Impacts of cage aquaculture on the farm dam ecosystem and its use as a multipurpose resource: Implications for irrigation

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Thesis presented in partial fulfilment of the requirements for the degree of

Master of Science at the University of Stellenbosch

December 2007

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DECLARATION

I, the undersigned, hereby declare that the work contained in this thesis is my own original work and that I have not previously in its entirety or in part submitted it at any university for a degree.

Signature: …….

Date: …….

ABSTRACT

Small farm dams (< 20 ha) in the Western Cape Province provide adequate water conditions for intensive cage production of rainbow trout (*Oncorhynchus mykiss*). A major environmental concern of cage aquaculture, however, is the high inputs of nutrients via commercial diets and the subsequent eutrophication of the water source. Eutrophication can result in the degradation of the general water quality (increasing pH levels, oxygen depletion, increased hydrogen sulphide and free ammonia) and shifts in the phytoplankton structure (increased biomass, single species dominance). Deterioration of water quality will affect the success of the fish farming enterprise as well as the performance of irrigation equipment by increasing the risk of clogging and corrosion. Water quality, phytoplankton and zooplankton compositions were monitored at four sites from June 2005 to November 2006 to determine the effects of cage culture on the farm dam environment, its associated biota as well as irrigation water quality. The distribution of nutrients, nitrogen and phosphorus, was mainly influenced by the stratification and mixing regime of the water bodies. Nutrient concentrations increased during the winter mixing period while in the summer months, they seem to settle to the lower part of the water column. Nutrient concentrations of production sites and reference sites were comparable except for the ammonia levels that were significantly higher at the production sites. Phytoplankton corresponded with nutrient availability resulting in high biomass during winter. In terms of biomass, phytoplankton was approximately two times more abundant in production sites compared to reference sites. Assemblage dominance by cyanophytes (*Anabaena circinalis*, *Microcystis* spp.) was found more often in production sites, while reference sites were dominated by dinophytes (*Ceratium hirundinella*, *Peridinium* spp.). Zooplankton biomass concurred with high phytoplankton biomass in winter. Zooplankton assemblages in production sites sustained much higher biomass. Effects of cage culture on irrigation water quality are evident from increased algal biomass and shifts in species composition. These results indicated that at its present production level, cage culture had impacts on the farm dam environment and irrigation water quality. The most significant evidence was given by increased plankton biomass and single species dominance in production sites. However, these findings can not solely be ascribed to the introduction of aquaculture as various other factors may also contribute to the water quality of these ecosystems.

OPSOMMING

Water toestande in besproeiingsdamme van die Wes Kaap kan geskik wees vir die produksie van reënboog forel (*Oncorhynchus mykiss*). Tydens die produksieseisoen word groot hoeveelhede fosfate en nitrate in die waterkolom vrygestel. 'n Oorvloed van voedingstowwe versnel die eutrofikasie tempo en dit kan gepaard gaan met 'n verlaging in die waterkwaliteit van die bron. Nadelige effekte kan sigbaar wees as variasies in pH en opgeloste suurstofvlakke, hoër totale ammonia konsentrasies asook 'n toename in die fitoplankton biomassa. Waterkwaliteit van substandaardgehalte kan beide die gehalte van produksie sowel as die meganika van besproeiingstelsels belemmer. Waterkwaliteit, fitoplankton en soöplankton gemeenskappe is tydens Junie 2005 tot November 2006 gemonitor om te bepaal of akwakultuur die waterkwaliteit en verskeie biologiese aspekte van die ekosisteem beïnvloed. Die ruimtelike en tydelike verspreiding van voedingstowwe is hoofsaaklik deur stratifikasie en destrafikasie in die water beïnvloed. Tydens somermaande was die voedingstowwe in die dieper sones gekonsentreer, terwyl die wintermaande gekenmerk was deur hoër konsentrasies wat meer eweredig versprei was. Daar was geen merkbare verskil tussen die voedingstof konsentrasies van produksie en kontrole-areas nie, behalwe totale ammonia konsentrasies wat aansienlik hoër in die produksie-areas was. Fitoplankton en soöplankton produksie se hoogste waardes is tydens die wintermaande gemeet. Produksie-areas het 'n hoër fitoplankton en soöplankton biomassa as kontrole-areas onderhou. Die fitoplankton-gemeenskap in produksie-areas is gekenmerk deur sianobakterieë (*Anabaena circinalis, Microcystis spp.*) dominansie. Dinoflagellate, veral *Peridinium* en *Ceratium,* was die dominante fitoplankton in die kontrole-areas. Die impak van akwakultuur op die besproeiingswaterkwaliteit was sigbaar in die toename van fitoplankton biomassa, asook in veranderings in die fitoplankton gemeenskapstruktuur. Die studie het aangedui dat die huidige vlak van produksie wel die ekosisteem en die kwaliteit van besproeiingswater beïnvloed het. Akwakultuur kan egter nie beskou word as die enigste oorsaak van veranderinge in waterkwaliteit nie, aangesien verskeie faktore die waterkwaliteit van hierdie ekosisteme kan beïnvloed.

ACKNOWLEDGEMENTS

I would like to express my gratitude to my supervisor, Dr A Leslie for her guidance on this project and careful comments throughout the text. I wish to thank the Water Research Commission (WRC) for the financial support without which the research would not have been possible. My thanks to our project leader, Mr K Salie for introducing me to the project and for his assistance throughout the study. A very special thanks goes out to Monika Cermak for her indispensable help and interest during the study, and for her constructive criticism of the manuscript. The farmers who participated in the case studies are gratefully acknowledged for information given and access to their farm dams. Sincere thanks to my family and friends for their endless patience and support over the course of the study.

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CHAPTER 1

GENERAL INTRODUCTION AND PROJECT OBJECTIVES

1.1 Background to the study

Aquaculture can be described as the beneficial and sustainable use of water for the cultivation and harvesting of aquatic species (e.g. finfish, shellfish, aquatic plants) for commercial consumption (DWAF, 1996b; Rouhani & Britz, 2004). Aquaculture activities have been practised for centuries, where it contributed to the aquatic food supplies of rural, food-deficit areas of the world. Recently, the rising concerns about the overexploitation of natural fisheries resources has shifted the emphasis towards aquaculture as a possible alternative for the production of aquatic species and has developed into a major industry throughout the world. According to Food and Agricultural Organization (FAO) statistics, global aquaculture production has grown at an annual rate of 8.8 % since 1970 and in 2005 the total production (inland & marine) was reported to be 47.8 million tons (FAO, 2007).

South Africa is still only a marginal contributor to world aquaculture production. In 1998, the total South African contribution to world aquaculture production was 5301 tons (ZAR 228.986 m) with koi carp (*Cyprinus carpio*) and rainbow trout (*Oncorhynchus mykiss*) production as the major sectors (Bekker & Brown, 1995; Hoffman *et al*., 2000). The trout industry experienced a stabilising period during the mid-nineties, but slowly increased from 1000 tons per annum to the current level of 1650 tons in 1998 (Hoffman *et al*., 2000). Currently, trout production in South Africa comprises of sport and recreational fisheries in the provinces of Mpumalanga, Kwazulu-Natal and the Eastern Cape and small-scale cage farming in the Western Cape Province (Rouhani & Britz, 2004). The conditions of water resources in the Western Cape favour the production of rainbow trout in cage systems and makes use of existing waterbodies such as on-farm irrigation dams and storage reservoirs. These cage systems require a low capital outlay and relatively simple technology, making this type of production system popular with rural communities and farm workers (Moloby, 2001).

On a local scale aquaculture offers rural communities and farm workers the opportunity to earn additional income, aiding in poverty alleviation and socio-economic improvement. However, the aquaculture industry in South Africa is currently experiencing various problems that inhibit the growth of the industry. Difficulties include the lack of extension services and technology, insufficient capital and financial support, limited access to international markets and a lack of food quality systems that are required by first world markets. Another major drawback for the industry is the lack of expertise and skills, as most of the enterprises are run by community members without the necessary knowledge and expertise (Hecht, 2000; Rouhani & Britz, 2004; Berold, 2005). The potential growth of freshwater aquaculture in South Africa is also constrained by the natural environment. The major constraints are the scarcity of suitable water resources, fluctuations in seasonal temperatures and variability in rainfall (Hecht, 2000; Rouhani & Britz, 2004; Berold, 2005).

In South Africa, surface waters are relied on as a source for urban, industrial and agricultural water demands. Climatically South Africa is described as a dry country with large semi-arid and hyper-arid regions and only a few humid areas. South Africa receives an annual rainfall of less than 500 mm of which the distribution is highly seasonal and unpredictable (Davies & Day, 1998). The growing human population and associated demands can not rely on available groundwater and existing surface resources alone (Davies & Day, 1998). With the current emphasis on climate change, users of inland water resources are encouraged to conserve water resources and use water more sparingly. To overcome the seasonal variability of water, large reservoirs and farm dams have been constructed to collect and store water in large enough quantities to ensure a sufficient supply to meet agricultural, industrial and domestic demands. This has given rise to the multiple usage of available water resources, such as for example, the integration of aquaculture into existing irrigation dams (Fernando & Halwart, 2000; Ingram *et al*., 2000).

Farm dams are artificial structures constructed to accumulate and store runoff water in order to meet agricultural demands. They are intended to store and divert water for various purposes including irrigation, livestock watering, human consumption and aquaculture. These small storage dams are most densely distributed in KwaZulu-Natal and the Western Cape Province, where they are primarily used for irrigation during summer months. According to the Dam Safety Regulations of the South African National Water Act (Act 36 of 1998, Section 117), farm dams with a capacity exceeding 50 000 m^3 and a dam wall of higher than 5 m need to be registered at the Department of Water Affairs and Forestry (Davies & Day, 1998). Currently, the Western Cape Province, including the Berg, Palmiet, Riviersonderend and Eerste River basins, hosts over 4000 farm dams with a total storage volume of 100 million m^3 (Berg *et al*., 1994). The hydrodynamics of farm dams are unique as it is both influenced by natural events and human activities. These systems gain water via precipitation and runoff or are supplied by water pumped from nearby rivers. Water exits these systems by evaporation, seepage and during extraction for irrigation application (Brainwood *et al*., 2004). Farm dams are often viewed negatively as they interfere with natural stream flow and the water stored is subjected to high evaporation rates. However, these artificial permanent reservoirs create a new kind of aquatic ecosystem that sustains a unique set of trophic interactions between aquatic plants, animals and waterbirds (Davies & Day, 1998).

South Africa is highly dependent on reservoir water. Surprisingly however, data on water quality of surface water in South Africa are mainly restricted to large reservoirs and lotic systems. Limnological research of privately owned farm dams in South Africa is still left unexploited (Hart & Hart, 2006). A program by the Council for Scientific and Industrial Research (CSIR), The Inland Water Ecosystems National Program, was initiated to conduct research on selected large artificial reservoirs. The program, however, collapsed and the focus was turned to river ecosystems (Hart, 1992). In the late 1970's and 1980's, research addressed the extent of eutrophication within major reservoirs in South Africa (Steyn *et al*., 1975; Toerien, 1975; Toerien *et al*., 1975; Steyn & Toerien, 1976; Grobler & Silberbauer, 1985). An examination of 64 man-made dams found that 75% of the dams could be

regarded as enriched, 10% being hypereutrophic (Thornton, 1987). In 1980, the then Department of Water Affairs, initiated a management plan as part of a National Eutrophication Monitoring Programme (NEMP) to reduce the high total phosphorous levels in South African surface waters. This strategy stated that wastewater or effluent should not contain soluble orthophosphate (as P) in a concentration higher than 1 mg/L (Van Ginkel *et al*., 2000). Another project, The Trophic Status Project, was initiated in 1990 to determine the trophic status based on chlorophyll *a* concentrations, total phosphorous levels, transparency and cyanobacterial presence (Van Ginkel *et al*., 2000). Evaluation of the 1 mg/L P standard implementation found no significant change in the trophic status of the selected reservoirs. The 1 mg/L P standard implementation did however have significant effects on the reduction of phosphorous concentrations in Bon Accord Dam, Hartbeespoort Dam and Rietvlei Dam (Van Ginkel *et al*., 2000). Currently, there is no national monitoring programme in place for privately owned farm dams and data on water quality of farm dams are lacking. Water quality of farm dams is influenced by numerous factors, including: geographic and climatic conditions, regional geology, basin morphology and surrounding land use (Brainwood *et al*., 2004). Agricultural activities surrounding farm dams often involve the application of nutrient rich fertilisers and pesticides to crops and orchards. During the winter months these substances are leached from the soil, thereby aggravating nutrient pollution of farm dams (Boaventura *et al*., 1997; Schulz *et al*., 2001; Brainwood *et al*., 2004).

Aquaculture activities pose a number of associated environmental problems. For example: environmental pollution and physical change of the aquatic ecosystem, seed collection from wild resources, disease spreading to wild populations, genetic contamination, and dependence of feed derived from natural stocks (Beveridge, 1996; Davenport *et al*., 2003; Pillay, 2004). Cage culturing of carnivorous species, such as *Oncorhynchus mykiss*, calls for large inputs of external nutrition during the production season. During cage fish farming, the culture species are confined in net-cages that are suspended from flotation structures and waste products produced (particulate and soluble) enter the water column directly (Phillips *et al*., 1985; Stirling & Dey, 1990; Cornel & Whoriskey, 1993;). Since fish feed and excreta are rich in nutrients (nitrogen and phosphorous), cage aquaculture poses the risk of increasing the rate at which cultural eutrophication will take place.

The small volume and low flushing rates of farm dams make them more susceptible to rapid eutrophication. Fish farming waste that is deposited will remain longer in the vicinity of the cages and eventually settle in the underlying sediment, causing more rapid deterioration of water quality than in large reservoirs. Problems associated with poor water quality include the growth of nuisance aquatic plant material, increased nutrient levels and internal cycling, release of toxic substances from bottom sediments, increased algal biomass, fluctuations in levels of dissolved oxygen and pH, cyanobacterial toxin production, increased turbidity and decreased species richness of plankton assemblages (Stirling & Dey, 1990; Cornel & Whoriskey, 1993).

Phytoplankton assemblages respond rapidly to changes in water quality and are often used as indicators for assessing aquatic health. During enrichment of waterbodies, phytoplankton assemblages undergo changes in terms of biomass and species richness. It is well documented in the literature that algal biomass increases with enrichment and changes from a stable community with a high degree of species diversity to a less diverse community that is dominated by a single species (Harding & Paxton, 2001). This can give rise to the development of noxious blooms of cyanobacteria with implications to the fish farming enterprise. These organisms are able to produce substances (geosmin and 2-methylisoborneol) that cause off-flavours in the culture organism, making it unacceptable for the consumer market (Wnorowski, 1993; Robertson *et al*, 2006). Fluctuations in the dissolved oxygen concentrations and pH during these cyanobacterial blooms and subsequent die-offs, can also cause physiological stress to the culture species. Increased loads of organic wastes from fish farming, cyanobacterial and algal bloom die-offs require large amounts of oxygen during bacterial decomposition at the sediment-water interface (Boyd *et al*., 1975; Boyd *et al*., 1978; Erez *et al*., 1990). Once anoxic conditions develop in the bottom waters, toxic compounds such as ammonia, nitrite and hydrogen sulphide are resuspended from the sediment (Mortimer, 1941). A sudden mixing or upward development of anoxic water and the subsequent distribution of toxic components to surface layers, could be detrimental to the health and growth of the cultured species.

