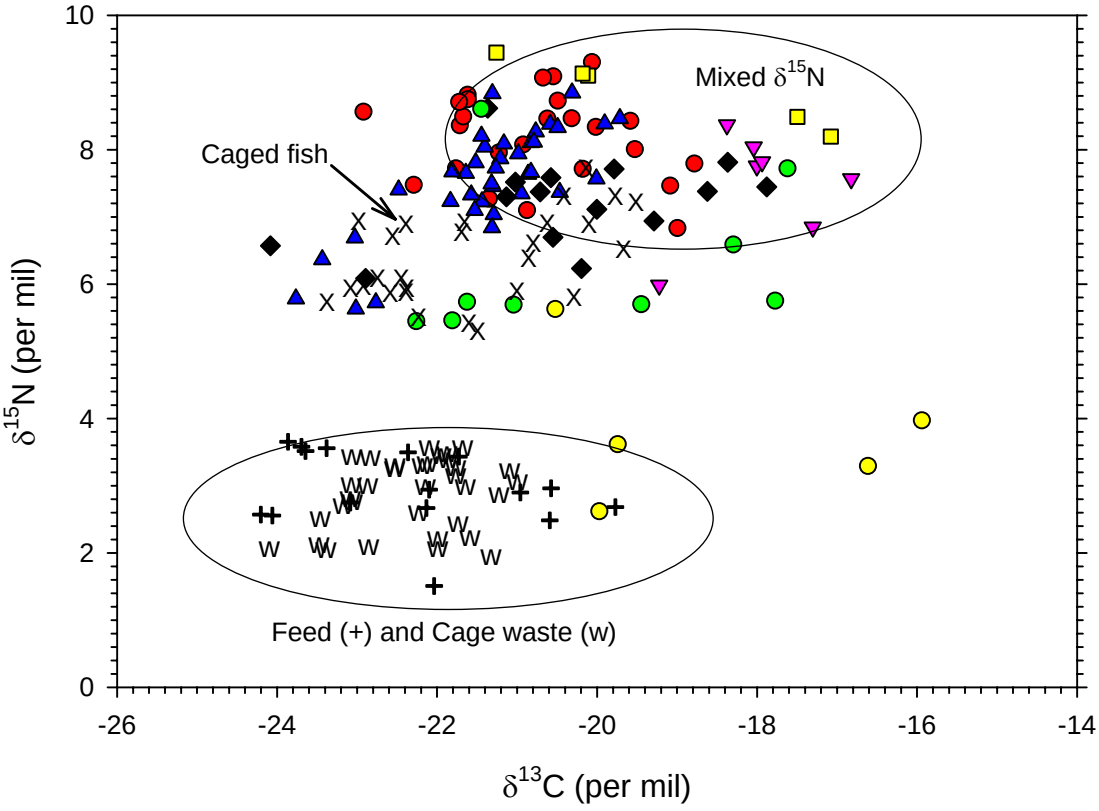


Fig. 4.2. Relationship between $\delta^{13}C$ and $\delta^{15}N$ values (‰) for feed (+), cage wastes (W), caged fish (X), and wild fishes – herbivorous (green circles), wild *O. karongae* (yellow circles**),** zooplanktivorous (blue triangles), zoobenthivores (black diamonds), molluscivores (pink triangles), piscivores (red circles), and benthivores (yellow squares). The species in the feeding groups were as follows: herbivores = *H. oxyrhynchus, O. karongae* and *P. similis;* zooplanktivores = *C. chrysonotus, C. eucinostomus, C. virginalis, D. limnothrissa* and *E. sardella*; zoobenthivores = *C. intermedius, C. pictus, F. rostratus, M. lateristriga, T. microstoma, P. elegans, P. subocularis, P. triaenodon* and *N. argyrosoma*.; molluscivore = *T. placodon*; benthivores = *A. calliptera* and *L. macrochir*; piscivores = *B. atritaeniatus, B. heterotaenia, B. nototaenia, C. caeruleus, N. livingstoni, N. polystigma, O. microlepis, S. woodi* and *Rhamphochromis* spp. For full scientific names of the wild fish species refer to Table 3.2.

Although the caged fish were frequently observed feeding on the periphytic algae on cage nets, the stable isotope data suggest the fish rely more strongly on the feed rather than on the periphytic algae. The contribution of phytoplankton to the total food supply for the caged fish was also assumed to be negligible due to the high fish density in the cages and the low standing phytoplankton biomass in the water column (annual mean extracted chlorophyll concentration of about 1.0 μg L^{-1}; Table 3.1 Chapter 3). The difference in $\delta^{15}N$ value of 3.43‰ between feed and caged fish shows that phytoplankton, at 0.04‰, was indeed unimportant in the diet of the caged fish. The $\delta^{13}C$ values of feces from caged fish tracked $\delta^{13}C$ values of the feed the caged fish consumed (Table 4.1). Feces were also significantly enriched by 3.33 and 1.39‰ in $\delta^{13}C$ and $\delta^{15}N$, respectively, compared to the feed ($p=0.0007$) (Fig. 4.1). As for the feed, the mean $\delta^{13}C$ and $\delta^{15}N$ values for the feces were calculated using signals in feces of medium and large caged fish consuming grower and finisher diets respectively.

The $\delta^{13}C$ and $\delta^{15}N$ values of particulate material collected in sediment traps deployed under the cages and at control stations indicated more significant enrichment ($p=0.004$) in $\delta^{15}N$ by 0.84‰ under the cages than at control stations, but no difference in $\delta^{13}C$ between the stations ($p=0.53$) was observed. The particulate material collected in sediment traps under cages was not significantly different from the feed material both in $\delta^{13}C$ and $\delta^{15}N$ ($p\geq0.73$) (Fig. 4.1). In contrast, $\delta^{13}C$ and $\delta^{15}N$ values were significantly heavier in fish feces than in the particulate material collected under the cages at $p=0.0009$ and $p<<0.0001$ respectively.

A total of 31 wild fish species were caught and analyzed for C and N isotopic ratios. In terms of numbers of species, it is clear that the wild fish community at the fish farm was

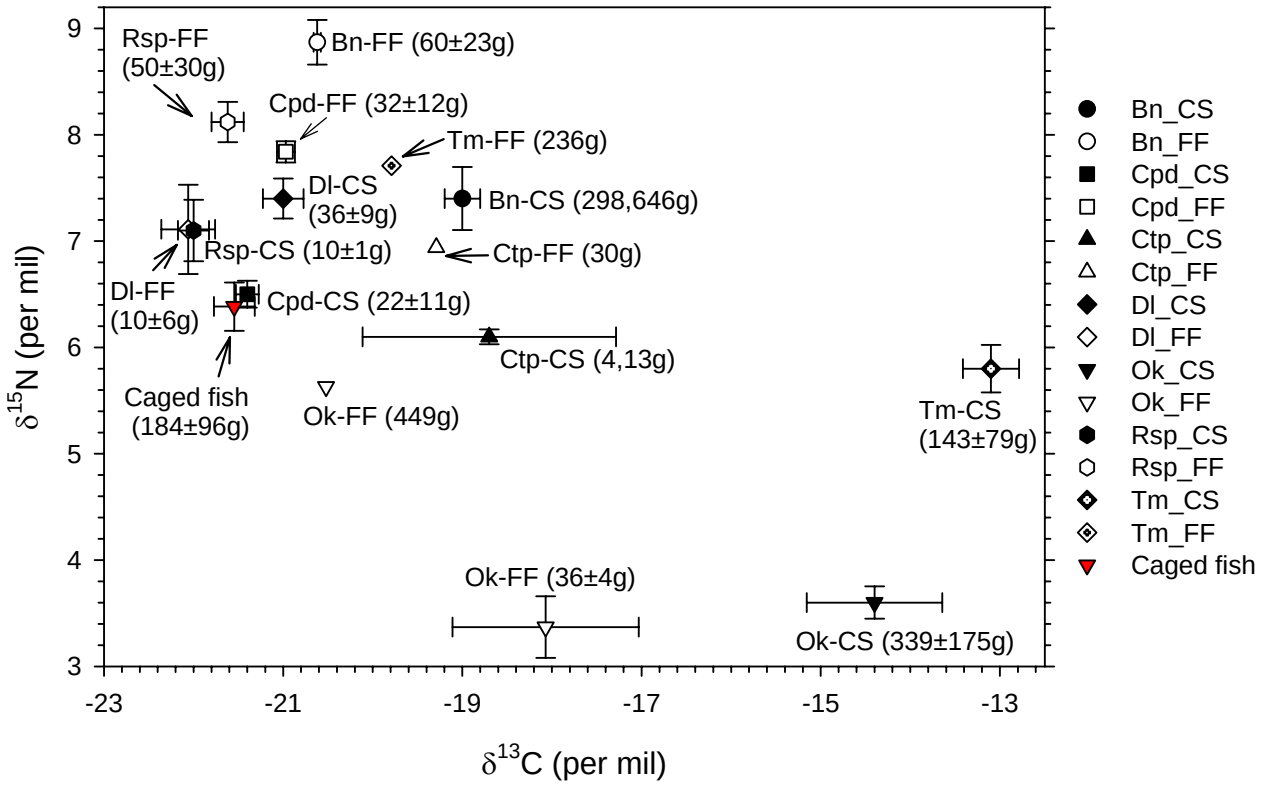


Fig. 4.3. Comparison of mean (±**SE)** $\delta^{13}C$ and $\delta^{15}N$ values (‰) for wild fish species collected between 1996 and 1997 by Kidd et al (2003) and fish collected in the vicinity of the fish farm (FF) in 2007. The open symbols represents fish samples collected in the vicinity of the fish farm (FF) while dark symbols represents samples collected by Kidd et al (2003) referred to here as control samples (CS). Bn = *Buccochromis nototaenia*, Cpd = *Copadichromis* spp., Ctp = *Ctenopharynx pictus*, Dl = *Diplotaxodon limnothrissa*, Ok = *Oreochromis karongae*, Rsp = *Ramphochromis* spp., and Tm = *Trematocranus microstoma*. The numbers in parentheses are mean fish body weights (g) ±SD.

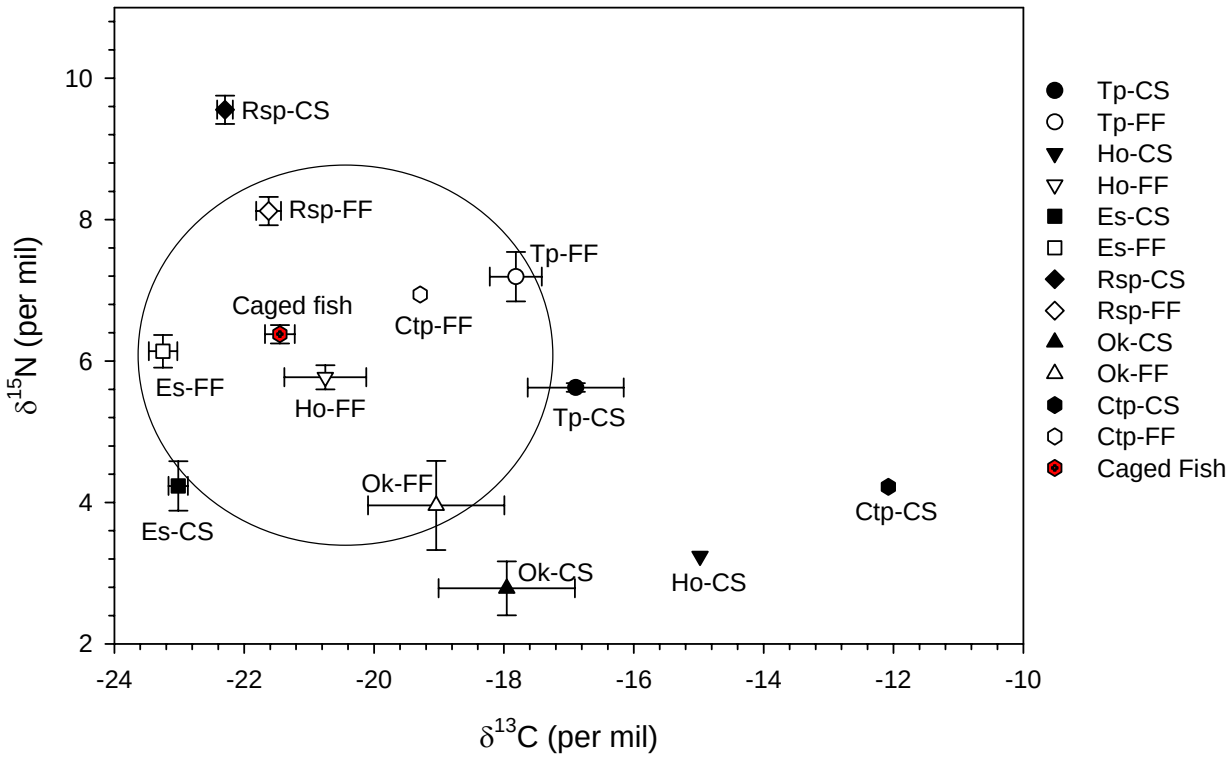


Fig. 4.4. Comparison of mean (±SE) $\delta^{13}C$ and $\delta^{15}N$ values (‰) for wild fish species collected in the vicinity of the fish farm (FF) in 2007and fish species analyzed by Hecky & Hesslein (1995). The open symbols represents fish samples collected in the vicinity of the fish farm (FF) while closed symbols represents samples analyzed by Hecky & Hesslein (1995) referred to here as control samples (CS). Tp = *Trematocranus placodon*, Ho = *Hemitilapia oxyrhynchus*, Es = *Engraulicypris sardella*, Rsp = *Ramphochromis* spp., Ok = *Oreochromis karongae*, and Ctp = *Ctenopharynx pictus*. Fish sizes have not been included because they were not provided in Hecky & Hesslein, 1995. The big circle indicates spread of signals of caged and wild fish collected from the fish farm in 2007.