The primary motivation for the construction of farm dams is to fulfil a multipurpose role of which irrigation is the most important application in the Western Cape Province. The performance of irrigation equipment is strongly subjected to the quality of the water resource and can therefore be seriously impacted by the introduction of fish farming. Increased algal growth due to enrichment can cause clogging of filters, emitters and sprinklers, which necessitate more frequent back washing of filters (Bucks *et al*., 1979). High decomposition rates in bottom waters will deplete oxygen reserves and create anoxic conditions. Anoxic conditions in bottom water will favour the release of iron and manganese, increasing the risk of emitter clogging (Mortimer, 1941; Nakayama & Bucks, 1991). Fluctuations in pH between alkaline and acid conditions can also determine the likelihood of water to act as a corrosive towards irrigation equipment or to cause the precipitation of calcium and magnesium carbonates. (Koegelenberg *et al*., 2002).

To make aquaculture an economically viable enterprise for the fish farmer in terms of fish production and irrigation activities, it needs to continue in an environmentally friendly manner. From an ecological point of view, the nature of the aquaculture activity can have short-term effects in the vicinity of the cages, but also possible long-term effects on all trophic levels. Due to the closed character of farm dams, any disruptions in the ecology of the water body can pose serious threats for fish farming and irrigation water quality. The continuation and expansion of future aquaculture projects will ultimately rely on the water quality and its suitability for the production species as well as for irrigation requirements. It is evident that the enrichment of farm dams could not only be detrimental to the natural ecological balance within the dam but also for the economic success of the fish farming project and irrigation performance. Aquaculture therefore needs to proceed in an environmentally conscious

manner to ensure the success of future fish farming enterprises without jeopardising water quality. It is therefore essential to gain information on potential changes in water chemistry and ecology in farm dams that are used for the integration of aquaculture and irrigation.

1.2 Project objectives

Although numerous studies on the seasonal changes in water quality, phytoplankton and zooplankton have been documented, little research has been carried out on water quality monitoring and plankton dynamics in enclosed farm dams used for fish farming.

The primary aims of this study were:

- to assess changes in the water quality status of two production dams during net-cage production of rainbow trout (*Oncorhynchus mykiss*)
- to assess and compare the water quality status of two non-production dams, subjected to similar environmental conditions with that of the two production dam
- to provide results on phytoplankton and zooplankton composition, diversity measures and biomass in farm dams containing aquaculture
- to determine the extent to which aquaculture activities affect water quality in terms of irrigation requirements

1.3 Approach used in this study

The first step of the study was to gain knowledge on limnological processes in closed farm dams. The second step was to identify water quality factors that could be affected by cage aquaculture and ultimately result in ecosystem degradation and production losses. Selected water quality parameters and plankton dynamics were then monitored to gain baseline data. Each chapter of the study includes a literature review. This is followed by a brief summary of materials and methods used and a section discussing the results. Final conclusions are presented at the end of each chapter.

CHAPTER 2

STUDY SITES & GENERAL METHODOLOGY

2.1 Study Sites

2.1.1 Location and description of study sites

The present study involved the investigation of four study sites, all situated in the Stellenbosch region of the Western Cape Province of South Africa. The study sites consisted of two farm dams containing net-cage aquaculture and two farm dams without fish farming. Study sites containing production cages were Nietvoorbij Dam at the Nietvoorbij Research Station (*Production site 1*: S33°55'4"; EO18°51'47"; Figure 2.1) and John Smith Dam at Rustenburg Wine Estate (*Production site 2*: S33°53'59"; EO18°53'0.7"; Figure 2.2). The other two study sites, Poplar Dam (*Reference site 1*: S33°53'37"; EO18°53'15", Figure 2.3) and Garden Dam (*Reference site 2*: S33°54'4"; EO18°53'0.9"; Figure 2.4) also form part of Rustenburg Wine Estate.

Figure 2.1: Aerial view of Nietvoorbij Dam at Nietvoorbij research Station (Production site 1)

Figure 2.2: Aerial view of John Smith Dam at Rustenburg Wine Estate (Production site 2)

Figure 2.3: Aerial view of Poplar Dam at Rustenburg Wine Estate (Reference site 1)

Figure 2.4: Aerial view of Garden Dam at Rustenburg Wine Estate (Reference site 2)

The dams were located in close proximity to minimise differences in groundwater, catchment geology and weather related variables such as temperature, volume of rainfall and evaporation rates. The dams chosen for the study are used for irrigation purposes in the summer months, and for livestock watering and aquaculture. The area immediately surrounding the dams included agricultural land with primarily vineyards and pastures for cattle grazing.

The youngest dam, production site 1, was constructed in 1978 and received structural changes in 1985 (Table 2.1). Production dam 1 was the largest of the four dams with an area of 2.3 ha, a mean depth of 8.7 m and a storage capacity of 209 000 m^3 (Table 2.1). This dam relied primarily on runoff from surrounding agricultural land and from water pumped from the Plankenbrug river. Each year during spring and summer, a large water bird population (> 450 individuals) inhabited the dam and surrounding banks. Water bird species included Egyptian goose (*Alopochen aegyptiacus*) and Cormorants (*Phalacrocorax africanus*) (Sinclair *et al*., 2002). Aquaculture activities in production site 1 commence annually in May and continue until November (Table 2.2). Reference site 1 was built in 1965 and covered an area of 1.8 ha with a mean depth of 6.8 m and a storage capacity of 122 945 m^3 (Table 2.1). Water resources for this dam included runoff from adjacent hillsides and water pumped from a nearby river. The banks of this dam supported numerous Poplar trees (*Populus* spp.). The oldest of the dams, production site 2, was constructed in 1946 and structural renovations were carried out in 1993. As Table 2.1 shows, this dam was also the smallest and covered an area of 1.0 ha. It was predominantly fed by runoff and by water pumped from a nearby river. The annual production season in production site 2 begin in May and carry on until November (Table 2.2). Reference site 2 was constructed in 1960 and underwent

structural changes in 1978. It covered and area of 1.5 ha with a mean depth of 4 m and a storage volume of 60 000 m^3 (Table 2.1). Water resources for this dam included runoff as well as water that was pumped from a nearby river. Reference site 2 was frequently visited by cattle from the surrounding pastures which had an impact on the condition of the topsoil around the dam.

Table 2.1 **Summary of morphometric features of study sites. Brackets indicate years when reconstruction or structural improvement took place**

	Production sites		**Reference sites**	
	Nietvoorbij Dam	**John Smith Dam**	**Poplar Dam**	**Garden Dam**
Year of construction	1978 (1985)	1946 (1993)	1965	1960 (1978)
Surface area (m^2)	23 978 (2.3 ha)	10 368 (1.0 ha)	18 000 (1.8 ha)	15 000 (1.5 ha)
Capacity (m^3)	209 000	76 000	122 945	60 000
Mean depth (m)	8.7	7.0	6.8	4
Elevation (m)	148	424	255	275
Water supply	Plankenbrug river, runoff	runoff, river	runoff, river	runoff, river
Surrounding land use	Vineyards	Vineyards	Vineyards	Vineyards, pastures
Resource utilisation	Irrigation Aquaculture	Irrigation Aquaculture Livestock watering	Irrigation	Irrigation, Livestock watering

2.1.2 Climate

Meteorological data for the entire duration of the study were obtained from respective regional stations of the South African Weather Services (SAWS, 2006). Data obtained were rainfall measurements (mm), daily maximum and minimum temperatures (°C) and cloud cover (oktas). Own observations were recorded during sampling events and those parameters included cloud cover, wind action and precipitation.

The prevailing climate of the region is a Mediterranean type with cold wet winters and warm windy summer months (Davies & Day, 1998). Figure 2.5 indicates that the average air temperature during the entire study period ranged between 11.6 ± 2.8 °C and 25.8 ± 3.5 °C. The warmest month was February 2006 with an average daily maximum of 32.8 ± 4.4 °C and the coldest month was August 2005 with an average daily minimum of 7.0 ± 2.9 °C (Figure 2.5).

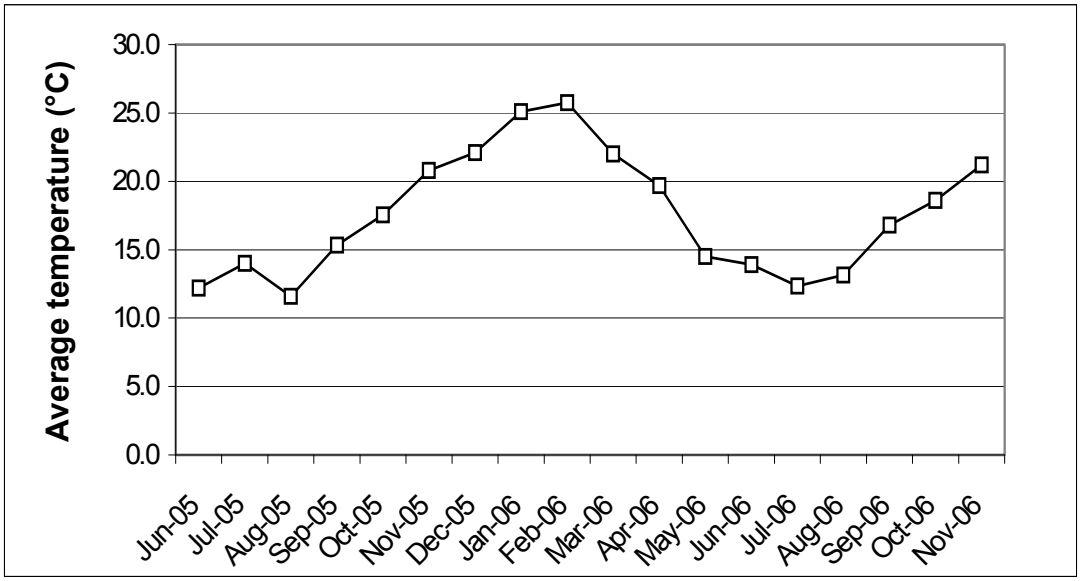


Figure 2.5: **Average air temperature recorded at the South African Weather Service station in Paarl from June 2005 to November 2006**

In terms of annual rainfall, the study sites were located in a typical winter rainfall region with the bulk of rain falling during the colder winter months (June – August) (Figure 2.6). The precipitation data collected showed characteristic local patterns with five different periods for the duration of the study. A dry period occurred from December 2005 to March 2006. A wet period from June 2005 to August 2005 and from May 2006 to August 2006. The highest monthly rainfall record of 145.7 mm was measured in May 2006. Additionally, two transition periods between the wet and dry seasons were experienced from September 2005 to November 2005 and from September 2006 to November 2006.

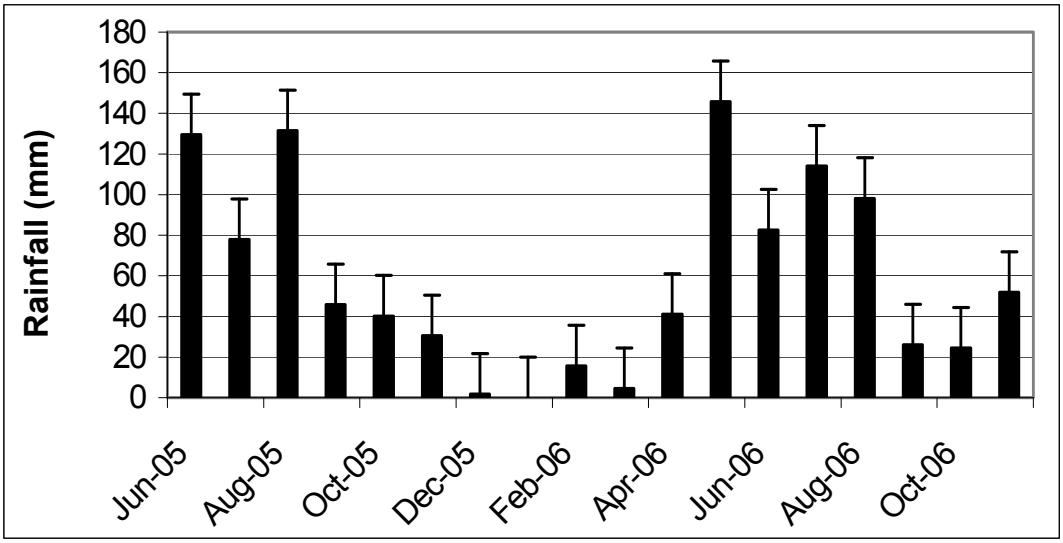


Figure 2.6: **Annual distribution of rainfall as measured by the South African Weather Service station located in Stellenbosch from June 2005 to November 2006**

2.1.3 Underlying geology

The reigning chemical characteristics of surface water are strongly influenced by the chemical weathering processes of the underlying geology of the catchment and basin. Day & King (1995) divided South African inland waters into four categories, by following the geographical distribution of the major ions and linking it to the underlying geological material. Geologically, the catchment is dominated by sedimentary rocks of the Cape Supergroup (*c*. 500 – 320 Myr), the heavily weathered Table Mountain Sandstone and shale of the Malmesbury Group (*c*. 950 – 600 Myr). Water flowing over the sedimentary rocks of the Cape Supergroup is characteristically low in nutrients and total dissolved solids (TDS < 1000 mg/L). Total dissolved solids in these waters are derived from rain, snow and other forms of precipitation. These farm dams are classified as "precipitation dominated" and the water is soft and pure, with sodium (Na^+) and chloride (Cl^-) as the most abundant ions (Day & King, 1995).

2.1.4 Cage aquaculture activities at production sites

Each farming enterprise consisted of two floating cages (each 100 m^2) extending from the water surface to a depth of 4 meters. The cage structures were situated in the deepest part of the dam and were accessed by means of a float. The net cages were surrounded by a wooden walkway that provided access to all cages and cages were secured to the bottom as well as to the sides of the dam by means of anchor ropes. Anti-predation nets surrounded the net-cage structure and the top of the cages was also covered to keep predatory animals and birds at bay.

The production season commenced in May as water temperature dropped to an optimum production temperature of < 18 °C and water levels increased. Production dams were stocked with juvenile trout ranging from 100 g - 250 g at an average density of 10 kg/m^3. Cultured fish were fed manually on a locally produced commercial fish feed. The daily amount of feed was calculated according to a feeding program that was based on average weight, stocking density and reigning water temperatures. A subsample of fish were captured, weighed and measured on a monthly basis in order to determine an average growth rate. The feeding programme was then adjusted accordingly. As water temperatures exceed 21 °C towards late spring (October), fish were harvested and sent to a local processing plant (Table 2.2).

Table 2.2 shows that the production dams were stocked at the end of May and beginning of June for the 2005 production season. In 2006 the dams reached optimal temperature conditions earlier and dams were stocked at the beginning of May. It is clear from Table 2.2 that the production season of 2005 delivered a higher fish biomass as opposed to production figures from 2006. Production site 2, in particular, experienced severe oxygen depletion in 2006 that resulted in the premature harvesting of fish.