dominated by piscivores (9 species), zooplanktivores (5 species) and zoobenthivores (9 species) which in their natural environment occupy trophic levels between 3 and 4.6 above the primary producers in the lake (Table 4.2). Only three algal eating (trophic level 2) species, *Hemitilapia oxyrhynchus, Protomelas similis* and *Oreochromis karongae*, were caught at the fish farm. The $\delta^{13}C$ and $\delta^{15}N$ values for wild fish caught at the fish farm varied broadly from -15.94 to -24.08‰ and from 2.62 to 9.44‰, respectively (Table 4.2). Some species such as *O. karongae, D. limnothrissa*, and *H. oxyrhynchus* showed significant intra-specific variability in both $\delta^{13}C$ and $\delta^{15}N$ values. For example, both *O. karongae* and *H. oxyrhynchus* covered a $\delta^{13}C$ range of 4‰ (Table 4.2). Most of the fish sampled were similar in size except for *O. karongae* where one fish was much larger than the other four (Table 4.2). The larger *O. karongae* was the lightest and heaviest in the group in $\delta^{13}C$ and $\delta^{15}N$, respectively (Fig. 4.3; Table 4.2). The mean $\delta^{15}N$ of *O. karongae* at 3.82‰, range of 2.62 to 5.45‰, completely isolated it from the rest of wild fish species analyzed. The mean $\delta^{15}N$ of caged *O. karongae* at 6.38‰ was more similar to the other wild fish species. The $\delta^{13}C$ and $\delta^{15}N$ values for *E. sardella* and *A. calliptera* obtained in this study are similar to values obtained by Bootsma et al (1996) for the same species. A general but small enrichment in $\delta^{15}N$ signals between trophic levels was observed in the fish species even though zooplankton was depleted relative to phytoplankton (Fig. 4.1). The phytoplankton-zooplankton signal may be due to improper sampling of the component of the phytoplankton assemblage that comprises the zooplankton diet as discussed below. The average difference in $\delta^{15}N$ between trophic level 3 (zooplanktivore, zoobenthivore and molluscivore) and 4 (piscivore) was 0.78‰ while that between trophic level 2 (herbivore) and trophic level 3 was 2.00‰.

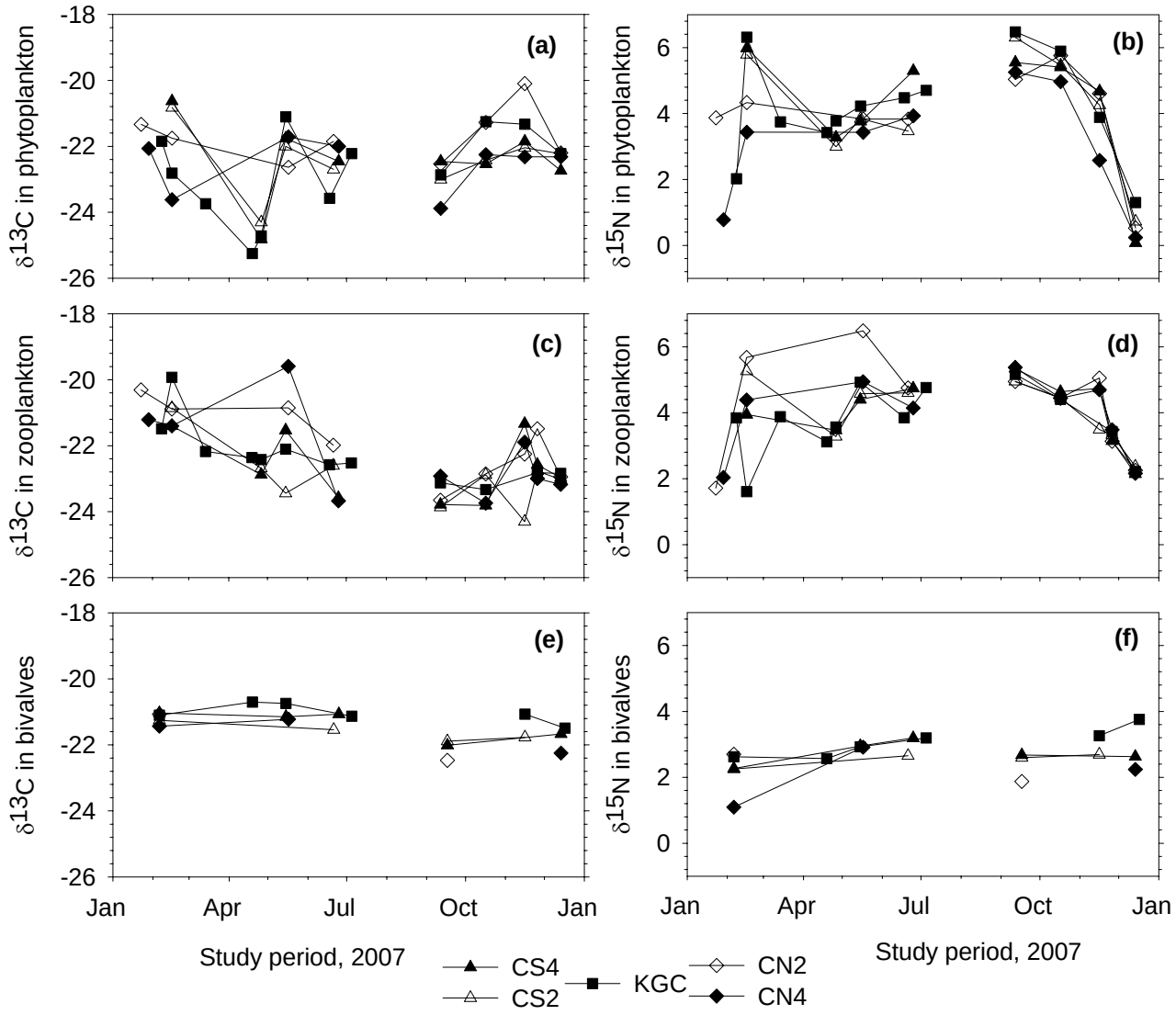


Fig. 4.5. Temporal and spatial patterns of $\delta^{13}C$ and $\delta^{15}N$ values (‰) for phytoplankton ((a) & (b)), zooplankton ((c) & (d)) and bivalves ((e) & (f)) at stations along the study transect.

The difference in $\delta^{15}N$ between trophic level 2 (herbivore) and trophic level 3 reduced to 1.19‰ if four *O. karongae* fish samples, which had low $\delta^{15}N$ of 3.37‰ compared to the rest of the herbivorous species (mean $\delta^{15}N$ of 6.23‰), were excluded from the analysis.

Figs. 4.3 & 4.4 show differences in both $\delta^{13}C$ and $\delta^{15}N$ values between fish samples collected between 1995 and 1997 by Hecky and Hesslein (1995) and Kidd et al (2003) (CS) and those collected in the vicinity of the fish farm (FF) during the course of this study in 2007. Some of the differences (e.g., between *Ctenopharynx pictus*: Ctp-CS (9g) and Ctp-FF (30g) in Fig. 4.3) may partly be related to ontogenetic shifts because of size differences in the collected fishes while other shifts (e.g., *Orechromis karongae*: Ok-CS (339g) and Ok-FF (449g) in Fig. 4.3) may be attributed to consumption of cage wastes as discussed below. Fish sizes were not given (except for *Oreochromis* spp.) in Hecky and Hesslein (1995).

The mean $\delta^{13}C$ and $\delta^{15}N$ values of phytoplankton sampled with a 10-µm Nitex Wisconsin net were –22.30±0.16 and 4.00±0.28‰ at the control stations and -22.14±0.31 and 3.58±0.38‰ at the fish farm. Phytoplankton samples showed no significant ($p>0.05$) enrichment in both $\delta^{13}C$ and $\delta^{15}N$ values between control stations and the fish farm. The mean $\delta^{13}C$ and $\delta^{15}N$ values of zooplankton sampled with a 63-µm Nitex Wisconsin net were –22.36±0.17 and 4.04±0.17‰ at the control stations and -21.82±0.19 and 3.42±0.21‰ at the fish farm. Zooplankton $\delta^{13}C$ values were slightly but significantly enriched at the fish farm compared to the control stations while $\delta^{15}N$ values were depleted ($p=0.03$) at the fish farm compared to control stations. In this study zooplankton were 2.14 and 0.84‰ heavier in $\delta^{13}C$ and $\delta^{15}N$, respectively, than values reported by Kidd et al (2003) which may be attributed to sampling site differences or seasonal effects.

Phytoplankton sampled with a 10-µm Nitex Wisconsin net was isotopically, both $\delta^{13}C$ and $\delta^{15}N$, similar to zooplankton ($p>0.05$) sampled with a 63-µm Nitex Wisconsin net both at the control stations and the fish farm. In addition, zooplankton was slightly depleted in $\delta^{15}N$ relative to phytoplankton at both the fish farm and control stations. The phytoplankton signals observed in this study are similar to values reported by Genner et al (2003) and are likely due to the presence of both phytoplankton and zooplankton in the sample even after screening to remove zooplankton. It may also be due to a presence of an isotopically enriched fraction of the phytoplankton community in the samples analyzed that was not part of the zooplankton diet. The plankton samples analyzed by Genner et al (2003) were collected using a 50-µm phytoplankton net but were not further separated into zooplankton and phytoplankton fractions. Kidd et al (2003) estimated the POM $\delta^{15}N$ signal at 0.04‰ which agrees with this study if an expected 3.4‰ trophic transfer value was subtracted from the zooplankton signals of 4.04‰ observed at control stations.

The $\delta^{13}C$ values of both phytoplankton and zooplankton at both the fish farm and control sites were not significantly different from the -22.39±0.35‰ mean signal in feed ($p>0.05$). The $\delta^{15}N$ for phytoplankton and zooplankton at control stations were significantly heavier than values in feed materials ($p\leq0.03$). At the fish farm, no significant differences were observed between the plankton and feed materials ($p\geq0.12$); however, it is likely that the material collected in the 10-µm net was actually mostly representative of zooplankton. According to Kidd et al (2003), the phytoplankton $\delta^{15}N$ is expected at ~0.04‰, and, therefore, would be significantly different from the feed signal after corrections.