Table 2.2: Fish production statistics per season during the study period

	2005		**2006**	
	Production site 1	**Production site 2**	**Production site 1**	**Production site 2**
Date of stocking	31 May 2005	2 June 2005	9 May 2006	10 May 2006
Date of harvesting	27 October 2005	17 October 2005	10 November 2006	6 October 2006
Biomass (kg) harvested	5917.1	5044.5	3111.6	3259.2

2.2 General methodology

2.2.1 Sampling stations and sampling frequency

At each study site a single off-shore sampling station was prepared to ensure that the sampling location remained consistent for the duration of the study. Manual soundings of depths were made by means of a weighted, calibrated line. The weight at the end of the line was fixed on a platform (1 m x 1 m) to minimise sinking into the bottom sediments (Lind, 1979; Wetzel & Likens, 2000). At each study site depth measurements were made along the longest transect over the length and width of the water body, supported by additional measurements. Data points were then used to draw a profile of each dam. The deepest region was chosen as the sampling station and was marked with a plastic buoy. During sampling events stations were accessed by means of a non motorised inflatable boat.

Study sites were visited at intervals of 14 days for a duration of 16 months from July 2005 to November 2006, during which time water samples were collected at each sampling station. Sample collection at each sampling station was conducted at more or less the same time of day during each sampling event and ranged between 10:00 and 16:00 hours. Sampling stations were sampled in the following sequence: Nietvoorbij Dam (production site 1), Garden Dam (reference site 2), Poplar Dam (reference site 1) and John Smith Dam (production site 2).

2.2.2 Sample collection and transportation

Water samples were collected at each sampling station at the surface (0 m) and at depths of 3 m, 6 m and near bottom (> 6 m). Plastic bottles with a 250 ml capacity were used to collect water samples. Prior to sampling, bottles were rinsed with water from the respective sampling site. Sampling bottles were filled to the top, leaving no headspace in order to prevent oxygenation and loss of volatile components (Cole, 1994; Wetzel & Likens, 2000). Water samples representing surface samples were collected 20 cm below the surface. For collection of samples from deeper depths, a 1.5 L capacity water sampler (The Science Source) was used. Immediately after collection, samples were placed in a cooler box with ice bricks for the duration of transportation to the laboratory. Upon reaching the laboratory, samples were stored at 4 °C until further analysis. All chemical and nutrient analyses were performed within 24 hours of sample collection (Cole, 1994; Hach, 1996; Wetzel & Likens, 2000; Hach, 2005).

2.2.3 Physical and chemical analyses

2.2.3.1 Physical and *chemical analyses*

Dissolved oxygen concentrations (mg/L) of surface water, 3 m, 6 m and > 6 m (depending on depth of dam) were determined on-site by using a portable oxygen meter (Oxyguard MKIII). Water transparency was measured using a standard secchi disk, 250 mm in diameter, painted in black and white quadrants (Wetzel & Likens, 2000). Other field measurements included vertical thermal profiles (°C), measured using a handheld Oxyguard MKIII meter (OxyGuard International) (Hargreaves & Tucker, 2002). At the laboratory unfiltered water samples from each site were analysed for the total suspended solids (mg/L) content using a Hach colorimeter (DR/700 & DR/890 Colorimeter) (Hach, 2005). Total dissolved solids (mg/L) and conductivity (µS/cm) were measured by means of a Hach CO 150 conductivity meter and pH measurements with a Hanna pH 211 microprocessor (Hach, 1996).

2.2.3.2 Nutrient analyses

Refrigerated water samples were left to reach room temperature before chemical analyses were performed. Prior to dissolved nutrient analysis, water samples were filtered through Sartorius cellulose nitrate filters, with a pore size of 45 µm. Filtered water samples were analysed for nitrate-nitrogen (NO_3-N), nitrite-nitrogen (NO_2-N), ammonia-nitrogen (NH_3-N) and orthophosphate (PO_4-P) using a Hach colorimeter (DR/700 & DR/890 Colorimeter). Unfiltered water samples were analysed for total phosphorous. Nutrient and chemical analyses were performed following the methods of the Hach water analysis handbook (Tables 2.3 and 2.4).

2.2.3.3 Trace elements

During the sampling events of December 2005 and July 2006, additional water samples were collected for trace element and a major inorganic determinants analyses. Samples were preserved by adding an $HgCl_2$ ampule before sending them to the Department of Water Affairs and Forestry for analyses.

2.2.4 Quality control

On four occasions additional water samples were sent to the Department of Water Affairs and Forestry, Pretoria, for physical and chemical analyses. Results were then compared to the results of this study to assure quality of analyses, methods and equipment used.

Table 2.3: Summary of physical parameters and analytical methods followed

PHYSICAL PARAMETERS AND ANALYTICAL METHODS EMPLOYED			
Parameter	**Unit**	**Analytical method**	**Reference**
Temperature	°C	Oxyguard MK III oxygen meter	
Turbidity	cm	0.25 m secchi disk	(Wetzel & Likens, 2000)
Total suspended solids	mg/L	Photometric method of determination at 810 nm with Hach DR/700 and DR/890	(Hach, 2005)

Table 2.4: **Summary of chemical parameters and analytical methods followed**

CHEMICAL PARAMETERS AND ANALYTICAL METHODS EMPLOYED			
Parameter	**Unit**	**Analytical method**	**Reference**
Dissolved Oxygen	mg/L	Oxyguard MK III oxygen meter	
pH	Standard unit	Hanna pH 211 microprocessor with automatic temperature compensation, one point calibration against pH 4 and 7.	Hanna pH 211 microprocessor Instruction Manual
Conductivity	µS/cm	Hach CO 150 Conductivity meter with automatic temperature compensation, using 25°C as reference temperature	(Hach, 1996)
Ammonia – N (NH_3-N)	mg/L	Nesslerisation method followed by colorimetric determination at 420 nm with Hach DR/700 Colorimeter Salicylate method followed by colorimetric determination with Hach DR/890 Colorimeter	(Hach, 2005) USEPA approved (Hach, 2005)
Nitrate – N (NO_3-N)	mg/L	Cadmium reduction method followed by colorimetric determination at 500 nm with Hach DR/700 and DR/890 Colorimeter	(Hach, 2005)
Nitrite – N (NO_2-N)	mg/L	Diazotisation method followed by colorimetric determination at 500 nm with Hach DR/700 and DR/890 Colorimeter	(Hach, 2005) USEPA approved
Ortho – P (PO_4-P)	mg/L	Molybdovanadate method followed by colorimetric determination at 810 nm with Hach DR/700 Colorimeter	(Hach, 2005) USEPA approved
Total P	mg/L	Acid Persulfate Digestion method followed by colorimetric determination of soluble reactive phosphorous	(Hach, 2005) USEPA approved
Total dissolved solids	mg/L	Hach CO 150 Conductivity meter with automatic temperature compensation, using 25°C as reference temperature	(Hach, 1996)

2.2.5 Phytoplankton

2.2.5.1 Sampling methodology and *preservation*

During routine biweekly monitoring at the off-shore sampling stations, phytoplankton samples were collected. Samples were collected in 250 ml plastic bottles at the surface and at depths of 3 m and 6 m respectively. Surface samples were collected just below the water surface, whereas a water sampler (1.5 L capacity) was lowered to collect water from the deeper layers. Phytoplankton samples were fixated in the field and Lugol's acetic solution (1 ml to 100 ml of sample) was added for preservation and dyeing of the planktonic material. One litre of Lugol acetic solution was prepared using 30 g Iodide, 100 g Potassium Iodide, 100 ml glacial acetic acid and 1 L of distilled water (Wetzel & Likens, 2000; Findlay & King, 2001). Samples were stored in a cool, dark place until identification and quantification was carried out.

2.2.5.2 Species identification and *quantification*

Samples were shaken vigorously to ensure proper mixing of settled material before decanting small volumes into self-constructed counting chambers (1 ml, 5 ml, and 10 ml). Counting chambers with sample aliquots were then allowed to settle in the chambers for at least 48 h (24 h/1 cm height) prior to analysis. After settling, cell counts and species identifications were performed using the Utermöhl inverted microscope technique (Lund *et al*., 1958). A Zeiss inverted microscope with a magnification ranging from 125x to 757.5x was used for inspection of phytoplankton samples. When an organism was dominant, further identification was undertaken to determine the species. The keys of Huber-Pestalozzi (1938), Huber-Pestalozzi (1941), Huber-Pestalozzi & Hustedt (1942), Huber-Pestalozzi (1950), Huber-Pestalozzi (1955), Huber-Pestalozzi (1961), Huber-Pestalozzi (1972), Ettl (1978), Prescott (1978), Rieth (1980), Förster (1982), Häusler (1982), Ettl (1983), Huber-Pestalozzi (1983), Kadlubowska (1984), Mrozinska (1985), Krammer & Lange-Bertalot (1986), Ettl & Gärtner (1988), Krammer & Lange-Bertalot (1988), Ettl (1990), Krammer & Lange-Bertalot (1991a), Krammer & Lange-Bertalot (1991b), Joska & Bolton (1994) and Van den Hoek *et al*., (1995) were used for identification.

After identification, individual cells, colonies and filaments were counted in transects to ensure that the entire sample was counted and recorded. Individual cells were compared to a basic geometrical shape that closely matched the cell shape (Hillebrand *et al*., 1999). Individual cell biovolumes were calculated by substituting measured cell dimensions into appropriate geometrical formulas. A minimum of 20 organisms was measured for each taxon present in the sample to determine individual biovolumes. After the determination of cell biovolumes, counts (cells and colonies, per ml) were transformed to biomass in mg per litre. Biomass was expressed as fresh weight, assuming the density of fresh algae to be 1 g/cm^3 (Wetzel & Likens, 2000).

2.2.6 Zooplankton

2.2.6.1 Sample collection and *preservation*

Water samples for macrozooplankton (cladocera and copepoda) determination were collected at 0 m, 3 m, 6 m and > 6 m depths (depending on the depth of the dam), by using a Schindler-Patalas plankton trap (10 L capacity). Additional samples were collected from water just above the sediment-water interface to account for vertical migrating zooplankton. Water samples from Schindler-Patalas plankton trap (10 L) were passed through a mesh with a pore size of 64 µm and concentrated to a final volume of 100 ml. Samples were preserved in the field by adding formaldehyde (final concentration of ± 5 %). In the laboratory, samples were left for 48 hours to settle before transferring them into a phenoxetol medium (Steedman Solution) for long-term preservation (Steedman, 1976). One liter of Steedman Solution was prepared by using 5 ml propylene phenoxetol, 45 ml propylene glycol and 950 ml distilled water (Steedman, 1976).

For the microzooplankton (protozoa and rotifera), water samples of 250 ml were collected from each study site. Surface samples were taken approximately 20 cm below the surface, whereas samples from deeper depths (3 m, 6 m and > 6 m) were collected using a water sampler (1.5 L capacity). Samples were preserved with a Lugol acetic solution to a final concentration of 1 %. Samples were stored in a cool, dark place until counting and identification commenced.

2.2.6.2 Species identification and *quantification*

For the identification of macrozooplankton species, the content of the samples was transferred into a Petri dish from where organisms were individually placed onto a slide for inspection. A Leitz compound microscope with a magnification ranging from 40x to 1000x was used for identification of the genera and where possible, species. Identifications were performed according to the keys of Davies & Day (1998), Thirion (1999), Day *et al*. (2000), Day *et al*. (2001a) and Day *et al*. (2001b). For quantitative analysis of macrozooplankton (cladocera and copepoda), samples were left overnight to settle. After settling, samples were concentrated to 50 ml by syphoning the upper liquid. The samples are then vigorously shaken to ensure that organisms were evenly mixed throughout the sample. Subsamples of 2 ml were transferred to a modified Bogorov counting tray and counted by means of a Leica stereomicroscope (6.3x to 50x magnification). A drop of Lugol solution was added to the subsamples to aid in the counting of organisms by staining them. Biomass estimates of zooplankton species were derived from length-weight relationships published by Hall *et al*. (1970), Dumont *et al*. (1975), Culver *et al*. (1985) and Wetzel & Likens (2000).

Microzooplankton (rotifera and protozoa) samples were counted and identified using a Zeiss inverted microscope and the Utermöhl inverted microscope technique (Lund *et al*., 1958). Sample aliquots of 1 ml, 5 ml and 10 ml were poured into self-constructed sedimentation chambers and left to settle for 48 hours. Identification of microzooplankton (rotifera and protozoa) were done to genus level and where possible to species level. The keys of Corliss (1979), Curds (1982), Patterson (1992), Foissner &

Berger (1996), Day & De Moor (2002) and Joska *et al*. (2005) were used for the identification of individual species. For community biomass determination, the entire content of the sedimentation chamber was counted and density expressed as cells/ml. Biovolumes of protozoa identified, were derived from Beaver & Crisman (1982). If biovolumes were not available, values were calculated by reducing organisms to geometric shapes and using formulas accordingly. Biovolumes were then converted to biomass by assuming the density of protozoa to be 1 g/cm^3 (Wetzel & Likens, 2000). Biomass of identified rotifera taxa were obtained from previously published values (Hall *et al.*, 1970; Dumont *et al*., 1975; Wetzel & Likens, 2000).

CHAPTER 3

ASSESSMENT OF CHANGES IN THE NUTRIENTS AND WATER CHEMISTRY IN TWO SETS OF FARM DAMS DURING NET CAGE PRODUCTION OF RAINBOW TROUT (*ONCORHYNCHUS MYKISS*)

3.1 Introduction

3.1.1 Farm dams as resource for production

Water quality is the essential requirement for the rearing of aquaculture species. The quality and quantity of the water resource determines to a great extent the success or failure of a fish farming enterprise. For the fish farmer, good water quality is of high importance as any deterioration of water quality can cause physiological stress to the culture species. Once an aquaculture operation commences it is logistically difficult to remove culture species from poor water quality conditions. Translocation also induces handling stress on the culture species that may lead to mortalities. Rainbow trout, as a culture species, requires a higher degree of good water quality when compared to the optimal requirements of other culture species (DWAF, 1996b; Moloby, 2001). Stress caused by poor water quality can cause massive fish kills, reduce growth rates and increase the susceptibility of culture organisms to diseases (Soderberg *et al*., 1983; Moloby, 2001). In terms of the long term sustainability of the enterprise, the maintenance of good water quality is crucial as poor water quality and associated algal growth could result in a substandard product and restrict future production (Robertson *et al*., 2006).

Water quality of inland water bodies are described in terms of the reigning chemical and physical characteristics of the resource. These characteristics are influenced by both natural processes and human activities in the surrounding catchment. Natural factors that influence the water quality of a water body include basin morphology, hydrology, landscape, degree of enrichment, parent material of the underlying rock and the reigning climatic conditions (Day & King, 1995; Smith *et al*., 1999; Brainwood *et al*., 2004).

3.1.2 Implications of cage aquaculture on nutrients

Fish farming produces large amounts of nutrient rich waste products that are added directly to the water and underlying sediment (Penczak *et al*., 1982; Stirling & Dey, 1990; Cornel & Whoriskey, 1993; Pechar, 2000; Temporetti *et al*., 2001). Waste products can be particulate or soluble. Particulate waste originates mainly from faecal matter, unconsumed fish feed and metabolic waste that falls through the cage structure. Lipids also often form a layer in the vicinity of cages following feeding. Dissolved products include nitrogen, phosphorous, and dissolved organic carbon and may be directly excreted, or dissolved from the feed and faeces. The sediment is therefore the most affected area due to the

build up of this additional organic and metabolic waste, which settles and binds to bottom soils. Increased deposits of organic matter require large amounts of dissolved oxygen during bacterial decomposition, thereby depleting oxygen concentrations in the sediment. Under conditions of low oxygen concentrations (anoxic), nutrients are released from the bottom sediments (Brunson *et al*., 1994; Mortimer, 1941). Some of these released compounds, such as ammonia and nitrite can be harmful to the health of the culture species and other aquatic organisms. A sudden collapse of stratification and subsequent mixing of anoxic hypolimnetic water with epilimnetic water and or the rise of toxic substances from the bottom can result in major fish kills.

3.1.3 Eutrophication

The word “eutrophic” is used for an aquatic system rich in biomass and nutrients, and “eutrophication” is the process through which water becomes enriched (Schindler, 1971). Eutrophication is a natural phenomenon that occurs during the ageing process of enclosed aquatic ecosystems (Schindler, 1971). The initial oligotrophic stages are characterised by low productivity and low species abundance. As nutrient enrichment takes place the water body will go through mesotrophic conditions (moderate productivity and high species diversity) to eutrophic conditions that are accompanied by high productivity, high species abundance and low species richness (Carlson, 1977; Smith *et al*., 1999; Brönmark & Hansson, 2005). Eutrophic and hyper-eutrophic systems ultimately result in algal or cyanobacterial blooms, oxygen depletion, fluctuations in pH levels, fish kills, and a decrease in aquatic biodiversity (Talling, 1976; Boyd *et al*., 1978; Smith *et al.*, 1999; Willen, 2000). Cultural eutrophication is an unnatural process caused by increased nutrient loading through human activities in the surrounding catchments (Toerien *et al*., 1975; Davies & Day, 1998). Antropogenic factors that contribute to eutrophication are for example: aquaculture, agricultural and urban runoff, atmospheric deposition (acid rain) as well as industrial and waste water leakage (Brönmark & Hansson, 2005). Farm dam systems are especially sensitive through surrounding agricultural land uses such as fertilisers (Boaventura *et al*., 1997; Schulz *et al*., 2001; Brainwood *et al*., 2004). These fertilisers are rich in nutrients and enter the farm dam system via groundwater or overland flow during rainy seasons (Schulz *et al*., 2001; Brainwood *et al*., 2004).

The negative effects of eutrophication are primarily determined by the degree of nutrient enrichment. The nutrients involved in eutrophication processes are nitrogen and phosphorus (Schindler, 1971). Phosphorus tends to be the limiting nutrient in freshwater systems, whereas nitrogen acts as the limiting nutrient in marine systems (Schindler, 1971; Hargreaves, 1998; Correll, 1999; Rabalais, 2002). Both nutrients exhibit high variation in occurrence, both seasonally and interannually.

3.1.3.1 Nitrogen

Nitrogen is found as organic and inorganic, particulate and soluble forms of which the inorganic, soluble forms are the most biological available for plant and algal growth. Inorganic, soluble nitrogen exists as nitrate, nitrite and ammonium. Particulate and dissolved forms of nitrogen are converted to

ammonium by bacterial action and oxidised to form nitrites and eventually nitrates (Hargreaves, 1998; Rabalais, 2002; Brönmark & Hansson, 2005). In aquatic ecosystems, particulate nitrogen is embedded in the live algal biomass. Nitrogen can enter the system via rainfall, soil erosion, agricultural runoff including nitrogen from fertilisers and animal waste, groundwater, nitrogen fixation by cyanobacteria and point sources such as waste water facilities (Hargreaves, 1998; Rabalais, 2002). Nitrogen can be lost from the system via denitrification (reduction of nitrate to the gaseous form of nitrogen in anoxic bottom sediments) by bacteria or deposition to the sediments. Another major pathway of nitrogen removal is the uptake of dissolved inorganic nitrogen by phytoplankton. In addition to nitrogen sources and losses (sedimentation and denitrification), nitrogen is continuously recycled between different forms within the system (Hargreaves, 1998; Wetzel, 2001; Rabalais, 2002).

Formulated fish feed consists of a large fraction of protein for better growth performance during production. The proteins are digested and excreted as ammonia through the gills and as part of the faeces. Another source of ammonia is through bacterial decomposition of organic material such as dead aquatic plants, plankton and fish farm waste (fish excretion and uneaten fish feed). Ammonia occurs as total ammonia nitrogen (TAN) and comprises of two forms: the toxic (un-ionised) ammonia (NH_3) fraction and the non-toxic (ionised) ammonium (NH_4) fraction. Ammonia and ammonium exist in a fine equilibrium that is dependent on the pH and water temperature of the water body (Hargreaves, 1998; Moss, 1998; Wetzel, 2001). An increase in water temperature and pH will cause the equilibrium to shift towards the toxic un-ionised form of ammonia. Ammonia is naturally assimilated by planktonic algae and cyanobacteria that play a major role in the reduction of ammonia levels in the water. Ammonia is also removed from the aquatic system through a two step nitrification process, whereby ammonia is oxidised to nitrate. The process of ammonia oxidation is mediated by two genera of nitrification bacteria and involves a two-step oxidation process. Ammonia is first converted into nitrite and then into nitrate which is not as harmful to the culture species (Hargreaves, 1998; Brönmark & Hansson, 2005). A lower water temperature during the colder months slows down the bacterial process of converting ammonia to nitrate. Ammonia accumulation can be toxic, and sublethal effects can be identified as reductions in growth rates and immunocompetence. Another potentially toxic nitrogenous substance that can be detrimental to fish health is nitrite. Nitrite is released as an intermediate product during the process of nitrification and denitrification (DWAF 1996c; Hargreaves, 1998; Brönmark & Hansson, 2005).

3.1.3.2 Phosphorus

Phosphorus can enter a farm dam ecosystem via soil erosion, surface runoff from residential and agricultural lands, and point sources such as waste water facilities (Correll, 1999). Phosphorous can be found as soluble inorganic phosphate (orthophosphates and polyphosphates), soluble organic phosphates and particulate organic and inorganic phosphates. As phosphorus inputs reach the water, phosphorus is released and converted to soluble inorganic orthophosphate, the only form phytoplankton are able to assimilate (Wetzel, 2001; Brönmark & Hansson, 2005). Most of the phosphorus used by aquatic plants and phytoplankton is recycled. During decomposition of organic

matter, phosphates are released back into the water to be reused by algae. Particulate phosphorus may settle to the bottom where it either binds to the sediment or remains until microbial communities utilise it. Binding to aluminium and ferric hydroxides are particularly strong interactions and is considered to be biologically unavailable (inert) until appropriate conditions arise and bounded phosphorus is converted to biologically available orthophosphates (Moss, 1998; Correll, 1999). Internal loading of phosphorus through the release of phosphorus from bottom sediments, is a slow process. Low or anoxic oxygen conditions bring about a more rapid regeneration of phosphorus from bottom sediments, thereby increasing the concentration of biologically available phosphates (Wetzel, 2001, Kisand & Nõges, 2003). The resuspended phosphorus will either be released to surface waters immediately in shallow water bodies or be distributed to surface waters during overturn of the water column (Baldwin *et al*., 2003)

3.1.4 Associated water quality parameters

3.1.4.1 Dissolved oxygen

Dissolved oxygen is a major limiting factor for the functioning and survival of fish and other aquatic organisms. Fish and aquatic organisms require oxygen for respiration and for the regulation of metabolic processes. Besides supporting the respiration of aquatic organisms, oxygen availability is also important for the regulation of all oxidation, nitrification and degradation processes within the aquatic system (Hargreaves & Tucker, 2002). The main sources of dissolved oxygen in aquaculture dams are photosynthesis and diffusion at the air-water interface or any disturbance at the water surface (wind turbulence, human induced turbulence) (Erez *et al*., 1990; Hargreaves & Tucker, 2002). The pressure of oxygen in the air drives the oxygen into the water until the pressure of the oxygen in the water is equal to that of the air. Furthermore, oxygen is added to the water environment as a by-product of photosynthesis by aquatic plants and phytoplankton in the system (Wetzel, 2001; Dodds, 2002; Hargreaves & Tucker, 2002). The reduction of dissolved oxygen levels in aquatic systems is regulated by the respiration of aquatic organisms (fish, macrophytes, plankton), re-suspension of anoxic sediments and turnover of oxygen depleted hypolimnetic water, chemical breakdown of pollutants and the bacterial decomposition of organic material (Boyd *et al*., 1975; Boyd *et al*., 1978; Hargreaves & Tucker, 2002). The concentration of dissolved oxygen declines as temperature and salinity of the water increases, as high water temperatures and high salinity concentrations lower the solubility of oxygen (Dodds, 2002). The concentration of dissolved oxygen is found to vary over a 24-hour period depending on the relative rates of consumption and production by the aquatic organisms (Erez *et al*., 1990). Dissolved oxygen cyclic patterns indicate a decline in concentration during night time reaching a minimum at dawn, followed by a rise to maximum values by mid afternoon, and then decreasing again during night (Erez *et al*., 1990). In natural systems, the presence of algal blooms and the subsequent collapse of a bloom, give rise to fluctuations between dissolved oxygen and carbon dioxide levels in the water column. Sufficient dissolved oxygen at the sediment-water interface is necessary to serve as a buffer against toxic metabolites released from the sediments, for example nitrite, free ammonia and hydrogen sulphide (Mortimer, 1941; Boyd, 1995).

3.1.4.2 Water temperature

Water temperature plays an important role in creating layers of different densities in the water column during thermal stratification. During stratification, layers water with different temperatures will facilitate the uneven distribution of nutrients and dissolved gasses (N_2, O_2, and CO_2) (Reid, 1961; Wetzel, 2001; Dodds, 2002). Dissolved oxygen concentrations in hypolimnetic water will decrease since contact with epilimnetic waters is reduced and depletion by microbial decomposition of organic matter will continue. Turnover or mixing events will result in dissolved gasses and nutrients being evenly distributed throughout the water column, especially to nutrient poor surface layers (Moss, 1998; Wetzel, 2001). During these turnover events, anoxic bottom water rises to the surface, bringing with it toxic compounds about that have been released from the sediments under anoxic conditions (Mortimer, 1941; Brunson *et al.,* 1994).

3.1.4.3 pH

Water pH can be described as the measurement of hydrogen ions in water and indicates whether the water source is acidic or alkaline. The controlling variables involved are the hydrogen and hydroxide ions. The equilibrium between these variables strongly depends on chemical reactions and biological activities in the environment (Wetzel, 2001; Dodds, 2002). The most important biological activities influencing pH are the respiration of aquatic organisms and photosynthesis by phytoplankton and macrophytes (Boyd, 1995; Moss, 1998; Brönmark & Hansson, 2005). The water pH in aquaculture ponds exhibits daily fluctuations or cycling. After sunset, dissolved oxygen levels decrease as photosynthesis ceases and biological components continue to consume oxygen (respiration & decomposition) (Richards *et al*., 1965; Talling, 1976; Boyd, 1995). Carbon dioxide released during respiration reacts with water to form carbonic acid, thereby lowering the pH values. The rise in carbon dioxide concentrations at night will cause pH values to decrease and shift towards the acidic side of the equilibrium. During daylight, when photosynthesis increases and more oxygen is added to the system, the pH will increase to form a more alkaline medium (Talling, 1976; Erez *et al*., 1990). Additionally, changes in the pH equilibrium can also be caused by anthropogenic processes such as industrial effluents, acid rain, agriculture and aquaculture (Boyd, 1995). Fluctuations of pH values outside the proposed optimum range for aquaculture can have detrimental effects on fish health. The toxicity of ammonia is greatly influenced by the reigning pH conditions in aquaculture ponds. Higher concentrations of the toxic form of ammonia are formed in alkaline water, whereas the non-toxic form is more prevalent in acidic waters. An increase in pH values subsequently increases the toxicity of free ammonia (Hargreaves, 1998). During phytoplankton blooms, pH values can rise as phytoplankton consumes large quantities of carbon dioxide during photosynthesis (Talling, 1976). A drop in pH values improves the solubility of heavy metals such as copper and zinc, which can be toxic to the culture species in their soluble forms. A higher pH will produce less toxic forms or increase the insolubility of metals in the aquatic system (Mortimer, 1941). An increase in depth is associated with lower pH values as oxygen concentrations in bottom waters are low due to decomposition processes and reduced photosynthesis (Moss, 1998).

3.1.4.4 Turbidity

Turbidity is a measure of the extent to which sunlight can penetrate water and is influenced by the content of suspended solids in the water (Walmsley *et al*., 1980). Suspended solids are made of substances such as clay and sand particles, plankton, silt and humic compounds. The catchment geology, soil structure and steepness of banks of the dam will have a direct effect on the turbidity levels (Walmsley, 1978). Agricultural land use surrounding farm dams can have a significant influence on the turbidity levels (Walmsley *et al*., 1980). Inadequate agricultural techniques, removal of riparian vegetation and overgrazing can be responsible for erosion and subsequent increases in turbidity levels during rainy seasons. Aquaculture can have an effect on the suspended solid content as it adds large amounts of uneaten feeds as well as faecal solids to the water (Tlusty *et al*., 2000). The turbidity of the resource water can affect fish farming by clogging the gills of production fish and can reduce visibility for feeding (Hart, 1986). Partial shading by suspended solids can also favour the development of cyanobacterial blooms (Harding & Paxton, 2001).

Although the principles of aquatic ecology and water quality are well-known, there is much yet to understand regarding the limnology of farm dams before these aquatic systems can be managed effectively for fish production without detrimental effects to the environment. The aim of this chapter is to gain knowledge on basic limnology of selected farm dams and to assess impacts of rainbow trout (*Oncorhynchus mykiss*) production on the dam ecosystem. Changes in water quality parameters, both physical and chemical, of two farm dams with fish production and two reference dams with no fish farming were investigated.

3.2 Materials and methods

The study sites included four farm dams, all located in the Stellenbosch region of the Western Cape. Two of the sites were subjected to fish farming of rainbow trout (*Oncorhynchus mykiss*) from May to October. The remaining two sites contained no fish farming and had no history of aquacultural activities. See Chapter 2 for more information regarding study sites.

Water samples for chemical and nutrient analysis were collected biweekly from June 2005 to November 2006, from a deep-water station at each study site. Other field measurements included vertical profiles (0 m, 3 m, 6 m, and > 6 m depending on depth of the dam) of water temperature (°C) and concentration of dissolved oxygen (mg/L). A standard secchi disk, with a diameter of 250 mm, was used to estimate light penetration (Wetzel & Likens, 2000). For further methodological details, see Chapter 2.