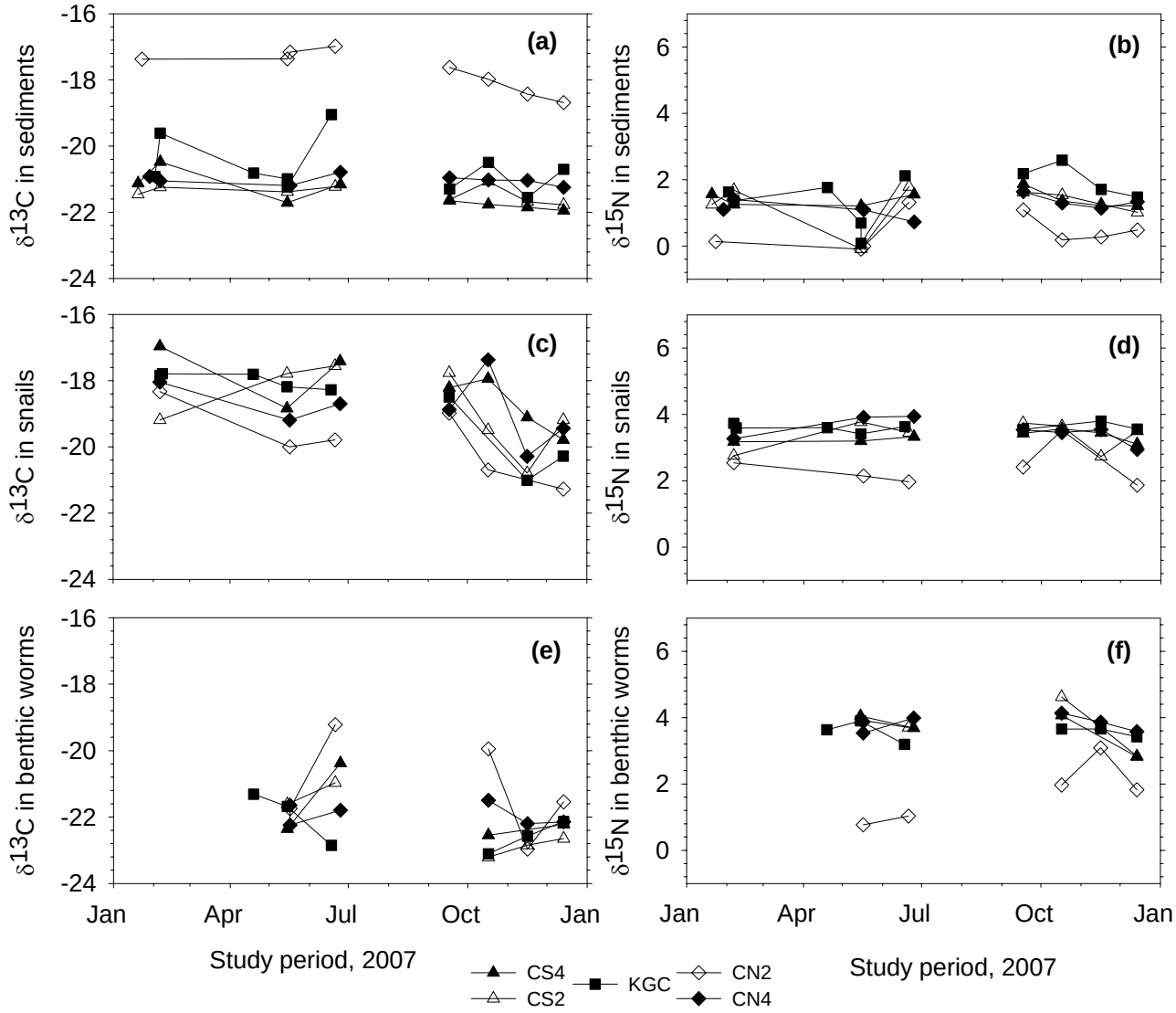


Fig. 4.6. Temporal and spatial patterns of $\delta^{13}C$ and $\delta^{15}N$ values (‰) for sediments ((a) & (b)), snails ((c) & (d)) and benthic worms ((e) & (f)) at stations along the study transect.

Between January and July, the zooplankton at southern control stations track ^{13}C of phytoplankton more closely than northern stations, while after September they all track together (Fig. 4.5a & c). The $\delta^{15}N$ values of phytoplankton and zooplankton declined between September and February (Fig. 4.5b & d, respectively) when the abundance of N_2-fixing cyanobacterial species is reported to be higher in the water column than during the rest of the year (Gondwe et al., 2008; Paterson & Kachinjika, 1993).

Sediments at the fish farm were significantly enriched in both $\delta^{13}C$ and $\delta^{15}N$ values ($p=0.03$ and $p=0.002$ respectively) compared with sediment samples from control stations (Fig. 4.1; Table4.1). The $\delta^{13}C$ values of sediments at station CN2 were significantly enriched compared with values at the other stations along the study transect (Fig. 4.6a). In contrast, $\delta^{15}N$ values at CN2 were depleted relative to values at the other stations (Fig. 4.6b). This station's sediments were also compositionally different (more clay) than the other stations likely reflecting the influence of a nearby tributary. Consequently $\delta^{13}C$ and $\delta^{15}N$ values of the sediments at CN2 were not included in calculating mean $\delta^{13}C$ and $\delta^{15}N$ values at control stations reported above. The temporal and spatial patterns also show that the KGC station was most often highest for both $\delta^{13}C$ and $\delta^{15}N$ compared to CS2, CS4 and CN4 stations.

Suspension feeding bivalves were significantly enriched in both $\delta^{13}C$ and $\delta^{15}N$ values ($p\leq0.05$) under the cages than at control stations (Fig. 4.1; Table 4.1). In contrast, snails and benthic worms collected from below the fish cages were not isotopically ($\delta^{13}C$ and $\delta^{15}N$) different from samples collected at control stations ($p>0.05$) (Fig. 4.1; Table 4.1). Snails had significantly enriched $\delta^{13}C$ values compared to bivalves and benthic worms ($p<0.05$) both at the

fish farm and at control stations (Fig. 4.1). On the other hand, bivalves had more enriched $\delta^{13}C$ values than benthic worms but only at the fish farm (Fig. 4.1). The $\delta^{15}N$ values for snails and benthic worms were significantly higher than values for bivalves sampled at both control and cage stations. However, $\delta^{15}N$ values for snails and benthic worms were not significantly different ($p>0.05$) from each other both at control and cage stations. The $\delta^{13}C$ values of snail species at all the five stations along the study transect declined between September and January 2007 from ~-18‰ to -20‰ (Fig. 4.6c). In contrast, $\delta^{15}N$ values remained relatively stable between 3 and 4‰ (Fig. 4.6d). Bivalves were not as numerous as snails in sediments, particularly at station CN2 where bivalves were observed during two visits only. The temporal and spatial patterns of $\delta^{13}C$ and $\delta^{15}N$ values for bivalve species were remarkably consistent and varied between -21 and -22‰ and about 2‰ respectively (Fig. 4.5e & f). The $\delta^{13}C$ values for benthic worms were more variable than $\delta^{15}N$ values (Fig. 4.6e & f respectively).

Table 4.1. $\delta^{13}C$ and $\delta^{15}N$ (Mean ± S.E.) of the main organic sources, zooplankton and benthic invertebrates along a transect through a fish farm in the southeast arm of Lake Malawi. Mean $\delta^{13}C$ and $\delta^{15}N$ for feed and feces were calculated using grower and finisher diets only.

Sample ID	n	$\delta^{13}C$	±S.E	$\delta^{15}N$	±S.E
Fish feed					
Starter	8	-25.15	0.45	2.70	0.16
Grower	8	-22.90	0.61	3.07	0.19
Finisher	8	-21.87	0.28	2.83	0.22
Fish feces					
Starter	2	-23.22	0.12	3.93	0.14
Grower	2	-20.57	0.03	3.70	0.03
Finisher	2	-17.54	0.21	4.98	0.17
Phytoplankton					
CS4	8	-22.41	0.42	4.25	0.68
CS2	8	-22.44	0.35	4.10	0.64
Fish Farm	27	-22.14	0.31	3.70	0.37
CN2	8	-21.71	0.29	3.97	0.55
CN4	8	-22.52	0.28	3.07	0.64
Sediments					
CS4	8	-21.46	0.18	1.40	0.08
CS2	8	-21.43	0.09	1.45	0.11
Fish Farm	36	-20.56	0.16	1.82	0.11
CN2	8	-17.70	0.22	0.58	0.20
CN4	8	-21.03	0.05	1.21	0.10
Zooplankton					
CS4	11	-22.65	0.28	4.06	0.33
CS2	10	-22.77	0.34	4.03	0.32
Fish Farm	28	-21.82	0.19	3.42	0.21
CN2	11	-21.83	0.31	4.27	0.53
CN4	11	-22.24	0.38	3.96	0.39

Table 4.1. continued.

Sample ID	n	$\delta^{13}C$	±S.E.	$\delta^{15}N$	±S.E.
Mussels					
CS4	5	-21.39	0.19	2.74	0.16
CS2	4	-21.61	0.14	2.55	0.10
Fish Farm	26	-20.93	0.08	2.80	0.10
CN2	2	-22.47	na	1.87	na
CN4	2	-21.74	0.51	2.57	0.34
Snails					
CS4	7	-18.32	0.37	3.32	0.06
CS2	7	-18.82	0.45	3.37	0.17
Fish Farm	37	-18.91	0.24	3.40	0.11
CN2	5	-20.15	0.39	2.38	0.30
CN4	7	-18.84	0.36	3.51	0.13
Benthic worms					
CS4	5	-21.88	0.50	3.65	0.29
CS2	5	-22.26	0.42	3.76	0.28
Fish Farm	20	-22.22	0.17	3.51	0.13
CN2	5	-21.09	0.67	1.73	0.41
CN4	6	-21.92	0.13	3.83	0.09
Periphytic algae (on nets)	8	-15.88	0.87	1.51	0.24
Trap material – Cages	34	-22.32	0.13	2.90	0.09
Trap material – Control	7	-22.12	0.27	2.06	0.43

Table 4.2. $\delta^{13}C$ and $\delta^{15}N$ of fish species caught in the vicinity of fish cages in the south east arm of Lake Malawi. Trophic levels (Tr. Level) are according to Froese and Pauly (2009). TL = fish total length in centimeters (cm).

Species	n	TL(cm)	Tr. Level	$\delta^{13}C$	$\delta^{15}N$
Piscivores					
Buccochromis atritaeniatus	1	11.5	4.5	-20.18	7.71
B. atritaeniatus	2	14.1	4.5	-20.49	8.73
B. atritaeniatus	3	16.0	4.5	-20.32	8.47
B. atritaeniatus	4	16.8	4.5	-19.58	8.43
Buccochromis heterotaenia	1	12.8	4.5	-18.78	7.79
Buccochromis nototaenia	1	13.8	3.0	-20.55	9.09
B. nototaenia	2	17.0	3.0	-20.62	8.46
B. nototaenia	3	17.1	3.0	-20.68	9.07
Champsochromis caeruleus	1	15.4	4.5	-21.77	7.72
C. caeruleus	2	21.3	4.5	-21.72	8.36
Nimbochromis livingstoni	1	11.9	4.5	-20.02	8.34
Nimbochromis polystigma	1	10.8	4.5	-19.09	7.46
N. polystigma	2	11.0	4.5	-19.53	8.01
Opsaridium microlepis	1	13.2	4.0	-19.00	6.83
Rhamphochromis spp.	1	15.1	4.5	-21.73	8.71
Rhamphochromis spp.	2	15.2	4.5	-21.62	8.81
Rhamphochromis spp.	3	15.3	4.5	-20.93	8.08
Rhamphochromis spp.	4	15.4	4.5	-21.62	8.75
Rhamphochromis spp.	5	15.4	4.5	-20.88	7.10
Rhamphochromis spp.	6	16.6	4.5	-21.35	7.27
Rhamphochromis spp.	7	17.3	4.5	-22.29	7.48
Rhamphochromis spp.	8	19.1	4.5	-21.23	7.96
Rhamphochromis spp.	9	20.2	4.5	-21.68	8.49
Rhamphochromis spp.	10	28.7	4.5	-22.92	8.56
Stigmatochromis woodi	1	17.1	4.5	-20.06	9.30
Trematocranus placodon	1	15.4	3.5	-17.30	6.85
T. placodon	2	15.3	3.5	-16.82	7.56
T. placodon	3	16.1	3.5	-17.74	7.82
T. placodon	4	17.4	3.5	-18.00	7.76
T. placodon	5	17.5	3.5	-19.22	5.98

Table 4.2. continued.