Unfiltered samples were analysed for conductivity (µS/cm), total suspended solids (mg/L), total dissolved solids (mg/L), pH and total phosphorous (mg/L P). Total phosphorous was determined by acid digestion with potassium persulphate and sulphuric acid and colorimetrically measured by analysis of solubilised phosphorous. Filtered samples were analysed for ammonia-nitrogen (mg/L NH_3-N), nitrate-nitrogen (mg/L NO_3-N), nitrite-nitrogen (mg/L NO_2–N) and orthophosphate (mg/L PO_4-P). Orthophosphate concentration was determined by a colorimetric reaction with ammonium molybdate reagent and expressed as mg/L P. The trophic state of the study sites was determined following the equations of Carlson (1977) and made use of the total phosphate concentration (TP), secchi disc transparency (SD) and chlorophyll *a* content (Chl *a*). Chlorophyll *a* content was derived from biovolume using the mean ratio of chlorophyll *a*: biovolume of 7.3 µg.mm^3 (Reynolds, 1984a). Analytical methods followed the methodologies of the Hach Water Analysis System and are summarised in Tables 2.3 and 2.4 (Chapter 2). Statistical computations were performed using Statistica 7 computer software.

3.3 Results

The Western Cape is subjected to a Mediterranean type of climate with rain falling during the cold winter months and the summer months being hot and dry (Chapter 2). Surface water temperatures in the selected study sites ranged from 25-28 °C in the summer months (December, January, February) and 12-15 °C during winter months (June, July, August) (Figure 3.1). The highest water temperatures (28 °C) were recorded during February 2006 and the lowest values (12 °C) during July 2005 (Figure 3.1). There was no significant difference in the mean monthly water temperatures of the different depths between the study sites (p = 0.91).

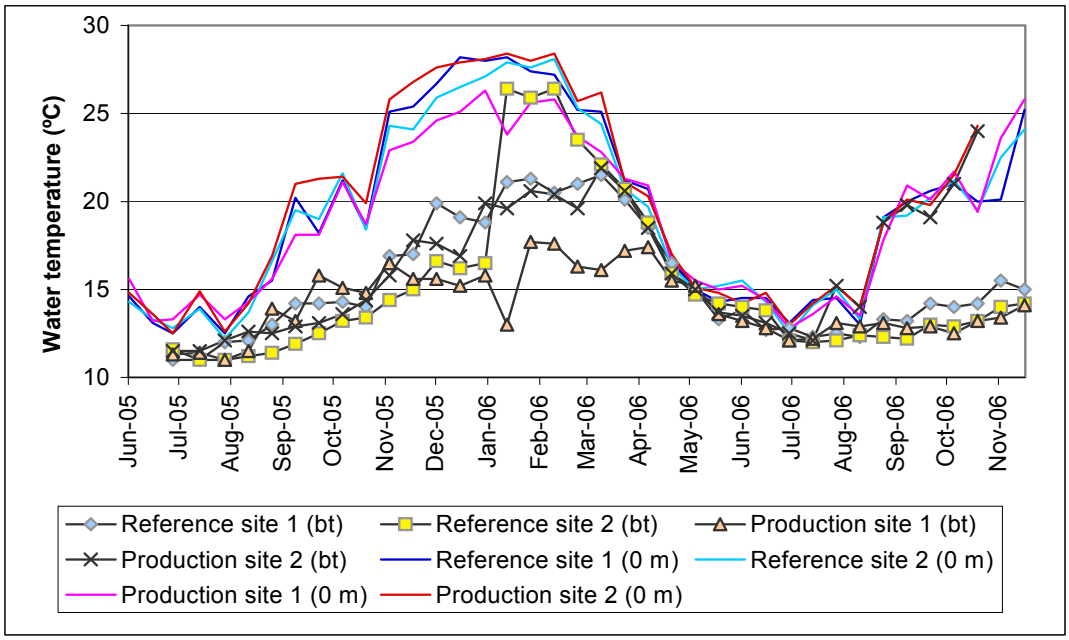


Figure 3.1: **Water temperatures of surface (0 m) and bottom (bt) water sampled in the study sites between June 2005 and November 2006**

As the temperature difference between upper and bottom layers becomes greater in the summer months, the density difference also becomes stronger and gives rise to a summer stratification period. Both, production and reference sites indicated significant differences between epilimnion and hypolimnion water temperatures during the summer (p = 0.047) and differences amounted to 11 °C (Figure 3.2). Annual thermal stratification in all the dams started to develop towards the end of September/early October and remained until breakdown in May. Production site 2 did not stratify during September 2006 as water was extracted and water levels dropped below 3 meters (Figure 3.1). A clear thermal stratification and overturn phase could also not be distinguished in reference site 2 between February and April 2006, as the water level dropped below 4 meters (Figure 3.1). The drop in water level (< 3 m) was mainly due to water extraction for irrigation use and evaporation during extremely hot summer conditions.

As winter approached, the surface water cooled down and the temperature difference between epilimnion and hypolimnion decreased. Figure 3.2 clearly shows the decrease in temperature difference between the epilimnion and hypolimnion as winter approached. Vertical thermal profiles indicated that all the dams underwent a holomictic turnover event in May, when the whole water column mixed. Following this event, the dams exhibited a monomictic mixing pattern, with continuous circulation throughout the winter. Figure 3.2 indicates a decrease in the difference between epilimnion and hypolimnion temperatures between the months May and September.

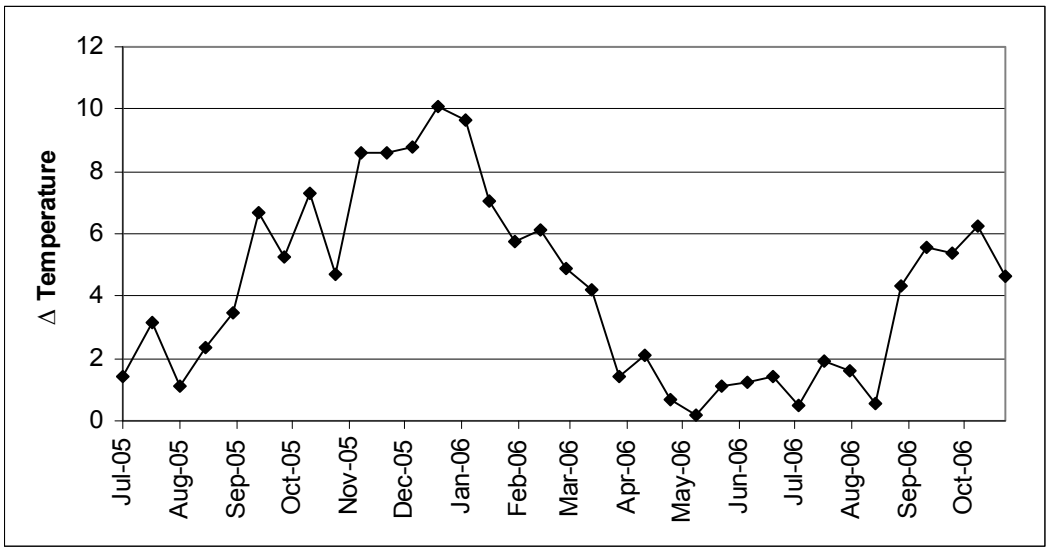
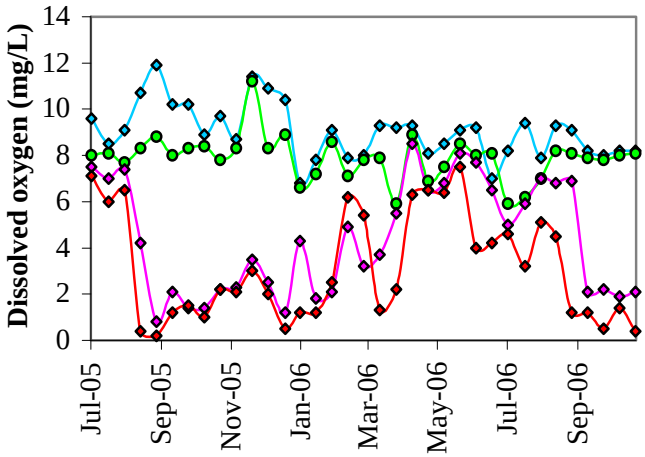
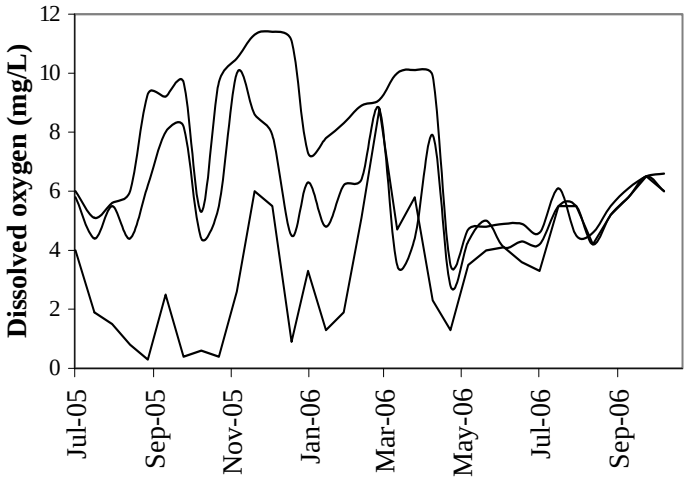


Figure 3.2: **Average difference between surface and bottom water temperatures in production and reference sites between June 2005 and November 2006**

Figure 3.3 illustrates the monthly distribution of dissolved oxygen (DO) measured over the course of the study period, providing a picture of seasonal trends. Dissolved oxygen in the epilimnion ranged from 7 mg/L to 10 mg/L, whereas dissolved oxygen concentrations in the hypolimnion ranged from 0.5 mg/L to over 12.7 mg/L.

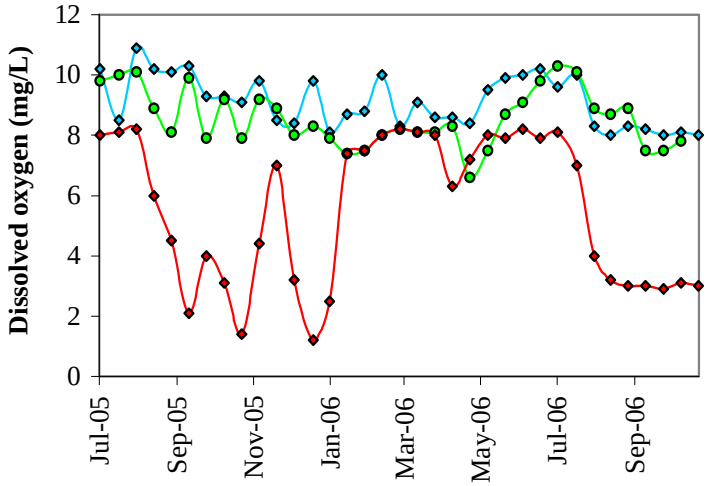


(a) **Production site 1**

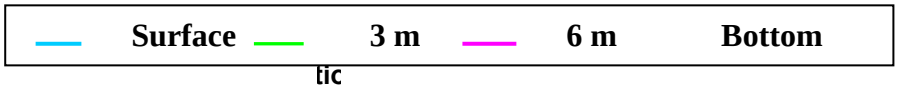


(c) **Production site 2**

(b) Reference site 1



(d) Reference site 2



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until November 2006

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A correlation analysis of the change in water temperature between the hypolimnion and epilimnion indicated a very significant positive correlation with a difference in dissolved oxygen concentrations between these layers (Table 3.1).

Table 3.1: Spearman rank correlation statistics between temperature gradients and dissolved oxygen concentrations within the hypolimnion and epilimnion

Dam	R_s	*p* - value
Reference site 1	0.52	<0.001
Production site 1	0.40	<0.01
Reference site 2	0.67	<0.001
Production site 2	0.72	< 0.001

Seasonal variation in dissolved oxygen concentrations is largely influenced by the water temperature and can be divided into two distinct phases, a summer stagnation phase (December – February) and a winter overturn phase (May – September). The summer phase is associated with anoxic conditions (anoxic conditions defined when DO concentrations are less than 2 mg/L) in the hypolimnion (Figure 3.3). In winter all the dams were subjected to a continuous mixing regime that caused the well-oxygenated upper layers to come into contact with oxygen depleted deeper layers. During this period, the water column became more oxygenated, with DO levels fluctuating between 6 mg/L and 8 mg/L. Elevated DO concentrations during winter also coincided with high phytoplankton biomass in dams (Chapter 4). Despite the high phytoplankton biomass, low DO levels (4 mg/L and 6 mg/L) were observed in production site 2. This may suggest that oxygen levels were more affected from the anoxic sediment when compared to the other sites . The reigning water level of the dams was an additional contributing factor to the distribution of dissolved oxygen. In the two deeper dams, production dam 1 and reference dam 1, a similar pattern was observed. Both dams' DO stratification developed in September and collapsed in May, coinciding with thermal stratification. The two shallower dams, reference site 2 and production site 2, also stratified as water temperatures increased during spring. However, in February 2006 the water level of reference site 2 dropped below 3 m, resulting in oxygenation throughout the water column. From August 2006 onwards the water level of production site 2 also remained below 4 m and dissolved oxygen levels remained consistent between 4 mg/L and 5 mg/L.

Total suspended solids followed a similar pattern in all the dams with greatest values in winter and lowest values measured during the summer months. Suspended solids fluctuated between 4 mg/L and 24 mg/L in winter, while summer concentrations ranged between 2 mg/L and 10 mg/L (Figure 3.4). Suspended solids started to increase in May which coincided with the annual holomictic mixing and could partly be the consequence of particulate matter being resuspended from the sediment. For the rest of the winter turbidity remained relatively high until spring. Statistical analysis showed a significant difference in suspended solids between production sites and reference sites (p = 0.031).