Species	n	TL(cm)	Tr. Level	$\delta^{13}C$	$\delta^{15}N$
Zooplanktivores					
Copadichromis chrysonotus	1	12.1	3.4	-20.95	7.35
C. chrysonotus	2	12.5	3.4	-20.99	7.94
C. chrysonotus	3	12.5	3.4	-20.01	7.57
C. chrysonotus	4	12.7	3.4	-20.77	8.27
C. chrysonotus	5	13.0	3.4	-20.31	8.85
C. chrysonotus	6	13.1	3.4	-21.41	8.04
C. chrysonotus	7	13.7	3.4	-21.52	7.81
C. chrysonotus	8	13.7	3.4	-21.30	7.45
C. chrysonotus	9	13.8	3.4	-20.83	8.11
C. chrysonotus	10	13.9	3.4	-20.86	7.64
C. chrysonotus	11	14.9	3.4	-19.90	8.39
C. chrysonotus	12	15.0	3.4	-20.59	8.38
C. chrysonotus	13	15.1	3.4	-21.31	8.84
Copadichromis eucinostomus	1	11.5	3.0	-21.32	7.49
C. eucinostomus	2	11.9	3.0	-21.31	6.84
C. eucinostomus	3	11.9	3.0	-21.29	7.04
C. eucinostomus	4	12.7	3.0	-20.47	7.37
C. eucinostomus	5	11.5	3.0	-21.32	7.49
C. eucinostomus	6	13.8	3.0	-21.21	7.87
Copadichromis virginalis	1	13.3	3.1	-21.44	7.23
C. virginalis	2	13.5	3.1	-21.57	7.33
C. virginalis	3	13.6	3.1	-21.81	7.67
C. virginalis	4	13.8	3.1	-21.27	7.73
C. virginalis	5	14.4	3.1	-21.64	7.65
C. virginalis	6	14.7	3.1	-20.83	7.66
C. virginalis	7	15.1	3.1	-20.49	8.33
C. virginalis	8	15.5	3.1	-20.79	8.12
C. virginalis	9	15.5	3.1	-19.71	8.46
C. virginalis	10	16.4		-21.17	8.09
Diplotaxodon limnothrissa	1	8.7	4.4	-21.53	7.10
D. limnothrissa	2	8.9	4.4	-23.02	5.63
D. limnothrissa	3	9.5	4.4	-22.48	7.40
D. limnothrissa	4	11.5	4.4	-21.83	7.23
D. limnothrissa	5	13.5	4.4	-21.45	8.21
Engraulicypris sardella	1	na	3.1	-23.44	6.36
E. sardella	2	na	3.1	-22.77	5.72
E. sardella	3	na	3.1	-23.03	6.69
E. sardella	4	na	3.1	-23.76	5.78

Table 4.2. continued.

Species	n	TL(cm)	Tr. Level	$\delta^{13}C$	$\delta^{15}N$
Zoobenthivores					
Ctenopharynx intermedius	1	14.4	3.4	-21.02	7.52
C. intermedius	2	15.0	3.4	-20.58	7.58
C. intermedius	3	17.2	3.4	-21.13	7.30
Ctenopharynx pictus	1	11.4	3.0	-19.29	6.94
Fossorochromis rostratus	1	16.4	3.4	-20.20	6.23
Mylochromis lateristriga	1	15.3		-24.08	6.57
M. lateristriga	2	15.4		-22.89	6.09
M. lateristriga	3	16.3		-18.37	7.81
Nyassachromis argyrosoma	1	11.8	3.0	-20.71	7.37
Pseudotropheus elegans	1	11.4	3.3	-20.55	6.69
P. elegans	2	12.6	3.3	-20.00	7.11
Placidochromis subocularis	1	10.7	3.4	-17.88	7.45
P. subocularis	2	12.4	3.4	-18.62	7.38
Protomelas triaenodon	1	16.7	3.0	-21.37	8.62
Trematocranus microstoma	1	23.3	3.4	-19.79	7.71
T. microstoma					
Benthivores					
Astatotilapia calliptera	1	11.1	3.0	-17.08	8.20
A.calliptera	2	13.3	3.0	-17.49	8.49
Lethrinops macrochir	1	16.0	3.0	-20.18	9.13
L. macrochir	2	16.3	3.0	-21.26	9.44
L. macrochir	3	16.6	3.0	-20.11	9.10
Herbivores					
Hemitilapia oxyrhynchus	1	~9	2.0	-21.63	5.73
H. oxyrhynchus	2	~9	2.0	-21.81	5.46
H. oxyrhynchus	3	~9	2.0	-19.45	5.70
H. oxyrhynchus	4	~9	2.0	-22.26	5.45
H. oxyrhynchus	5	~9	2.0	-21.05	5.69
H. oxyrhynchus	6	~9	2.0	-18.30	6.59
Oreochromis karongae	1	12.5	2.0	-16.62	3.29
O. karongae	2	12.5	2.0	-19.74	3.61
O. karongae	3	12.5	2.0	-19.97	2.62
O. karongae	4	13.3	2.0	-15.94	3.97
O. karongae	5	31.5	2.0	-20.52	5.63
Protomelas similis	1	10.6	2.0	-17.62	7.72
P. similis	2	10.8	2.0	-17.77	5.75
P. similis	3	16.3	2.0	-21.45	8.61

Table 4.3. $\delta^{13}C$ and $\delta^{15}N$ signals of *Oreochromis* species reared in fish cages in the south east arm of Lake Malawi. TL = total length in centimeters (cm).

Species	TL (cm)	Date Sampled	Diet at time of sampling	**$\delta^{13}C$**	**$\delta^{15}N$**
O. karongae	18.2	10/01/2007	Grower	-19.51	7.22
O. karongae	18.4	10/01/2007	Grower	-20.29	5.79
O. karongae	18.8	10/01/2007	Grower	-20.42	7.30
O. karongae	18.8	10/01/2007	Grower	-20.10	6.89
O. karongae	19.0	10/01/2007	Grower	-19.67	6.51
O. karongae	19.1	10/01/2007	Grower	-20.80	6.60
O. karongae	19.2	10/01/2007	Grower	-20.86	6.38
O. karongae	19.6	10/01/2007	Grower	-20.14	7.72
O. karongae	20.0	10/01/2007	Grower	-20.62	6.90
O. karongae	20.3	10/01/2007	Grower	-19.77	7.30
O. karongae	14.3	21/05/2007	Grower	-22.39	5.87
O. karongae	16.6	21/05/2007	Grower	-22.59	5.85
O. karongae	19.3	21/05/2007	Grower	-22.93	5.97
O. karongae	20.1	21/05/2007	Grower	-21.00	5.89
O. karongae	22.1	21/05/2007	Finisher	-22.98	6.93
O. karongae	22.2	21/05/2007	Finisher	-21.69	6.77
O. karongae	22.6	21/05/2007	Finisher	-21.65	6.92
O. karongae	23.1	21/05/2007	Finisher	-22.39	6.89
O. karongae	23.3	21/05/2007	Finisher	-22.56	6.70
O. karongae	16.2	19/12/2007	Grower	-21.61	5.41
O. karongae	17.9	19/12/2007	Grower	-22.45	6.08
O. karongae	23.5	19/12/2007	Finisher	-22.38	5.93
O. karongae	24.2	19/12/2007	Finisher	-23.08	5.93
O. shiranus	25.2	19/12/2007	Finisher	-23.38	5.73
O. shiranus	25.4	19/12/2007	Finisher	-22.74	6.08
O. shiranus	27.5	19/12/2007	Finisher	-21.49	5.29
O. shiranus	27.8	19/12/2007	Finisher	-22.23	5.51

4.5 Discussion

The results presented in this study show that cage wastes were incorporated into the food web around the fish cages as indicated by differences in $\delta^{13}C$ and/or $\delta^{15}N$ in different organisms and sediments collected in the vicinity of the cages compared to control stations. These differences may be a result of uptake of dissolved nutrients released from fish cages by primary producers and direct consumption of the cage wastes and/or the algal material by animals in the vicinity of the cages (Dolenec et al., 2007). The phytoplankton $\delta^{13}C$ and $\delta^{15}N$ signals were rather insensitive to cage wastes, which could be attributed to their rapid dispersion from the cage area as water moves through the cage area at high velocity which was on average 9.3 cm s^{-1}. Estimates show that at this mean current velocity one water mass would be translocated to ~0.5 km from the fish farm within 1-2 hrs and replaced by a different water mass with different phytoplankton abundance. These results agree with data presented in Chapter 3 (Fig. 3.18) which showed that algal biomass as chlorophyll fluorescence at the fish farm changed significantly over short time periods.

It has generally been observed that $\delta^{13}C$ and $\delta^{15}N$ in a consumer enrich systematically by an average of 1‰ and 3.4‰ relative to diet, respectively (Minagawa & Wada, 1984). Consequently, $\delta^{13}C$ has been used to identify sources of carbon and nitrogen in a consumer's diet while $\delta^{15}N$ has been used to estimate trophic transfers from source to the consumers. It was therefore expected that $\delta^{15}N$ of the plankton samples retained by the coarser zooplankton net (73-µm Nitex) would be heavier than that of the plankton samples retained by the finer phytoplankton net (10-µm Nitex). The lack of $\delta^{15}N$ enrichment between these plankton samples

suggests that the phytoplankton net might have retained both phytoplankton and even more zooplankton even after screening to remove zooplankton. Relatively similar results ($\delta^{13}C$, -21.6‰ and $\delta^{15}N$, 2.4‰) have been reported by Genner et al (2003) in plankton samples collected in Lake Malawi using a 50-µm phytoplankton net but no further separation into zooplankton and phytoplankton fractions was done. The lack of enrichment also suggests that the phytoplankton samples might have contained a significant amount of an enriched fraction of the phytoplankton community that was not part of the zooplankton diet. Guildford (pers. comm.) reported that picophytoplankton (<2 µm) and net phytoplankton (> 20 µm) in Lake Malawi account for about50% and 15% of total chlorophyll (<1 µg L^{-1}), respectively. Thus the plankton material retained by the 10-µm phytoplankton net may not be representative of the actual phytoplankton signal consumed by zooplankton, which Irvine and Waya (1999) have described to have small body sizes. Hecky and Kling (1987), Hecky and Kilham (1988), FAO (1993), and other studies have reported that the phytoplankton community in the southeast arm of Lake Malawi is often dominated by large filamentous *Aulacoseira* species which were not separated out of the plankton samples to obtain $\delta^{13}C$ and $\delta^{15}N$ signals of the phytoplankton material that constitutes the zooplankton diet. The dominant *Aulacoseira* would alter the phytoplankton signal since phytoplankton species or size fractions vary in $\delta^{13}C$ and $\delta^{15}N$ signals depending on their growth rate and other environmental factors such as temperature and dissolved nitrogen source (Goering et al., 1990). Rau et al (1990) observed consistently depleted $\delta^{13}C$ and $\delta^{15}N$ signals in <8 µm fraction relative to larger fractions of the plankton material in the Mediterranean Sea. In Lake Malawi, Kidd et al (2003) reported particulate organic matter (POM) mean $\delta^{15}N$ signatures of

0.04 ± 1.5‰ and also similar bivalve $\delta^{15}N$ signatures as found here. So it seems likely that the mean phytoplankton $\delta^{15}N$ would be quite similar to that reported by Kidd et al (2003). In addition, wild caught *O. karongae* (with exception of one large specimen in Table 4.2) which feed on phytoplankton and benthic algae had a mean $\delta^{15}N$ signature of 3.3‰ which, after subtracting 3.4‰ for trophic fractionation, would be consistent with a mean phytoplankton signature of approximately 0.0 ‰. In this study, $\delta^{15}N$ of a few phytoplankton samples varied between 0.1 and 2.0‰ (mean, 1.2‰) in agreement with the estimate of 0.0‰ by Kidd et al (2003).

The differences in $\delta^{13}C$ and $\delta^{15}N$ of starter, grower and finisher diets observed in this study, where $\delta^{13}C$ of starter was significantly lighter by 2.2‰ and 3.3‰ relative to grower and finisher diets, respectively, (Table 4.1) was due to differences in diet formulations rather than the raw materials used. All three diets were prepared from the same raw materials which included fish, soy and maize meals. The percent compositions of the ingredients in the diets were, however, different since the starter diet had a higher protein content than the other two diets. To facilitate comparisons of $\delta^{13}C$ and $\delta^{15}N$ in diets with values in caged fish (which were on grower and finisher diets) and other potential consumers and organic sources, average $\delta^{13}C$ and $\delta^{15}N$ values were calculated for the feed at -22.39±0.35 and 2.95±0.14‰, respectively, using grower and finisher signals only. Starter diet was not included in the calculations because only grower and finisher cages were sampled. In addition, starter (~20% of total feed used) was supplied to the caged fish during the first 89±28 days of the 376±42 day production cycle where it added between 50-70g of fish biomass. The lighter starter signal in the initial 50-70g biomass of the

fish was significantly diluted at harvest as indicated by the stable isotope data. The results are in agreement with observations by Hesslein et al (1993) which showed that $\delta^{34}S$, $\delta^{13}C$ and $\delta^{15}N$ values of a rapidly growing fish on changing diets will be determined by the isotopic composition of the food that adds the most tissue or biomass to the fish. Although the caged fish were frequently observed to feed on the periphytic algae on cage nets, the isotopic data show that the periphytic algae was poorly assimilated as indicated by the differences in $\delta^{13}C$ and $\delta^{15}N$ of 0.85 and 3.43‰, respectively, between feed and caged fish which are consistent with changes generally observed between an obligate consumer and diet (DeNiro & Epstein, 1981; Peterson & Fry, 1987; Dempson & Power, 2004). The data, therefore, show that the caged fish relied heavily on the prepared feed for growth as expected from the high fish biomass in the cages and low autochthonous pelagic and benthic primary production rates in the lake. Data in chapter 2 showed that total fish harvest in the cages varied between 3346.76 and 23104.00 kg wet weight (mean of 12,869.25 kg) and required an average C input (excluding losses) of about 30.25 g C m^{-2} d^{-1} (Fig. 2.5). The daily amount of C utilized in the cages is much higher than what could be provided by the autochthonous pelagic and benthic primary productions in the lake estimated at 0.462 g C m^{-2} d^{-1} (Guildford et al, 2007) and 1 g C m^{-2} d^{-1} (Bootsma & Hecky, 2001) respectively. Degnbol (1993) estimated pelagic primary production in the lake at 0.74 g C m^{-2} d^{-1}. Although the productivity of the periphytic algae on cage nets in Lake Malawi was not measured, it is also likely to contribute minimally to the nutrition of the caged fish because of the small surface area it covers on the cage nets relative to the fish biomass in the cages. Norberg (1999) showed in Lake Kariba that the periphytic algae is in general a minor (~1%) source of

energy for tilapias reared in cages in large water bodies. In pond aquaculture, the contribution of the natural food to the nutrition of the reared fish may, however, be significant particularly at low stocking density as the fish range freely and have access to natural food resources. Anderson et al (1987) studied shrimp (*Penaeus vannamei*) in pond cages and observed lack of dependence of the reared shrimp on the prepared feed at low stocking density because their C demand was easily satisfied by the primary productivity in the ponds.