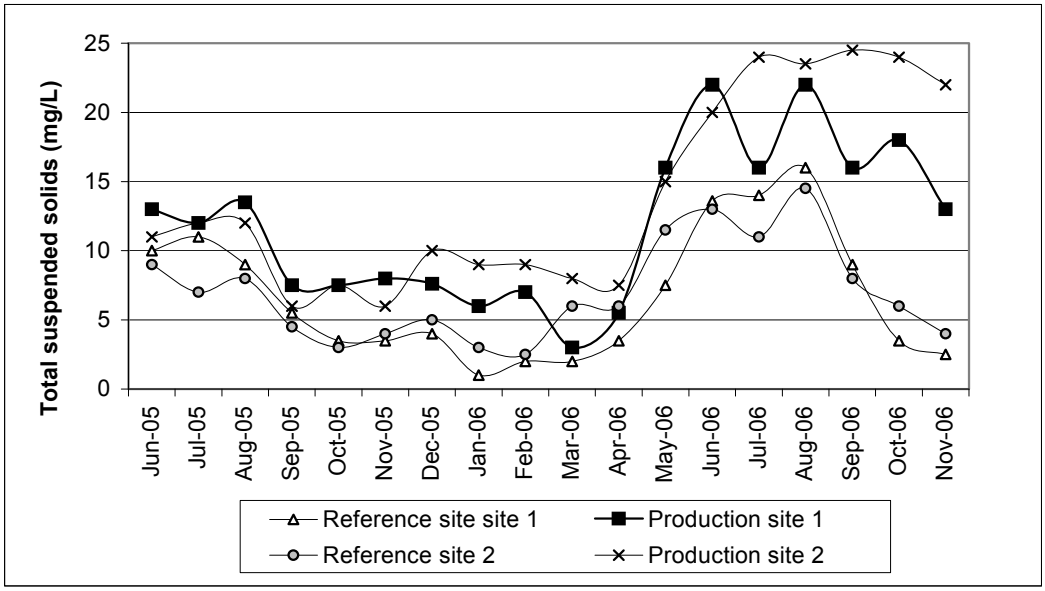
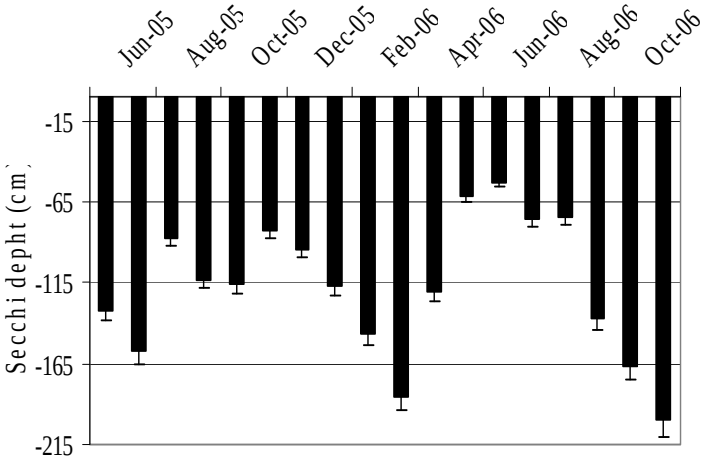
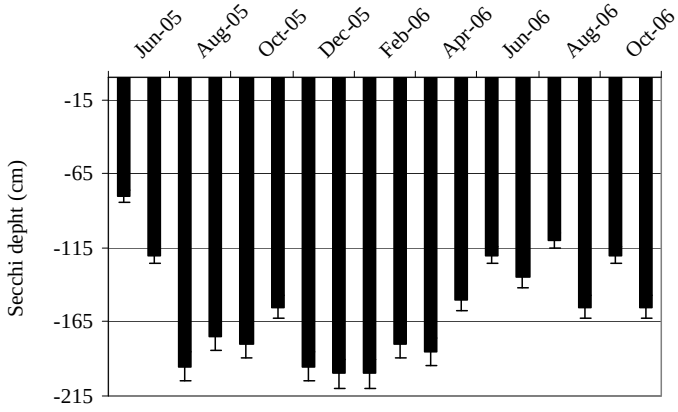


Figure 3.4: Annual variation of total suspended solids (TSS) at production sites and reference sites between June 2005 and November 2006

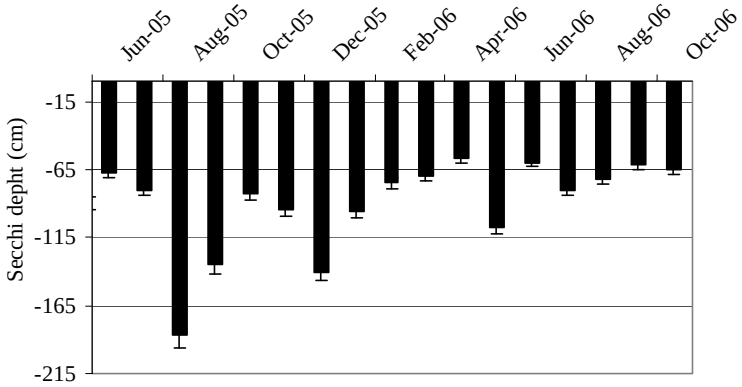
Figure 3.5 presents the variations in turbidity in terms of secchi depth transparency and results fluctuated between 0.5 m and 2.0 m. Production site 2 experienced a higher turbidity than the other dams with an average of 0.9 m (*n* = 35). The average secchi disc measurement of production site 1 was 1.18 m (*n* = 35). Both reference sites had higher secchi depth readings, indicating lower turbidity levels. The secchi disc readings of reference site 1 and reference site 2 averaged 1.5 m (*n* = 35) and 1.3 m (*n* = 35) respectively. The results show a seasonal pattern in the turbidity of both reference sites (Figure 3.5).



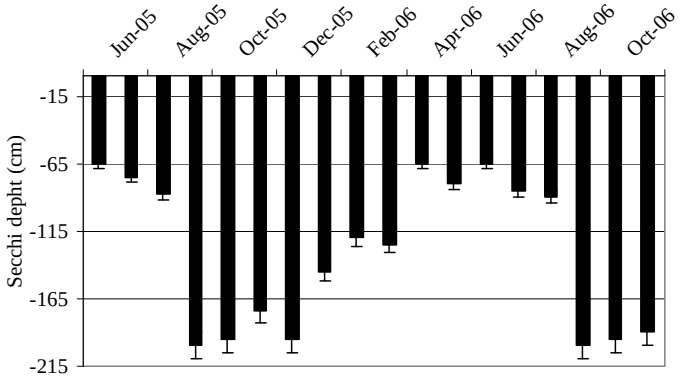
(a) Production site 1



(b) Reference site 1



(c) Production site 2



(d) Reference site 2

June 2005 until November 2006

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At these sites turbidity levels increased during April and remained high until July. From August onwards, turbidity decreased and secchi disc readings up to a depth of 2 m were recorded. Water transparency remained high during the summer months and decreased gradually during autumn. The turbidity levels of the production sites did not show any seasonal pattern.

Total dissolved solids at production site 1 were significantly higher ($p < 0.001$) than the other dams and fluctuated between 113 mg/L and 263 mg/L. Trend analysis of production site 1 indicated an increase in the total dissolved solid (TDS) values towards the end of 2006 (Figure 3.6). This dam received effluent water from a wine cellar nearby, which may have influenced the TDS concentration. TDS levels in all other dams were much lower, ranging from 30 to 85 mg/L.

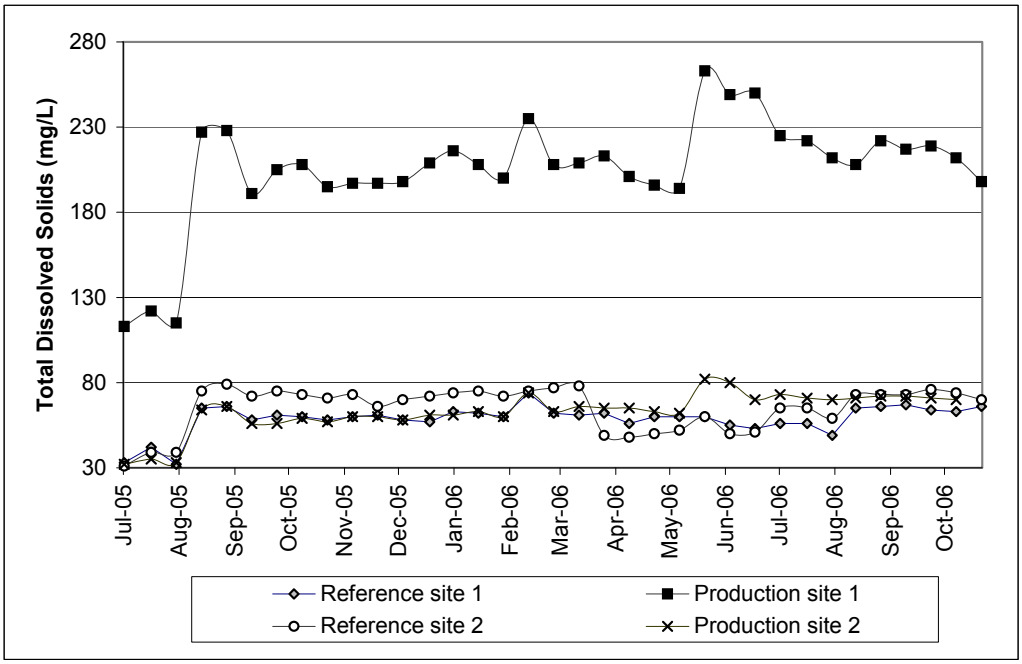


Figure 3.6: Annual variation of total dissolved solids (TDS) at production sites and reference sites between June 2005 and November 2006

The TDS data also showed differences in seasonal distribution. A decrease in TDS values was observed during and immediately after the winter months in 2005. Rainfall during the winter months could have been responsible for the dilution effect of TDS concentrations. During 2005, TDS levels in all the dams were higher during spring and summer when compared to the winter months. The same pattern was observed in 2006 for the two reference sites. Production sites however showed higher TDS values during winter sampling events.

The pH values of surface samples measured at all the study sites fluctuated between 6.5 and 8.9 standard units (Figure 3.7). In the reference sites the pH values appeared to be more stable varying between 7.1 and 8.7. The pH values of surface water in the production dams exhibited slightly more extreme fluctuations that at times reached values as low as 6.5 and as high as 8.9.

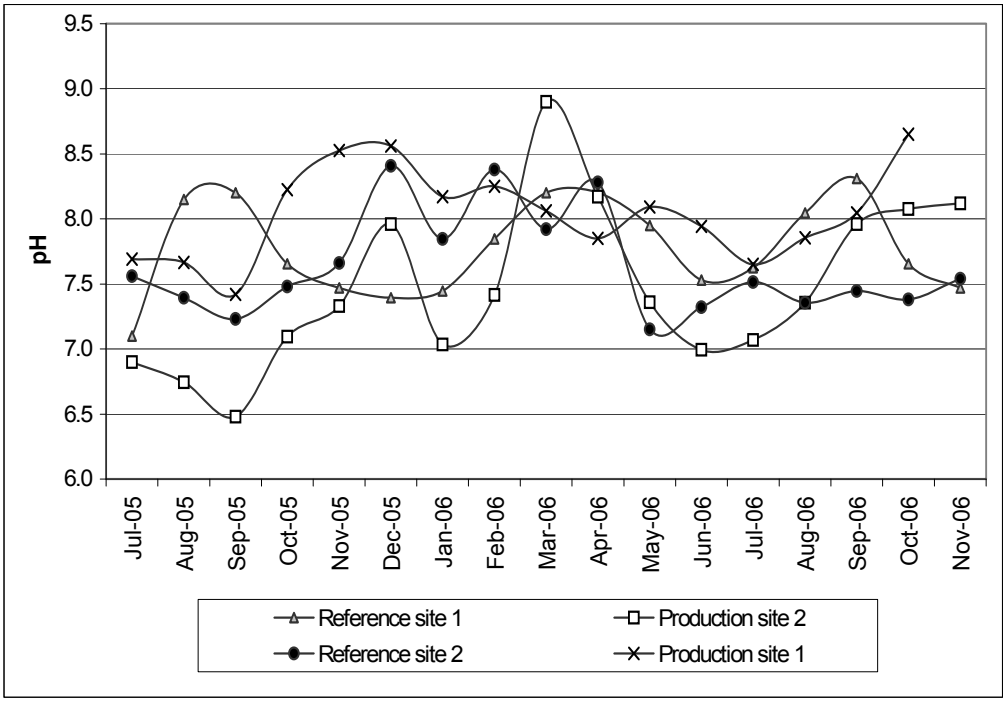


Figure 3.7: **Seasonal variation in surface pH values of production sites and reference sites between June 2005 and November 2006**

Linear regression indicated that surface pH value of both production sites and increased from the beginning of the study (R^2 = 0.27; R^2 = 0.051). The pH values of reference site 1 showed a slight increase during the study period (R^2 = 0.023). Trend analysis of pH development in reference site 2 illustrated a slight decrease towards the end of the study (R^2 = 0.024).

Figure 3.8 indicates that the pH values of surface waters were significantly higher ($p < 0.05$) than those measured in bottom samples, particularly during the summer stratification period, in the production sites. In winter, the difference in surface and bottom pH was less extreme (Figure 3.8).

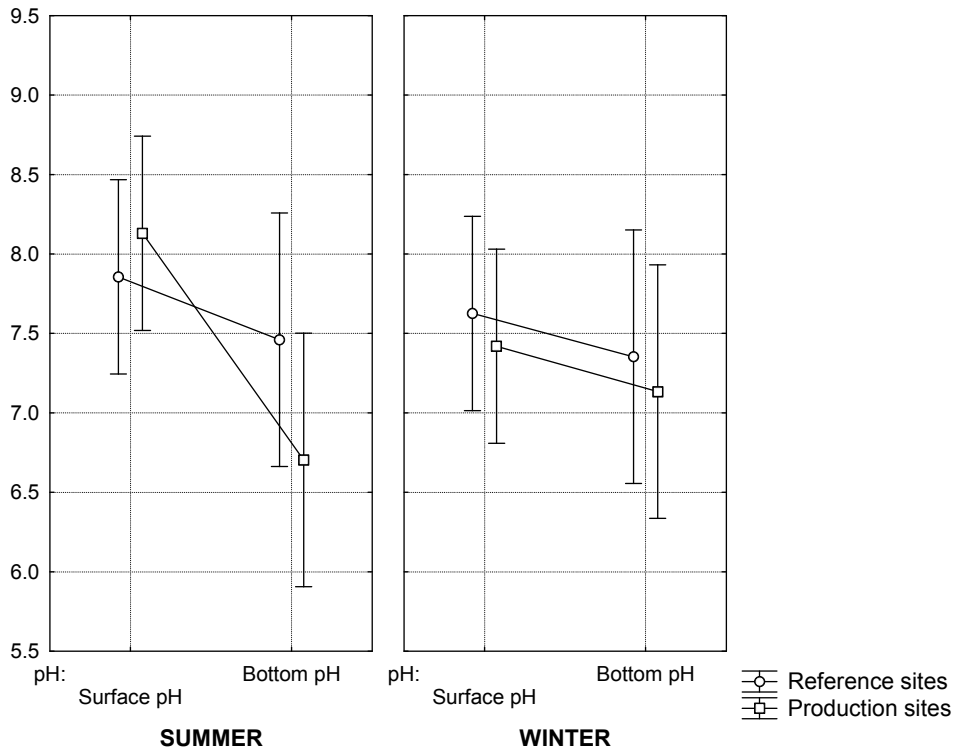


Figure 3.8: **Comparison of pH in surface and bottom water samples measured in production and reference sites between June 2005 and August 2006. Significant differences ($p = 0.05$) were observed during the summer stratification phase**

Figure 3.9 shows the seasonal variation in nitrate levels, which varied between 0.01 mg/L and 0.3 mg/L. All the dams followed a similar pattern in nitrogen seasonality, with the highest concentrations in winter and lower concentrations in summer.