Sedimentation of cage wastes is one of the main impacts of cage operations on the receiving waters and sediments (Gowen & Bradbury, 1987; Holmer, 1991; Pergent et al., 1999; La Rossa et al., 2001; Ruiz et al., 2001, Kovac et al., 2004; Sarà et al., 2004, Dolenec et al., 2006). The use of stable isotope analyses to demonstrate accumulation of organic matter on sediments below fish cages has however provided conflicting results with positive identification of the cage wastes in some studies (Ye et al., 1991; McGhie et al., 2000, Sarà et al, 2004) and negative in others (Grey et al., 2004). In the current study, stable isotope analyses confirmed that sediments below the fish farm were receiving enriched organic material falling from the fish cages since both $\delta^{13}C$ and $\delta^{15}N$ were significantly enriched below the fish farm compared with control stations. These results agree with sediment trap measurements presented in chapter 2 which showed that sedimentation rates of particulate C and N were, respectively, 21 and 18 times higher under the cages than at control stations. The sediment traps were deployed ~1.5m below the cage net so that a larger portion of the particulate material falling from the cages was likely consumed by the large numbers of wild fish which aggregated around the cages or dispersed by the high water current so that only a small portion was deposited in the sediments.

Consequently, the radius around the fish farm impacted by the falling cage wastes could not be easily determined using the data. Many studies have shown that impacts are usually observed within 300 m radius of the fish farm (Holmer et al., 2007), but the transect that was sampled in this study involved sites within the fish farm and control stations which were much further away (> 1 km, Table 3.1) from the area expected to be affected by the cage wastes. However, as the fish farm continues to operate and expand, the isotopically enriched cage wastes may spread and be detectable over a larger area (Sarà et al., 2004). Analysis of the sediments collected below the fish farm as well as at control stations also revealed that the sediments were unexpectedly quite depleted in $\delta^{15}N$ compared to organic-N sources such as plankton and cage wastes collected in sediment traps. The sediments were actually expected to be heavier than phytoplankton and cage wastes due to preferential loss of lighter $\delta^{13}C$ and $\delta^{15}N$ signals during mineralization in the water column and sediments. Altabet (1988) observed heavier $\delta^{15}N$ signals in sinking particles below the euphotic zone and depleted particles in surface waters in the open ocean. The low $\delta^{15}N$ of sediments (close to 0‰) may, therefore, be attributed to benthic fixation of dissolved N_2 which has a $\delta^{15}N$ of 0‰. Gondwe (2004) and Gondwe et al (2009) studied the sandy communities of two beaches ~ 4 and 80 km from the fish farm in Lake Malawi and demonstrated benthic N_2-fixation by eubacteria. The N_2-fixation rates were higher at deeper depths (the deepest station sampled was 10 m in that study) than at shallower stations suggesting that N_2-fixation by eubacteria might extend under the cage farm and at the control stations. Bacterial N_2-fixers would benefit from high input of organic C under the cages compared to control areas especially if the waste were somewhat N deficient (Capone, 1988). It is, therefore, possible that N_2-fixation

may be offsetting the input of isotopically heavier organic particles falling to the bottom both at the fish farm and at the control sites.

The impact of the cage wastes on the benthic organisms in Lake Malawi was conflicting, as bivalves showed significant enrichment in both $\delta^{13}C$ and $\delta^{15}N$ under the cages compared to control sites while snails were negligibly affected. The insensitivity of the snails to the impact of cage wastes may be attributable to their selective feeding behavior which suggests that snails may have preferred attached benthic algal material to cage wastes during feeding. In a similar study of cage waste utilization at a fish farm in Esthwaite Water, UK, Grey et al (2004) invoked selective feeding on pellet feed material by chironomids to explain their large shifts in $\delta^{13}C$ and $\delta^{15}N$ signals compared with the other organisms below the fish farm. Such selective feeding displayed by snails and chironomids may not be as strong with filter feeders such as bivalves.

In this study, snail species were always enriched in both $\delta^{13}C$ and $\delta^{15}N$ compared to bivalve species collected from the same control location. Hecky and Hesslein (1995) also studied snails and bivalves in Lake Malawi and reported similar trends in their isotopic signals. The trend between snails and bivalves has been attributed to their feeding methods since bivalves filter particulate materials in the water column which have lighter phytoplankton signal between -21 to -22‰ while snails scrap isotopically heavier benthic material (Hecky & Hesslein, 1995). Whilst this study and that of Hecky and Hesslein (1995) obtained similar $\delta^{13}C$ signals for bivalves (around -21.5‰), $\delta^{13}C$ signals of snails and $\delta^{15}N$ of both bivalves and snails were very different. Hecky and Hesslein (1995) reported $\delta^{13}C$ signals of snails in Lake Malawi at -11 to -12‰ but in this study the values ranged between -16 and -22‰ with a mean of about -19‰. The

$\delta^{15}N$ signals were reported at about 0.3‰ for bivalve species and at -2.5‰ for snail species (Hecky & Hesslein, 1995) but in the current study snails ranged between 1 and 4‰ (mean of 3.3‰) while bivalves ranged between 1.0 and 3.8‰ (mean of 2.7‰). These differences may be due to sampling locations since samples in this study were collected from mainly sandy stations that were approximately 1 km from shore and water depths between 13 and 23m while samples analyzed by Hecky and Hesslein (1995) were collected from rocky shoreline areas with higher N_2-fixation (Higgins et al., 2001; Gondwe et al., 2009) and primary production rates (Bootsma & Hecky, 2001) which minimized ^{13}C fractionation. Benthic worms sampled below the fish farm had significantly lighter $\delta^{13}C$ signals than snail and bivalve species ($p \leq 0.05$).

Although the attached periphytic algae displayed a broad variation in the $\delta^{13}C$ signal that ranged between -20.2 and -12.7‰ (n = 8), the mean $\delta^{13}C$ signal was significantly enriched at -15.9‰ compared with phytoplankton and fish feed (Fig. 4.1). This enrichment in $\delta^{13}C$ of the periphytic algae could primarily be attributed to high nutrient concentrations in the vicinity of the fish cages as discussed below. Under normal circumstances plants, including periphytic algae, prefer CO_2 to HCO_3^- and CO_3^{2-} as the carbon substrate for photosynthesis because use of CO_2 does not require energy expenditure for uptake and charge balance (Sharkey & Berry, 1985 in Hecky & Hesslein, 1995). CO_2 is also isotopically lighter than HCO_3^- and CO_3^{2-} due to equilibrium fractionation (Hecky & Hesslein, 1995) so that the photosynthate will also be lighter in $\delta^{13}C$ signal. At high algal growth rates induced by high nutrient availability and high irradiance, photosynthetic uptake of CO_2 can exceed rates at which CO_2 diffuses from the atmosphere into the water so that a shift to the use of isotopically heavier HCO_3^- may occur to

meet the photosynthetic demand. The demand-induced uptake of HCO_3^- makes the photosynthate isotopically heavier than if CO_2 was used throughout the photosynthetic process. High photosynthesis rates can also increase the $\delta^{13}C$ signal of photosynthate by depleting the dissolved CO_2 so that uptake rates and isotopic discrimination depend solely on CO_2 transport processes (i.e. no enzymatic discrimination) (Hecky & Hesslein, 1995). Such nutrient induced uptake of HCO_3^- may occur in waters adjacent to fish cages since the water immediately around fish cages contains high nutrient concentrations in readily bioavailable forms of PO_4^{2-} and NH_4^+ from fish feed and fish excretes. High concentrations of dissolved organic carbon (La Rossa et al., 2002), PO_4^{2-} and NH_4^+ (Pitta et al., 1998) have been reported in waters adjacent to fish farms in the Mediterranean Sea. Troell et al (1997) and Chopin et al (1999) have measured elevated algal growth rates supported by high levels of dissolved inorganic nutrients in waters adjacent to fish farms. In Lake Malawi, the shift to using HCO_3^- may occur even at moderate photosynthesis rates since studies have shown that dissolved CO_2 in Lake Malawi waters is naturally low, well below levels in high latitude lakes (Hecky & Hesslein, 1995).

The $\delta^{13}C$ of periphytic algae observed in this study which ranged between -20.2 and -12.7‰ (mean = -15.9‰) also suggest that uptake of fish-respired CO_2 by the periphytic algae for photosynthesis was low. Because respired CO_2 is isotopically lighter at $\delta^{13}C$ between -35 and -20‰ (France, 2000 cited in Post, 2002), any significant uptake of the respired CO_2 could have depleted the $\delta^{13}C$ signal of the periphytic algae to $\leq$ -20‰. Uptake of respired CO_2 for photosynthesis by phytoplankton was previously invoked by Vander Zanden et al (2006) to

explain extreme negative $\delta^{13}C$ signatures (-44‰) of profundal primary consumers in Castle Lake, California.

A significant difference was observed in both $\delta^{13}C$ and $\delta^{15}N$ between periphytic algae on cage nets and phytoplankton moving with current through the cage area. The main contrast between periphytic algae and phytoplankton in this study was probably the availability of nutrients which was much higher for periphytic algae on cage nets than for phytoplankton washed through the cage area. Consequently, periphytic algae might have experienced higher nutrient-induced photosynthetic rates and less ^{13}C fractionation while phytoplankton had lower photosynthetic rates and increased ^{13}C fractionation. Whilst periphytic algae on the cage structures reflect the $\delta^{15}N$ signal of excreted NH_4^+, the phytoplankton should reflect the $\delta^{15}N$ of the inorganic N pool in the mixed layer as they were rapidly washed through the cage area by water currents.

In Chapter 2, I indicated that large numbers of wild fish aggregated around the fish cages in Lake Malawi (see Figs 2.1 & 2.2). The $\delta^{13}C$ data for the wild fish species indicate that the fish species which aggregated at the fish farm initially belonged to two distinct food webs, namely the benthic food web in the littoral zone and pelagic food web in the upper water column. Littoral species at $\delta^{13}C$ signatures between -16 and -18‰ included *P. similis, H. oxyrhynchus, M. lalestriga, A. calliptera*, zoobenthivorous *P. subocularis* and two small *O. karongae* (Table 4.2). The rest were probably pelagic species with lighter $\delta^{13}C$ signatures between -20 and -24‰ supported by phytoplankton (see Kidd et al. 2003). Trophic positions of the fish species caught at the fish farm have been reported as assigned by Froese and Pauly (2009) (Table 4.2) and show

that most of the fish species would occupy trophic levels between 3 and 4.6 in their natural environments. Only three species (*H. oxyrhynchus, O. karongae and P. similis*) caught at the fish farm belonged to the second trophic level (trophic level = 2) in the food web.

The $\delta^{15}N$ data indicate a high degree of diet overlap in all species and certainly did not allocate them to distinct trophic levels (Fig. 4.2). Generally a $\delta^{15}N$ enrichment of 3.4‰ is expected between trophic levels. However, if a consumer has more than one food source with distinctly different isotopic signals, the $\delta^{15}N$ may be <3.4‰. The difference in $\delta^{15}N$ between the top consumer piscivore and its potential food resources (zooplanktivore, herbivore, zoobenthivore and molluscivore) varied between 0.64 and 1.9‰ which clearly shows use of multiple food sources. The $\delta^{15}N$ results suggest that the caged fish and the wild fish were utilizing the same food source, namely the artificial feed and feces. Large differences in $\delta^{15}N$ values of organisms in the first consumer level (zooplankton, bivalves, snails and benthic worms and some *O. karongae* compared to *H. oxyrhynchus, F. rostratus* and one large *O.karongae*) were observed, which might be attributable to the different degrees the cage wastes were utilized as food source by the different organisms. Consumption of the cage wastes has similarly affected consumers in the upper trophic levels such as piscivores, zooplanktivores and molluscivores. The lack of clear separation between the trophic levels in the fish species caught at the fish farm, particularly in Fig. 4.2, clearly shows a substantial integration of $\delta^{15}N$ signals of the cage wastes with their own tissue signals carried-over from previous feeding behavior or with some continued utilization of wild foods.

To further demonstrate consumption of cage wastes by the wild fish, I compared $\delta^{13}C$ and $\delta^{15}N$ signals in control samples collected between 1995 and 1997 by Hecky and Hesslein (1995) and Kidd et al (2003) and in fish samples collected in the vicinity of the fish farm in 2007. Despite the differences in fish sizes in Figs. 4.3 & 4.4, control fish samples (CS) analyzed by Hecky and Hesslein (1995) and Kidd et al (2003) have different $\delta^{13}C$ and $\delta^{15}N$ values from fish samples collected in the vicinity of the fish farm (FF) in 2007. Some of the differences in $\delta^{13}C$ and $\delta^{15}N$ signals between the two groups of samples, particularly for *Ramphochromis* spp. (Rsp) and Ctenopharynx *pictus* (Ctp), may partly be attributed to ontogenetic shifts (Fig. 4.3). Differences displayed by *Buccochromis nototaenia* (Bn), *Copadichromis* spp. (Cpd), *Trematocranus microstoma* (Tm) and *Oreochromis karongae* (Ok) are most likely due to consumption of cage wastes (feed and feces) (Fig. 4.3). *T. microstoma*, a zoobenthivorous species, displayed a remarkable shift both in $\delta^{13}C$ and $\delta^{15}N$ of 6.7‰ and 1.9‰ respectively. *O. karongae* also showed a remarkable shift, particularly for the large (339 and 449g) samples. Kidd et al (2003) measured mercury concentrations in Lake Malawi fish and reported increasing concentrations with size for all species except for *O. karongae*,which showed decreasing Hg concentration with size. The Hg trend in *O. karongae* species was attributed to shifts in their feeding habits with size. Small *O. karongae* species are generally planktivorous with more negative $\delta^{13}C$ signals, but shift to less Hg-contaminated benthic food sources as the fish grows bigger. Consequently, larger *O. karongae* have more enriched $\delta^{13}C$ values (-11.6 to -16.6‰) and less tissue Hg concentrations (Kidd et al., 2003) while smaller ones have depleted $\delta^{13}C$ signals and higher Hg concentrations. Comparison of $\delta^{13}C$ signals for large *O. karongae* sampled by

Kidd et al (2003) (Ok-CS, 339 g at -14.40‰) and by this study at the fish farm (Ok-FF, 449 g at -20.52‰) shows contradiction because the size-dependent feeding behavior for *O. karongae* presented above predicts a comparable or heavier $\delta^{13}C$ value for Ok-FF (449g) than Ok-CS (339g) at -14.40‰ under natural conditions. The contradiction may be explained by the consumption of cage wastes which are isotopically lighter than the benthic food resources utilized by the larger *O. karongae* species. The same applies to Bn-CS (472g) and Bn-FF (60g) where ontogenetic shift predicts a larger $\delta^{15}N$ value for Bn-CS (472g) than for a Bn-FF (60g), relatively comparable $\delta^{15}N$ values for Cpd-CS (22g) and Cpd-FF (32g) samples, and a much larger $\Delta\delta^{15}N$ value between Dl-CS (36g) and Dl-FF (10g), i.e., much lower $\delta^{15}N$ value for Dl-FF (10g) than observed here. The fish sizes analyzed by Hecky and Hesslein (1995) were not available but Fig. 4.4 clearly shows that $\delta^{13}C$ and $\delta^{15}N$ signals in wild fish caught in the vicinity of fish cages are shifting towards signals observed in cultured fish as depicted by a big circle.

4.6 Conclusion

This study has confirmed the exploitation of the cage wastes by the biota in the vicinity of the fish cages as a source of inorganic and organic C and N. Primary producers, especially periphytic algae on cage structures and benthic algae, assimilated the cage-derived dissolved N while the animal component of the biota consumed diets which contained the cage-derived C and N. Significant differences in $\delta^{13}C$ and $\delta^{15}N$ signals were observed in most elements of the biota and sediments collected at the fish farm and at control sites. In other words, the study was able to trace $\delta^{13}C$ and $\delta^{15}N$ signals of the cage wastes in the food web in the vicinity of the fish cages.

These results agree with the large aggregation of wild fish around fish cages to consume wastes falling from the cages. The study also confirmed the accumulation of cage wastes on sediments below the fish cages in agreement with the results from the particle sedimentation study in chapter 2. Accumulation of the cage wastes below the fish cages was, however, reduced by the consumption by wild fish and rapid dispersion facilitated by high water current velocities (mean = 9.3cm s^{-1}). Benthic invertebrates have generally been considered ideal bioindicators for environmental assessment because they, among other reasons, provide site-specific information because of their low mobility (Dolenec et al., 2007), but because of the high water current velocity (range of 0.4-47.2 cm s^{-1} and mean of 9.3 cm s^{-1}) and rapid consumption of the cage wastes by large numbers of wild fishes which aggregated around the fish cages in Lake Malawi, their role as assessment indicators under the fish cages in the lake may be compromised. Significant shifts in $\delta^{13}C$ and $\delta^{15}N$ signals presented in this study were observed in most elements of the biota and sediments collected at the fish farm and at control sites. In this study, most wild fish species collected in the vicinity of the fish cages showed significant differences in their $\delta^{13}C$ and $\delta^{15}N$ signals compared to similar species collected 7-8 years prior to the cage installation. Although it may not be all fish species, wild fish generally seem to be the best monitors for the cage wastes because they can also be resident at the fish farm. Future monitoring studies might therefore concentrate more on wild fish that aggregate around the cages rather than phytoplankton and zooplankton. Sedimentary organisms, such as snails, bivalves and other organisms may become important as the fish farm activities continue and expand.

Chapter 5
The future of cage aquaculture in Malawi

5.1 Introduction

Previous chapters of this thesis have discussed several potential impacts of a new cage farm at the southern end of Lake Malawi. This chapter looks at the lessons learned from these studies and makes recommendations for the ecological, social and economic sustainability of cage culture in Malawi. The lessons for Lake Malawi will likely be applicable to the other African Great Lakes where cage culture is developing or may develop in the near future. The results of this study are most applicable to the culture of the native tilapia species which are the current choice of the aquaculture operation. This choice makes our studies of production efficiency most strictly applicable to this species and different species should require new studies of conversion efficiency and waste production for those species and feed combinations. Use of native species also reduces potential negative consequences of any escaped cultured fishes into the environment. In fact, the chosen *Oreochromis* species are the subject of restoration efforts by the Department of Fisheries so that the fish which escape from the cages can be viewed as a positive contribution to that restoration program.

5.2 Lessons from the current study

This study has shown that in a large and dynamic lake such as Malawi, environmental assessment should employ various methods so that if one method is unable to detect an effect due to rapid dispersion of the waste over a large area, impacts can still be detected by another method. This study employed three methods to assess environmental impacts of fish farming in the south east arm of Lake Malawi. The methods were (1) the mass balance model using production records from the cage operators to establish how much waste material enters the environment from each cage or unit of fish production, (2) chemical and biological analyses of water and particulate organic matter (POM) samples in the vicinity of the cages as well as at sites more remote from the cages, and (3) $\delta^{13}C$ and $\delta^{15}N$ analysis of biota, sediments and POM collected at the fish farm and at control stations to determine how waste materials may influence food webs in the vicinity of the cages.

The mass balance and isotope methods provided strong indications that the fish farm at its current scale of operation was adding significant amounts of C, N and P into the surrounding environment and that these added material were being assimilated into the food web supporting wild fishes in the vicinity of the fish cages. Data presented in chapter 2 show that waste production in fish cages in Lake Malawi was high, with a feed conversion ratio (FCR) of 2.7 or 2.7 kg of feed per 1 kg gain in fish weight. The mass balance method indicates that 86% of C, 71% of N and 88% of P in feed were discharged into the surrounding environment in dissolved and particulate forms. Fish production varied between 2870 and 19253 kg wet weight/cage/yr (mean = 12,380 kg/cage/yr). The grow-out period (production cycle) between stocking and

harvesting varied between 272 and 438 days, with a mean of 376 days. Waste generation in cages varied between 727.42 and 1478.41 kg C t^{-1} fish harvest (mean = 1000.27 kg C t^{-1} fish harvest), 69.42 and 167.48 kg N t^{-1} fish harvest (mean = 105.61 kg N t^{-1} fish harvest), and between 46.38 and 92.74 kg P t^{-1} fish harvest (mean = 63.27 kg P t^{-1} fish harvest). As has been discussed in chapter 4, the stable isotope method showed that the cage wastes were incorporated in the food web and supported the wild fish species which aggregated around the fish farm (Figs. 4.1, 4.3 & 4.4). In contrast, the chemical and biological methods indicated that the cage wastes had little measurable impact on the water quality in the immediate surrounding environment. This result is consistent with many previous studies on large lakes with large nutrient assimilative capacities that have shown that fish farm operations generally cause no discernable reduction in the water quality (Wu, 1995). Wilson and Boyd (2000) observed no reduction in surface water quality in the Georgian Bay, Lake Huron, where cage culture had been operated for over 2 decades.

In European salmonid farms, approximately 29% of C, 24% of N and 56% of P in feed input have been estimated to accumulate as sediments below fish cages where they cause major impacts (Wu, 1995; Handy & Paxton, 1993). On the contrary, accumulation of cage wastes on sediments below the fish farm in Lake Malawi was minimal. Consequently, the abundance of opportunistic wild fishes on the sediments below the fish cages was significantly lower compared to fish abundance in the water column surrounding the cages. Dissolved oxygen concentrations in bottom waters below the cages were relatively similar to the rest of the water column except during a few days (e.g., Jan 20 and Mar 30, Fig. 3.9) when cage wastes might

have partly depleted the oxygen levels. The low impact of the cage wastes on the water quality and sediments below the fish farm in Lake Malawi was attributable, in chapter 3, to the rapid dispersion of the cage wastes by water currents which varied between 0.4 and 47.2 cm s^{-1} (mean of 9.3 cm s^{-1}) at the fish farm, and high consumption of the particulate cage wastes (unconsumed feed and feces) by the large number of opportunistic wild fish species which aggregated around the fish cages. However, deeper offshore sites would be much desirable for cage fish farming, especially at higher production capacities, than these nearshore areas.

The mean water current velocity of 9.3 cm s^{-1} measured at the fish farm at the southern end of Lake Malawi is higher than mean current velocities reported in the other great lakes of the world. The water current velocity at the fish farm was strong throughout the sampling period between January and December, 2007. Currents >10 cm s^{-1} were frequently measured below the fish farm at almost on a daily basis during the period the current meter was deployed. Even stronger surface currents of at least 27.8 cm s^{-1} (1 km h^{-1}) have been reported by Eccles (1974) in the southeast arm and in other parts of the lake. In conclusion, environmental impact of cage wastes in Lake Malawi was minimal during the study period in 2007 when total fish biomass in the cages was ~200 tonnes from about 25 cages. However, as fish production continues and expands to attain, and even exceed, the operator's target annual harvest of 3,000 tonnes, the amount of cage wastes that may be generated may surpass the assimilative capacity of the ecosystem (dilution and fish consumption) so that impacts of the cage wastes may become significant and may affect a larger area. At a target production capacity of 20 tonnes fish/cage/yr, at least 150 cages would be needed to produce the target harvest of 3,000 tonnes fish/yr, and the

required number of cages would be three times the number of cages currently deployed in the lake. As such, the current assessment should be used with caution as it could be underestimating the potential long-term effect of the cage operation. As already discussed (pages 128 & 173), continued monitoring in the vicinity of the fish farm will be necessary to examine possible long-term effects at a large scale operation.