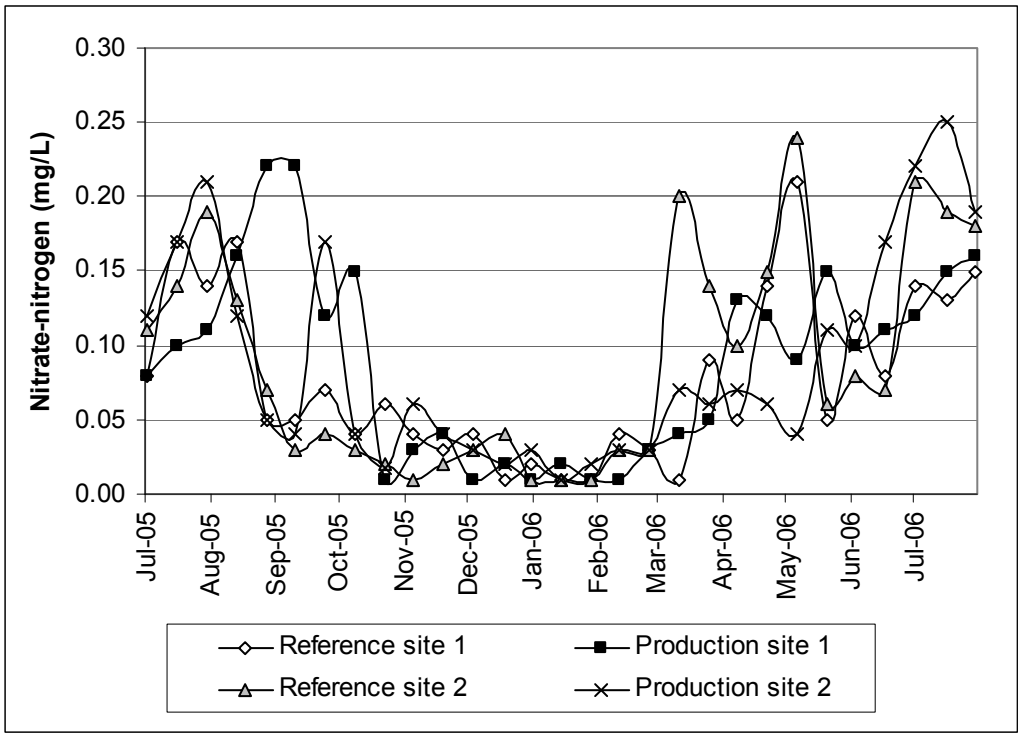


Figure 3.9: Seasonal fluctuation of nitrate-nitrogen (NO_3-N) in surface water of production and reference sites between June 2005 and August 2006

No significant trends ($p > 0.05$) in the spatial distribution of nitrate could be detected. However, surface concentrations were more comparable to deeper layers in winter. Winter months were characterised by continuous mixing of the water column, suggesting that enriched bottom water could have lead to the higher nitrate concentrations in surface water.

On a temporal scale, there was significant interaction ($p = 0.0009$) of nitrate concentrations between production sites and reference sites and between the winter and summer intervals (Figure 3.10). Nitrate values in production site 2 were significantly higher than in all the other sites, both during the production season (the winter months) and recovery season (the summer months). During the production season (the winter months) only reference site 1 did not experience significant increases in nitrate concentrations. During the recovery season (the summer months), nitrate values were significantly lower than during the winter production season (May through October), in all sites except reference site 1.

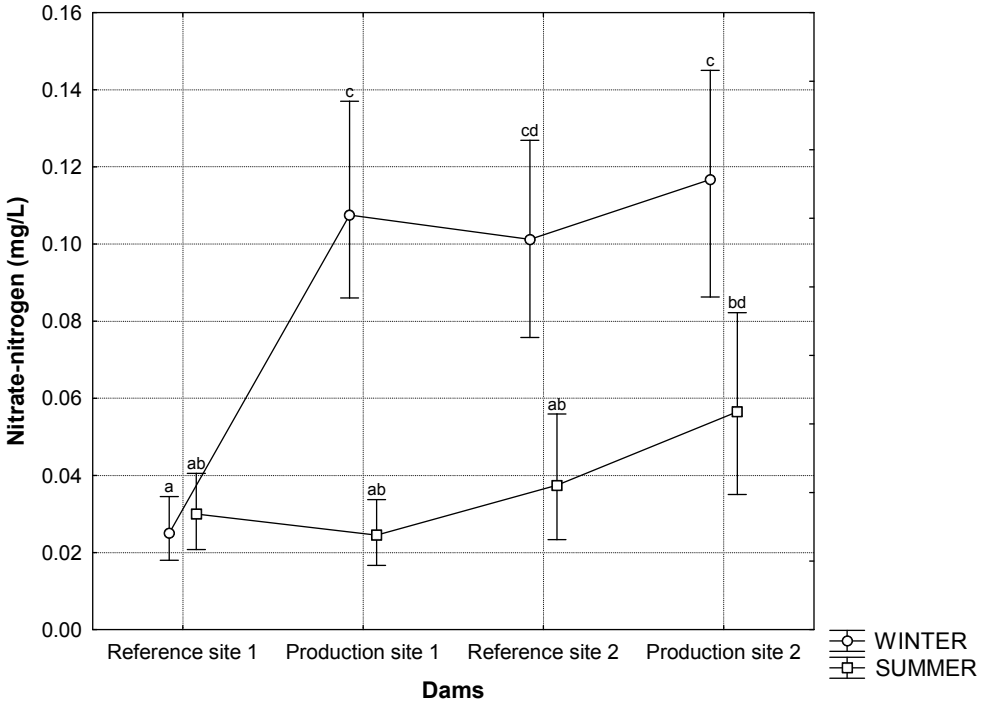


Figure 3. 10: Bootstrap means of nitrate-nitrogen measured in production sites and references sites. Significant differences were observed between winter and summer nitrate-nitrogen levels in all dams, except reference site 1. Compared to all other sites, production site 2 showed significantly higher (p = 0.0009) nitrate-nitrogen levels in both winter and summer periods

Nitrite levels were fairly low in all sites and fluctuated between 0.001 mg/L and 0.033 mg/L (Figure 3.11). No significant differences ($p > 0.05$) were detected between production sites and reference sites or at different depths. No seasonal trend was observed in nitrite distribution. However, production site 1 showed higher values in the winter months.

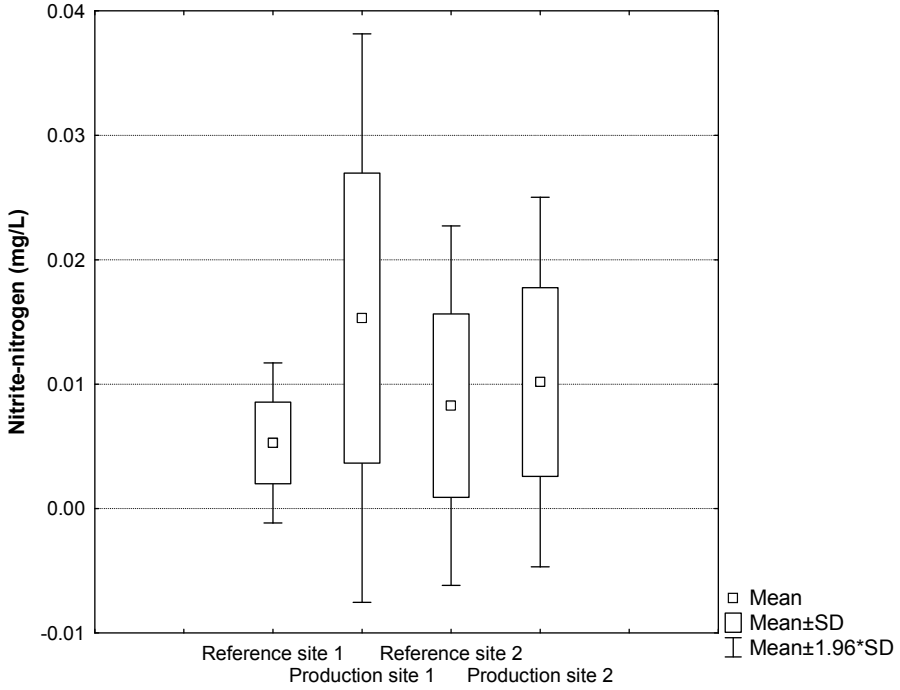


Figure 3.11: **Boxplot of average nitrite-nitrogen (in mg/L) in production and reference sites between June 2005 and August 2006**

Ammonia-nitrogen levels ranged from 0.02 mg/L to 0.65 mg/L. The spatial distribution of ammonia was not significantly different (p > 0.05) at different water levels, however, deeper waters did show slightly higher ammonia-nitrogen values in summer (Figure 3.12).

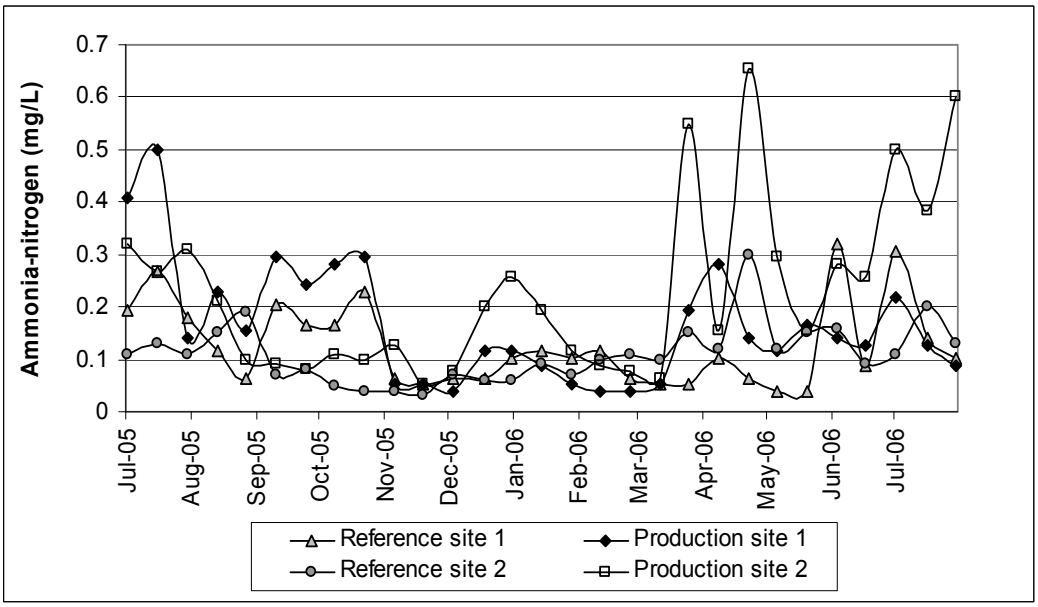


Figure 3.12: **Seasonal variation in ammonia-nitrogen (NH_3-N) in surface water of production and reference sites between June 2005 and August 2006**

Figure 3.13 indicates the significant interactions of ammonia-nitrogen between production sites and reference sites as well as differences in terms of fish farming and recovery intervals with fish farming. It is clear that ammonia-nitrogen concentrations were significantly higher in both production sites during fish farming activities compared to concentrations in the recovery period. In both reference sites no significant difference ($p > 0.05$) was found in ammonia-nitrogen concentrations between periods of fish farming in winter and the summer recovery periods. Ammonia-nitrogen concentrations were significantly higher at production site 2 compared to all the other sites during the winter production season.

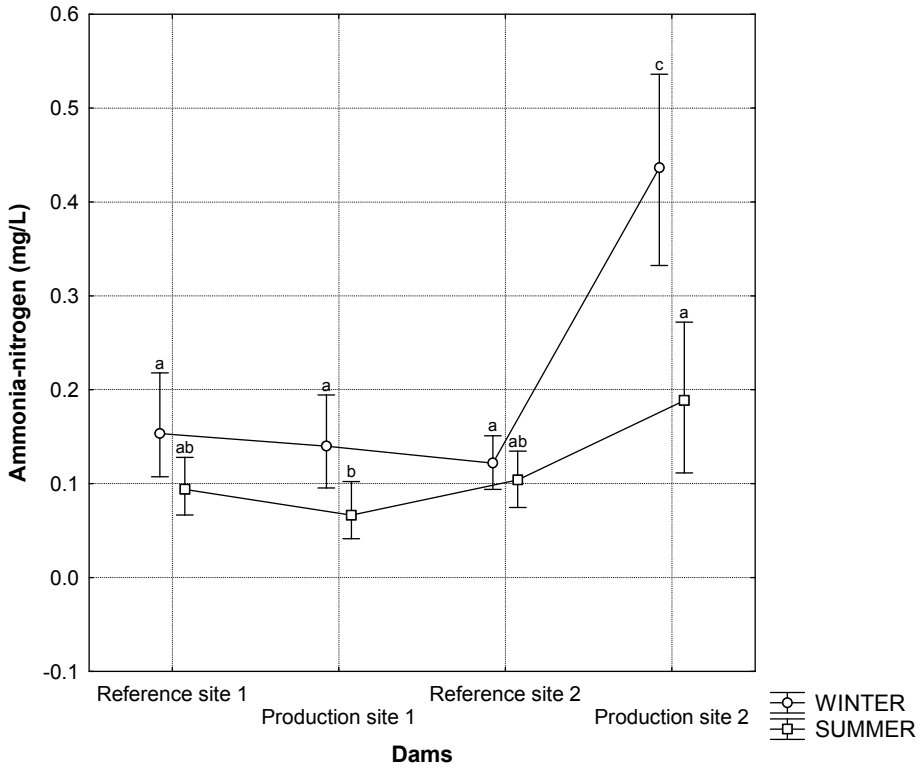


Figure 3.13: **Bootstrap means of ammonia-nitrogen measured in production sites and references sites. Significant differences ($p = 0.002$) were observed between winter and summer ammonia-nitrogen levels in both production sites. Reference sites showed no significant difference ($p > 0.05$) between ammonia-nitrate levels in the summer and winter periods**

Orthophosphate (as mg/L P) levels at the selected sites fluctuated between 0.01 mg/L and 0.5 mg/L. No marked seasonality of orthophosphate distribution was found, however, after grouping the data orthophosphate concentrations were higher in the winter than in the summer (Figure 3.14). There was no significant difference ($p > 0.05$) between orthophosphate concentrations of reference and production sites.