5.3 Production capacity

The carrying capacity of an ecosystem, or part of it, for cage culture has been divided into four functional categories as follows (Inglis et al., 2000; Mckindsey et al., 2006):

1. Physical carrying capacity – the total area of the fish farm that can be accommodated in the available physical space. It depends on the overlap between the physical requirements of the target species and the physical properties of the area of interest (e.g., type of substrate, depth, hydrodynamics, temperature, dissolved oxygen concentrations, culture techniques)
2. Production carrying capacity – the stocking density of fish at which harvests are maximized.
3. Ecological carrying capacity –the maximum level of production which is possible without causing unacceptable ecological impacts.
4. Social carrying capacity – the level of farm development that causes unacceptable social impacts.

Estimation of an overall carrying capacity for a given area is a complex matter since it must incorporate all the four categories listed above. Consequently, estimation of carrying capacity for the southeast arm of Lake Malawi is beyond the scope of this study especially as no study of the social impacts or impacts on local capture fisheries was included. Because the southeast arm is a large (~1700 km^2) and well flushed area, it will be assumed that the production permit of 3,000 tonnes/yr issued to Maldeco Aquaculture Ltd is well below the carrying capacity of the area. However, the fish farm may experience serious potential impacts from local pollution if farms were concentrated in one location to the point of disrupting the dispersion of cage wastes by water currents. Data presented in Table 2.3 (chapter 2) show that a fish farm producing 15,000 tonnes fish/yr will discharge large amounts C, N and P that would be 1 to 6 times higher than dissolved C, N and P levels observed in the most disturbed river systems, such as Linthipe and Songwe Rivers, draining into the lake. As such, Maldeco Aquaculture will have to expand gradually and cautiously, after consultations, as they monitor local environmental impacts, especially oxygen concentrations in the water column, on an on-going basis (see pages 128 and 173).

5.4 The future of fish farming in Malawi and recommendations

The importance of fish as the main and cheapest source of dietary protein in Malawi has been affected by the declining capture fish stocks in natural water bodies in the country and the rapid population growth rate which has increased consumer demand for food fish. In the 1980s, when the population was about 7 million (Mlia & Kalipeni, 1987), fish provided >70% of the

total dietary protein consumed in the country, but rapid population growth (~13 million in 2009), over-fishing and environmental degradation of breeding grounds overwhelmed the capture fish stocks in most water bodies which reduced the protein supply from fish to the current <30% (Banda et al, 2005). To increase food fish supply, earthen pond aquaculture has been promoted for many years by the government. However, current pond aquaculture production is still low at 800 tonnes from 7000 earthen ponds all over the country (National Aquaculture Center, 2004 *In* Maluwa & Gjerde, 2006). In 2004, cage culture of indigenous *Oreochromis* spp. was launched in Lake Malawi by Maldeco Aquaculture Ltd. Fish production in the cages in the past 4 years has shown that, with availability of low-cost-nutritious feed and proper management, cage culture could be the best alternative to capture fisheries and pond aquaculture in Malawi mainly because of the relatively low capital costs needed to establish a cage farm and the availability of suitable water bodies that may be used for cage culture. Maldeco Aquaculture Ltd, which has a permit to produce 3,000 tonnes fish/yr in Lake Malawi, is currently (2009) operating at ~750 tonnes/yr capacity from 48 cages (M. Mkandawire, farm manager, pers. comm.). Owing to the growing local and international demand for tilapia, cage culture in Malawi is likely to expand in the near future as more and more investors join the industry. A high international demand for fresh and frozen tilapia fillets in the USA and EU has enabled expansion of cage culture in Lake Kariba, a large reservoir on the Zambezi River between Zimbabwe and Zambia, to the current production capacity of 5,000 tonnes/yr. At Maldeco Aquaculture's production target of 20 tonnes/cage/yr, a total of at least 150 cages would have to be deployed in the lake to produce 3,000 tonnes/yr. Reports from the Department of Fisheries in Malawi indicate that more permits may be issued to

investors in the near future to undertake cage culture operations in the lake which would then potentially increase total cage culture production to much >3,000 tonnes/yr. Expansion of cage culture can be accompanied by degradation of the environment in the vicinity of the cages (Phillips et al, 1985; Gowen & Bradbury, 1987). As the fish farm expands, more feed will be used and a greater amount of waste will be generated and discharged into the surrounding environment in dissolved and particulate forms as discussed above. Similar levels of discharges of C, N and P have been reported from temperate cages by Gowen and Bradbury (1987), Holby and Hall (1991) and Hall et al (1990, 1992).

As the number of deployed cages increases in a relatively small area of the lake dictated by culture logistics and security, water current velocity through the cages may be significantly reduced by the cage structures. Interrupted water currents may facilitate immediate sedimentation of the cage wastes in the vicinity of the fish farm (Chapter 3). At fish production capacity of ≥3,000 tonnes/yr, the aggregation of wild fish species around the cages may not be large enough to consume most of the cage wastes as was observed in 2007 when total fish production in the cages was ~200 tonnes. Findlay and Watling (1994) reported that between 5 and 30% of feed supplied to reared fish falls out of the cages unconsumed and combined with fish feces, can accumulate to a large amount over the production cycle. Consequently, a localized accumulation of cage-derived organic matter beneath fish cages would be inevitable if all the required cages (>150 cages) were deployed in a single cage culture site. The concomitant organic enrichment of the sediments below fish cages may have major impacts on the environment as well as on the production performance of the cages as discussed in earlier chapters. The

decomposition of the accumulated organic matter would deplete oxygen levels leading to anoxia and buildup of hydrogen sulphide in the sediments and bottom waters particularly during stratification (Hamblin & Gale, 2002). As a result of mineralization of the organic matter, more dissolved nutrients may be released into the water column leading to further potential eutrophication of the water column and proliferation of toxic algal species (Stirling & Dey, 1990; Hamblin & Gale, 2002). Upwelling of the anoxic bottom water in the vicinity of the fish cages may lead to massive fish kills in the cages. Frequent fish kills have been reported in Taal Lake in Philippines as high nutrient losses from densely stocked and intensively fed fish cages deteriorates the water quality in the lake (Vista et al., 2006). The anoxic conditions in bottom waters will also eliminate the habitat below and around the cages for aquatic animals, including fish. It is, therefore, important that new sites should be identified for expansion so that cage wastes can be diluted over a larger area to minimize negative impacts to the environment and to the performance of the fish farm. In Lake Kariba, farm separation has been employed to minimize impacts of cage wastes by spreading cages over larger areas. The selected sites should be reasonably separated so that there is enough area around the cages and fish farms for nutrient assimilation and oxygen production through photosynthesis, which has been estimated at 0.74 g C m^{-2} d^{-1} in Lake Malawi (Degnbol, 1993). Enough dissolved oxygen, primarily from photosynthesis during prolonged periods of calm weather, will be required for fish respiration and biological oxidation (BOD) of the organic cage wastes. The following estimates for photosynthetic production of dissolved oxygen were calculated to carter for prolonged periods of calm weather when oxygenation of the water column by turbulence (oxygen exchange between

the atmosphere and water) would be minimal. The total required area around the cages has been estimated at 320 times the area of the cage itself (see Berg et al., 1996; Kautsky et al., 1997; **Appendix 2**). At an average fish density of 75.85 kg m^{-2} in Lake Malawi cages, a production of 3,000 tonnes of fish would require about 13 km^2 of open water around the cages for sufficient photosynthetic oxygen production or invasion from the atmosphere to meet the oxygen consumptive effects from fish respiration and biological oxygen demand from cage wastes. According to the Redfield C/P ratio of 40:1, the current primary productivity of 0.74 g C m^{-2} d^{-1} is equivalent to an assimilation of about 18.5 mg P m^{-2} d^{-1}. To maintain the current primary productivity in the lake that would assimilate the cage-derived P at 18.5 mg P m^{-2} d^{-1}, an area of about 14 km^2, or 327 times bigger than the fish farm itself, would be required around the fish farm for nutrient (N and P) assimilation at cage production capacity of 3,000 tonnes/yr. These calculations (see **Appendix 2**) have been done according to Berg et al (1996) and Kautsky et al (1997).

Lake Malawi is seasonally limited by both N and P so that enrichment of the water column and underlying sediments by either N or P from fish cages and other sources may potentially cause eutrophication (Hecky et al., 1996; Guildford & Hecky, 2000; Guildford et al, 2000, 2003, 2007; North et al., 2008). As the fish farm expands, more wastes will inevitably be discharged into the environment which may lead to degradation of the nearshore waters where the cages are currently located, unless appropriate mitigation measures are put in place. The nearshore zone in the lake is already experiencing significant environmental degradation that requires immediate remediation measures. In fact, degradation of the littoral zone has been

partially blamed for the collapse of the chambo fishery which uses the zone for breeding (Banda et al., 2005). To avoid exacerbating degradation of the littoral zone with cage wastes, expansion of the fish farm should therefore be done in deeper and more open offshore waters where dispersion and dilution of cage wastes may be relatively greater than in the nearshore waters.

Because consumption of cage wastes by wild fish played a significant role in the dispersion and mineralization of the cage wastes discharged from the fish cages in Lake Malawi (Chapter 2; also see Vita et al., 2004) as in many cage fish operations, e.g., the Gulf of Aqaba, Red Sea (Katz et al., 2002), more wild fish species will have to be attracted and retained around and below the fish cages as a way of mitigating the waste problem as the fish farm continues and expands. While wild fish species will naturally be attracted to the floating cage structures where they consume cage wastes (Dempster et al., 2002), retaining demersal fish species may require deployment of artificial reefs within the fish farm area as demonstrated at fish farms in the Gulf of Aqaba, Israel (Angel et al., 2002). Hansen and Blackburn (1992), Kristensen et al (1992), and Angel and Spanier (2002) have demonstrated that the benthic infauna (fish, invertebrates, bacteria and others) that colonize the sediments and artificial reefs below fish cages can significantly increase the degradation rate of organic wastes settling in the sediments from the cages. While benthic fish forage for food and swim near the bottom, they resuspend detritus, increase oxygen supply to benthos for respiration and to buried organic matter for decomposition.

The aggregation of wild fish species around fish cages for shelter and food can be viewed as a form of habitat modification, which many studies have shown to foster hybridization or

interbreeding between reproductively isolated plant and animal species (see a review by Rhymer and Simberloff, 1996). Hybridization has a potential consequence of decreasing fish biodiversity in the lake. Lamb and Avise (1986) and Schlefer et al (1986) showed that disturbance of immediate pond surroundings in Alabama increased introgressive hybridization between native green treefrog *Hyla cinerea* and barking treefrog *H. gratiosa* species, which were reproductively isolated by their different habitat requirements. Similarly, aggregation of wild fish species around cages for shelter and food could breakdown spatial barriers that might naturally hinder interbreeding between stenotopic but closely related fish species in the vicinity of the fish farm in Lake Malawi. Hypothetically, the close spatial proximity resulting from the aggregations of the wild fish species around cages should, in the long term, negatively affect species diversity in the lake through hybridization, particularly if cage installations continue in nearshore waters which host most of the fish species in the lake. Studies should, therefore, be urgently conducted to quickly gain more understanding of the negative effects of the fish farm on the wild fish species aggregating around the cages for shelter and food.

Feed quality and quantity are the most important factors which determine nutrient losses from fish cages by determining excretion losses and feed wastage, respectively, so feed quality improvement and feeding management can significantly reduce the total amount of wastes discharged from fish cages (Cho et al., 1994; Cho & Bureau, 1997). Simultaneously, economic benefits would be registered from lower feed losses and higher feed assimilation. Although an attempt was made in 2007-2008 by Maldeco Aquaculture Ltd to improve their fish diets for growth performance, the quality of the developed diets is still similar to the previous diets as

indicated by lack of improvement in the feed conversion ratio (FCR) and therefore more work is needed to improve the feed quality. The desired feed must be highly digestible with an increased energy density (increased fat: protein ratio) and optimal composition of nutrients adapted to the nutritional requirements of the reared species. A proper feeding schedule which does not impose slow growth or result in intensive feeding, as practiced in Taal Lake, Philippines (Vista et al., 2006), is also important in reducing waste discharge from the cages. A proper feeding schedule maximizes feed conversion efficiency and nutrient retention by fish (Ballestrazzi et al., 1998).

5.5 Lessons from the capture fisheries

Malawi, through the Department of Fisheries, intends to restore the chambo fisheries to pre-1990 production levels and meet the food fish demand in the country by enhancing chambo production through aquaculture by 2015 as part of its National Strategy for Sustainable Development (Banda et al., 2005). One approach identified for capture fishery restoration is to control overfishing as it is the main factor that led to the collapse of the chambo fisheries. Overfishing is suggested to have been promoted by "open access" to the fishery, inadequate participatory fisheries co-management and poor enforcement of regulations concerning gear, season and size limits (Banda et al., 2005). Open access means that fishers did not require licenses to fish chambo, as a result the number of gears and fishers increased beyond what the fisheries could sustain. In addition, most of the gears used to catch chambo were undersized so that the catches were dominated by juvenile and immature fish which directly reduced the recruitment rate of the fisheries. These are some of the lessons that can help the Department of

Fisheries to properly manage cage culture in Lake Malawi and future investment in other water bodies in the country. Just as it affected the wild stocks, open access to the cage culture industry could have drastic impacts on the environment and the industry itself. Firstly, open access would promote overcrowding of fish cages in culture sites which could drastically reduce water currents through the cages and increase sedimentation of cage wastes in the vicinity of the cages. For example, cage culture in a 244 km^2 Taal Lake, Philippines, has an almost "open access" entry because local residents are allowed by law to own at most 5 cages as wealthy non-residents intrude and hire local residents as caretakers and install illegal cages in the lake (Vista et al., 2006). In 2001, the number of 10x10x6m cages in the lake rose to 7,433 and discharged large amounts of wastes which caused serious water quality problems and frequent massive fish kills in cage areas which affected both reared and wild fish stocks (Vista et al., 2006). Secondly, open access would promote poor cage management in form of intensive feeding to hasten growth, high stocking densities, use of fast-growing exotic species and theft. Intensive feeding and high stocking density would lead to high nutrient losses from the cages, deterioration of water quality and frequent occurrences of fish kills which would affect both the reared and wild fish including the wild chambo fisheries currently under restoration. Thirdly, as the number of cages increases due to open access, the demand for feed, fishmeal and fish oil would increase proportionately. Fishmeal and fish oil are currently imported from South Africa but, as the demand increases, local catches which form the main source of dietary protein in Malawi households may be diverted to fishmeal production. In addition, the demand for fishmeal may lead to over-exploitation of the already staggering wild stocks in the lakes. In an uncontrolled cage culture,

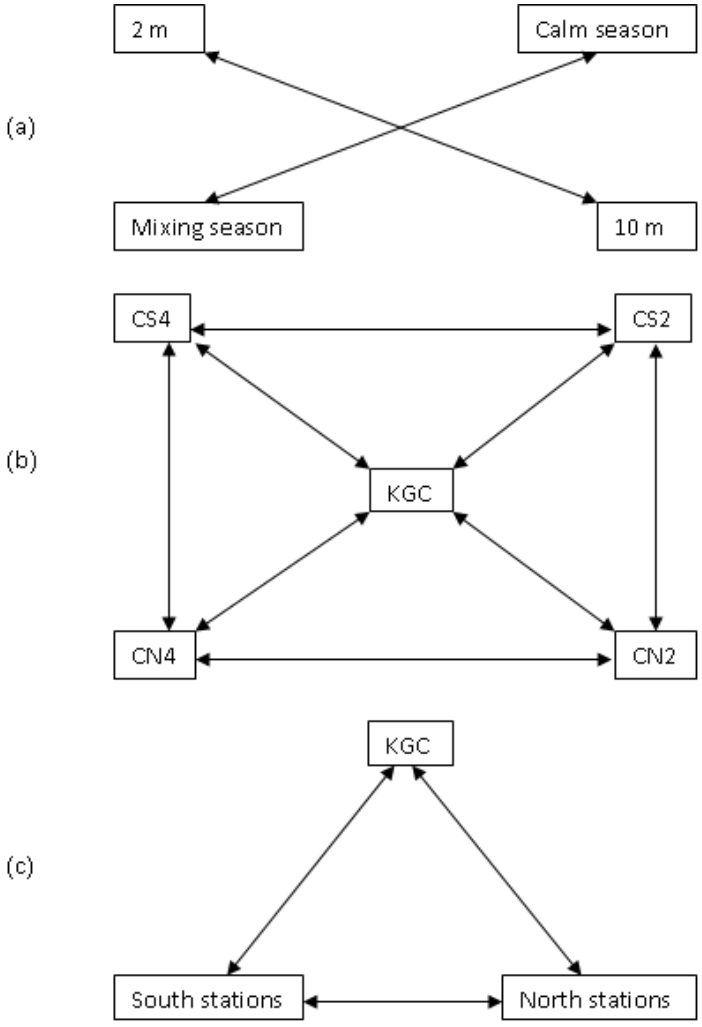
the threat of exotic species always exists, although introductions of exotic species are prohibited in Lake Malawi. Introduction of exotic species could affect wild and farmed fish through biological pollution in form of hybridization (i.e., loss of gene pools), spread of pathogens, competition with indigenous species for resources such as food and breeding grounds and predation on small indigenous species as experienced in Lake Victoria with the Nile Perch. Similar experiences have been documented in Lakes Malawi and Malombe involving an Asian gastropod introduced into the lakes about 25 years ago (Genner et al., 2004; Michel et al., 2008). The exotic gastropod, which has explosively multiplied in Lake Malombe, is partly blamed for the collapse of indigenous gastropods and herbivorous fish species including chambo (*Oreochromis*) and kambuzi (small demersal haplochromines) fisheries due to competition for food resources and breeding grounds.

It follows from the discussion above that the success of cage culture in Lake Malawi and future fish farms in other water bodies in the country is primarily the responsibility of the Department of Fisheries. It will require good and dedicated leadership and expertise of the department to properly guide the future of the industry in the country by making use of the many lessons gained from the collapsed capture fisheries to the cage culture industry. There will be need to limit entries into the industry to avoid the many problems that may come with "open access" cage culture industry. It is important to realize that the main constraint to the development of commercial aquaculture in Malawi and sub-Saharan Africa in general is the lack of locally produced, low-cost and high quality feed (Blow & Leonard, 2007). As soon as high quality fish feeds become readily available and cheap, pond and cage aquaculture investments

will bloom in Africa and in Malawi in particular. Limited entry will make sure that permitted cage farms are properly sited and monitored for efficient dispersion and assimilation of cage wastes.

Appendix 1

A flow chart of statistical analysis of data for chapter 3



At each station, data from 2 and 10 m depths were tested for differences using Student t-test at alpha of 0.05 (a). If no significant difference was indicated, the 2 and 10 m data at each station

were combined and tested for differences between calm and mixing seasons (e.g., CS4-calm vs CS4-mixing) (a). Seasonal data at each station were compared against data for similar seasons at other stations along the study transect as indicated by the arrows in (b) (e.g., CS4-calm vs CS2-calm, CS4-mixing vs CS2-mixing). Station data were combined again into south stations, KGC and north stations groups at seasonal and annual (data for both seasons combined) levels and tested again for differences as shown in (c). Because all the statistical tests indicated no significant differences, data were plotted as means (per given sampling day) for south stations, KGC and north stations. On any given sampling day, data from 2 and 10 m depths and from stations in the same direction (south, KGC or north) were treated as replicates and a mean was calculated.

Appendix 2
Estimation of ecosystem support areas for fish cage aquaculture in Lake Malawi

a) Dissolved oxygen production and consumption

Estimating total area required for photosynthetic production of dissolved O_2 needed for fish respiration and biological oxygen demand (BOD) around the fish farm in Lake Malawi. This estimation is based on a worst-case-scenario assumption for fish production in cages of a prolonged calm weather condition over the lake during which time O_2 exchange between the atmosphere and the water would be at its lowest rate. During this period of prolonged calm weather, photosynthetic O_2 production would dominate O_2 supply in the epilimnion.

- Tilapia fish have a respiration rate of 0.2 g O_2 kg^{-1} h^{-1} (= 4.8 g O_2 kg^{-1} d^{-1}) (Beveridge, 1991).
- Average fish density in cages in Lake Malawi = 75.85 kg m^{-2}
- Therefore, O_2 consumption by fish = 4.8 g O_2 kg^{-1} d^{-1} x 75.85 kg m^{-2}
  = 363.94 g O_2 m^{-2} d^{-1}
- Primary production in Lake Malawi has been estimated at 0.74 g C m^{-2} d^{-1} (Degnbol, 1993).
- For each gram of C fixed in photosynthesis, 2.6 g O_2 is produced (Pruder, 1986).
- O_2 produced in photosynthesis in Lake Malawi = 2.6 x 0.74 g m^{-2} d^{-1}
  = 1.924 g m^{-2} d^{-1}

- Therefore, the area needed around cages for fish respiration

  $= 363.94 \text{ g } O_2 \text{ m}^{-2} \text{ d}^{-1} / 1.924 \text{ g } O_2 \text{ m}^{-2} \text{ d}^{-1} = 189.16$

  The area needed to produce O_2 for respiration should be 189.16 times larger than the fish farm itself.

- A feed input of 0.515 kg m^{-2} d^{-1} into Lake Malawi and an FCR of 2.7 would result in a BOD of approximately 250 g m^{-2} d^{-1} (see Berg et al., 1996)

- The size of the area for BOD $= 250 \text{ g } O_2 \text{ m}^{-2} \text{ d}^{-1} / 1.924 \text{ g } O_2 \text{ m}^{-2} \text{ d}^{-1}$

  $= 129.94$

  The area needed to produce O_2 for BOD should be 129.94 times larger than the fish farm itself.

- At 3,000 fish production capacity, the size of the farm will be

  $= 3{,}000000 \text{ kg} / 75.85 \text{ kg m}^{-2}$

  $= 39{,}552 \text{ m}^2$

- Total area required for O_2 production would be

  $= 39{,}552 \text{ m}^2 \text{ x } (189.16 + 129.94)$

  $= 12.6 \text{ km}^2$

b) Phosphorus assimilation

To determine the area requires for the assimilation of P and N at the current primary production of 0.74 g C m^{-2} d^{-1}, I used total fish biomass gain and total feed added in cages between 01st January and 31st December, 2007.

- Total fish biomass gain

  = Final weight – Initial weight = 326,159.60-104,253.62 kg fish

  = 221,905.98 kg fish

  where Final weight is biomass at harvest within the year or on 31st December, 2007 while Initial weight is biomass on 01st January or at stocking within the year.

- Total feed supply in 2007= 446,355.70 kg feed
- FCR in 2007 = 2.01
- Total P in feed = 1.2/100*446,355.70*2.29 = 12,265.85 kg P
- Total P in fish biomass = 0.34/100*221,905.98 kg fish*2.29 = 1,727.76 kg P
- P loading into the environment in 2007 = 88.7/100*12.265.85 kg P = 10,879.81 kg P

  = 49.03 kg P/tone fish

  = 29.81 kg P d^{-1}
- 29.81 kg P d^{-1} was released from 29 cages with total area of 4921.78 m^2 = 0.0049 km^2
- According to the Redfield C/P ratio of 40:1, the current primary productivity of 0.74 g C m^{-2} d^{-1} is equivalent to an assimilation of about 18.5 mg P m^{-2} d^{-1}.
- Assimilation of 29.81 kg P discharged from the cages per day will required an area

  = (29.81*10^6 mg P d^{-1})/(18.5 mg P m^{-2} d^{-1}) = 1.61 km^2
- This area is 327 times bigger than the total area of the 29 cages.
- At fish production capacity of 3,000 tonnes the total area required for the assimilation of P loading will be equal to 13.97 km^2 assuming PP in the lake remains at the current rate.

Appendix 3
Farmed fish in cages

This appendix is a video file (13.53 minutes long in "wmv" format) of farmed fish in cages and shows the farmed *Oreochromis karongae* and *O. shiranus* fish species in cages in Lake Malawi. The file name of this video file is "FarmedFish.wmv".

This video file shows the fish species cultured in cages as compared to the wild fish which aggregate around the cages in video files in Appendices 4 and 5 below. This video file also shows the density of the farmed fish inside the cages which may be compared to the abundance of wild fish aggregating around the cages in Appendices 4 and 5 below.

Appendix 4
Wild fish abundance around cages

This appendix is a video file (14.15 minutes long in "wmv" format) of wild fish and shows their abundance around fish cages in Lake Malawi, Africa. The file name of this video file is "WildFishAbundance.wmv".

Consumption of cage wastes by the wild fish which aggregate around fish cages helps to reduce negative impacts of the wastes on the surrounding environment. The higher the abundance of wild fish around fish cages, the lesser the impact cage wastes will have on the environment surrounding the fish cages.

Appendix 5
Wild fish abundance and distribution around cages

This appendix is a video file (10.13 minutes long in "wmv" format) of wild fish around cages and shows their abundance and vertical distribution around fish cages in Lake Malawi, Africa. The file name of this video file is "WildFishDistribution.wmv".

Consumption of cage wastes by the wild fish which aggregate around fish cages helps to reduce negative impacts of the wastes on the surrounding environment. The higher the abundance of wild fish around fish cages, the lesser the impact cage wastes will have on the environment surrounding the fish cages.

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