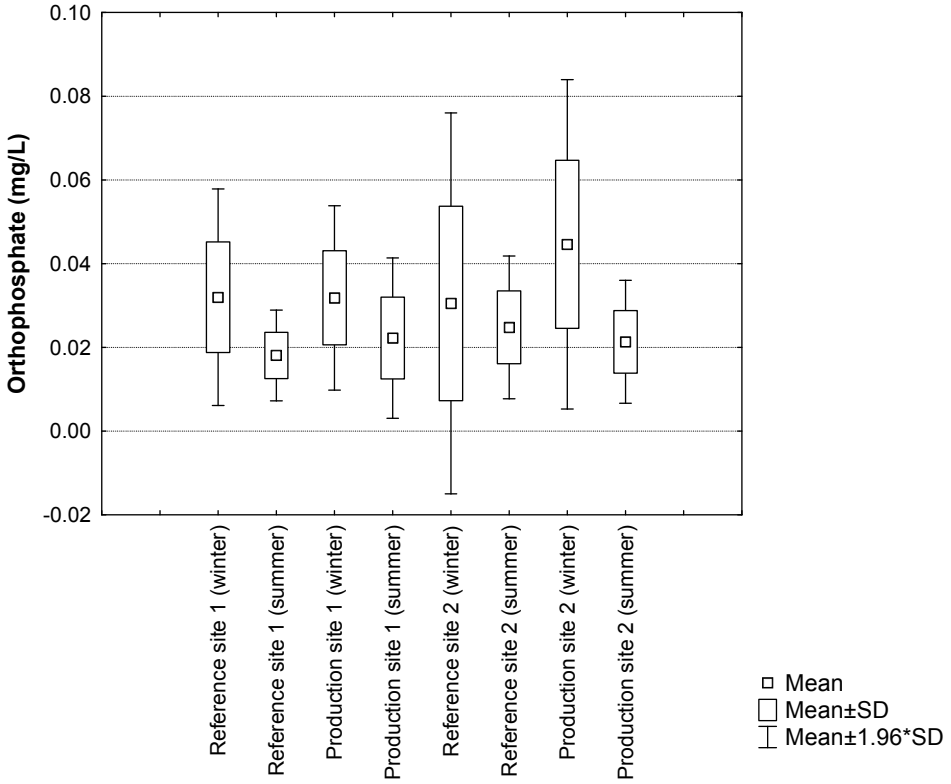


Figure 3.14: **Boxplots of orthophosphate (in mg/L) in production and reference sites between June 2005 and August 2006**

Additionally, there were no significant differences ($p > 0.05$) in orthophosphate concentrations in water samples from different depths. Differences were more visible in the distribution of phosphorus within each system. Figure 3.15 indicates that in summer in both production and reference sites, orthophosphate values were higher in bottom water when compared to surface measurements. In winter, however, orthophosphate values were more evenly distributed and concentrations in deeper waters were more comparable to values measured in surface samples (Figure 3.15).

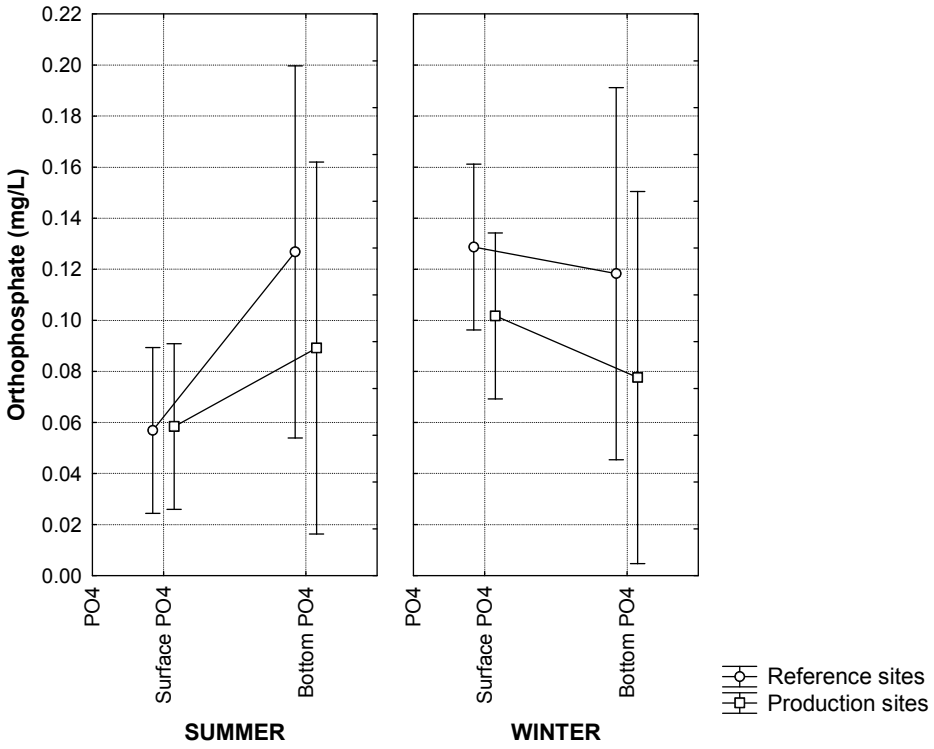


Figure 3.15: **Comparison of measured orthophosphate in surface and bottom water samples of production and reference sites between June 2005 and August 2006. No significant differences were observed ($p > 0.05$) between summer and winter grouping**

Figure 3.16 indicates the trophic status of each site determined from the average values of three parameters, namely total phophorus (mg/m^3), chlorophyll *a* (mg/m^3) and secchi depth (m). The trophic state values of both production sites was higher than those calculated for the reference sites. Following the trophic status index of Carlson (1977), the trophic status index (TSI) for the production sites ranged between 79 and 97 and for the reference sites between 35 and 92. Calculations indicated that all the sites had a eutrophic character, except for reference site 1 that showed oligotrophic conditions during March 2006.

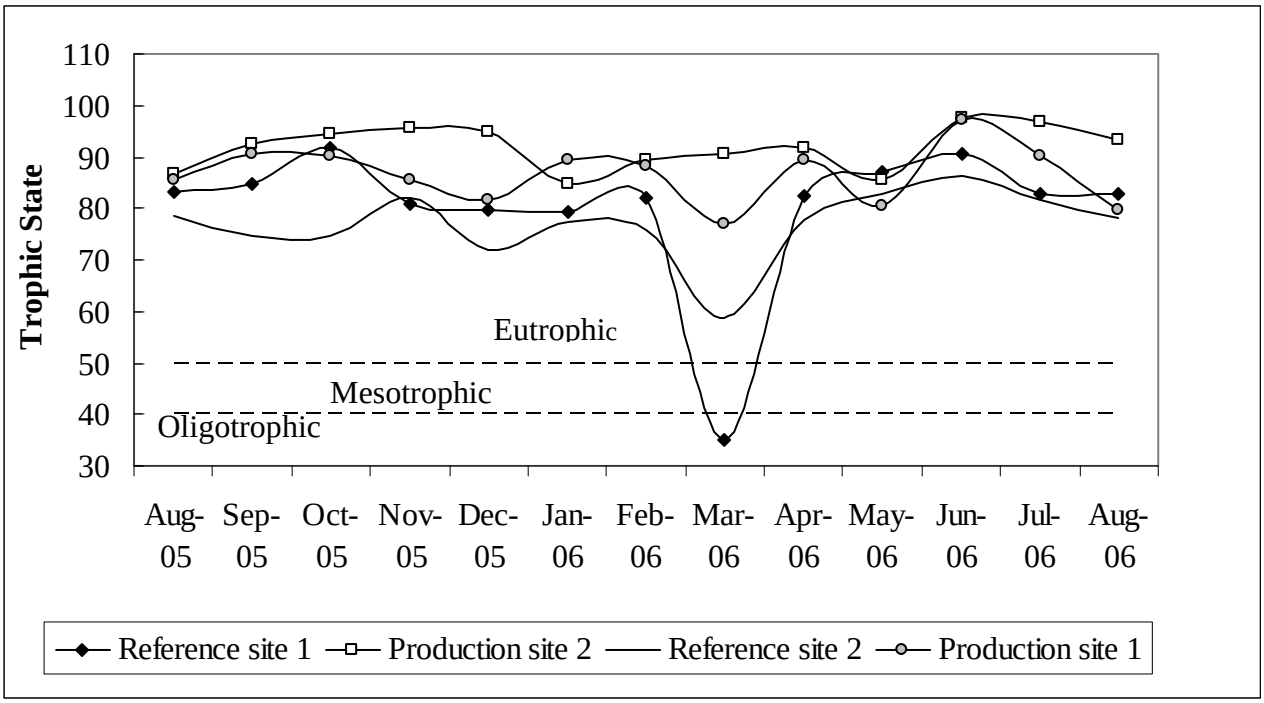


Figure 3.16: Trophic state index of production and control sites determined by building the average from the 3 parameters (total phosphorus (mg/m^3), chlorophyll *a* (mg/m^3) and secchi depth (m) for each sampling date of the study period

3.4 Discussion

This discussion focused on water quality trends and eutrophic conditions observed over the course of the study. Seasonal periods are defined in terms of fish farming operations (months during which production takes place) and recovery seasons (months with no production present), and were divided as autumn through winter (May - October) and late spring through summer (November – April). Seasonal effects and variations in the chemical and physical parameters, nutrient loading and overall patterns in stratification are discussed.

The thermal characteristics of inland farm dams depends largely on various structural characteristics (volume, depth, surface area), prevailing climatic conditions (cloud cover, rainfall, air temperature, winds) and hydrological (source of inflow) factors (Smith *et al*., 1999; Brainwood *et al*., 2004). The reigning climatic conditions of the Western Cape are Mediterranean, which has a major affect on the water regime of farm dams (Imberger, 1985; Davies & Day, 1998). The summer months are associated with gradual heating of the surface water by higher air temperatures and longer periods of light penetration. Differences between hypolimnion and epilimnion water temperatures may reach 11 °C and are supported by findings of limnological studies on larger reservoirs in Southern Africa (Robarts *et al*., 1982; Imberger, 1985). The stable stratification period that dominated the water column from September to April are similar to patterns found in the larger South African reservoirs (Pieterse & Röhrbeck, 1990; Hart, 1992; Hart, 2001; Hart, 2006). Reference to Figure 3.1 suggests that the decrease in air temperatures during May led to the reduction in surface water temperatures of all the dams. The loss of heat from surface water was enough to destabilise the stratified water column to a point where an overturn of the whole water column was caused by increased wind action on the surface of the dam. The period of continuous overturn remained throughout the winter as increased wind action and associated rainstorms dominate the winter months and has been widely observed and documented for South African reservoirs (Robarts *et al*., 1982; Pieterse & Röhrbeck, 1990; Hart, 2001). Additionally, the temperature gradient in a water body is also responsive to changes in water level and two of the study sites were affected by a sudden reduction in water levels. During the summer months, large volumes of water were extracted for irrigation purposes, resulting in a water level drop to below 3 meter. Under such conditions, the lack of a thermocline and weak stability of the water column will cause a dam to act temporarily as a polymictic system, which circulates continuously (Milstein *et al*., 1995; Milstein & Zoran, 2001; Wetzel, 2001). In reponse to the reduction in water levels, the two affected dams did indeed show a similar polymictic pattern as found by Harding (1997) in a shallow Western Cape lake.

Dissolved oxygen levels were strongly affected by the stratification and mixing events in the dams. The late summer months were characterised by extremely oxygen depleted bottom waters, while production by phytoplankton (Chapter 4) and atmospheric deposition and turbulence (e.g. wind action) caused surface waters to be more oxygenated (Figure 3.3). Oxygen depletion and high ammonia concentrations in hypolimnion waters can be indicative of high loads of organic matter originating from

phytoplankton die-offs, decomposing plant material or fish farming waste (Boyd *et al*., 1975; Boyd *et al*., 1978). Increased water temperatures during summer could have accelerated microbial activity, which elevated the requirements of oxygen for decomposition, thereby aggravating oxygen depletion (Boyd, 1995; Troell & Berg, 1997). In May, however, oxygen stratification was interrupted by a holomictic overturn event that brought deoxygenated hypolimnetic water in contact with oxygen rich surface layers. This continued for the remaining winter months as the dams followed a monomictic pattern for the rest of the winter. Winter months were also associated with a high algal biomass that could have contributed to changing dissolved oxygen concentrations (Chapter 4). During the second year of study one of the study sites (production site 2) showed somewhat lower DO concentrations in winter compared to the other dams. This may reflect a source of oxygen depletion, possibly due to a high organic load and fish respiration, which was so severe that it exerted a lowering effect on the dissolved oxygen content of the whole water body (Stirling & Dey, 1990)

The pH values in the production sites were more unstable and exhibited greater fluctuations when compared to the reference sites (Figure 3.7). The fluctuations could be the consequence of biological processes, as both production sites sustained high concentrations of phytoplankton biomass (Chapter 4). Extreme rates of photosynthesis can increase the rate at which oxygen is extracted from the water, thereby forcing the CO_2-bicarbonate-carbonate equilibrium towards the formation of carbonate (Talling, 1976; Moss, 1998). This will give rise to alkaline pH values particularly in surface water as photosynthetic communities are more restricted to upper well-lit zones (Wetzel, 2001). A sudden collapse of high algal biomass would add additional organic matter to the sediment as it settles to the bottom. More oxygen would be consumed for decomposition, thereby lowering the pH values of near bottom water (Richards *et al*., 1965; Boyd, 1995). During the production season (May - October) fish farming allowed the input and settling of additional organic matter (uneaten fish feed, metabolic waste) on the sediment. The lower pH values in the hypolimnion could be attributed to the accelerated oxygen depletion (0.5 mg/L < $[0_2]$ < 5 mg/L) of higher organic loads produced by aquaculture activities as well as elevated summer temperatures that stimulated higher rates of bacterial decomposition (Wikner & Hagstrom, 1991).

The total dissolved solids (TDS) content represents the total amount of soluble material in a water sample and may include all organic and inorganic, ionised and unionised material. The content of TDS in natural waters is influenced by anthropogenic, geological (weathering) and atmospheric processes (precipitation, evaporation). The lowest TDS values recorded in South African freshwaters ranged from 10 - 27 mg/L and 17 - 37 mg/L in the Waterkloof stream (Gauteng) and Swartboskloof stream (Stellenbosch) respectively (Dallas & Day, 2004). Some of the highest TDS values recorded reached 65 000 mg/L (Burgerspan, South Western Cape) (Silberbauer & King, 1991). The major influence on TDS values of farm dams is the geological nature of the region. The underlying parent material consists of Table Mountain Sandstone, which is poor in leachable ions (Allanson *et al*., 1990; Day & King, 1995). Consequently, the TDS values of the studied farm dams were very low. TDS content in the farm dams was more concentrated during the summer months. The concentrating effect during

summer months could be the result of lower rainfall, irrigation activities, increased evaporation rates by extreme summer temperatures and resuspension from the sediment during anoxic conditions. Extreme summer temperatures forced farmers to make use of additional irrigation for crop production. During irrigation, crops take up water, leaving solutes behind. Solutes accumulate in the soils, and are then leached out by rainfall and return to the dam through agricultural runoff (Moolman *et al*., 1983). Another contributor to increased TDS values during the summer months is due to evaporation losses at extreme temperatures (Brainwood *et al*., 2004). Production site 1 showed significantly higher TDS values than the rest of the farm dams (Figure 3.6). This could be due to one of its inflows that receive waste water from a wine cellar in the vicinity.

The level of turbidity of inland waters is strongly affected by the amount of suspended material, organic (faeces, metabolic waste) as well as inorganic (clay, silt), in the water column (Bruton, 1985). Turbidity levels in the studied dams were greatest in winter, and lowest in summer (Figure 3.4). This was due to a high phytoplankton and zooplankton biomass (Chapter 4 and Chapter 5) and an increase in the suspended solids content of the water. Suspended solids started to increase rapidly from May and remained high for the rest of the winter months (Figure 3.4). Winter months, characterised by extended periods of rainfall, could have experienced an increase in inorganic solids entering the dams via runoff. Additionally, the monomictic mixing regime caused particulate matter to be resuspended from the sediment, adding to the suspended solid content of the water (Imberger, 1985). Organic suspended solids comprised primarily of phytoplankton biomass that reached peak densities during the winter months. The winter peaks of phytoplankton assemblages were ascribed to the continuous upwelling of nutrients during the annual mixing period (Ashton, 1985; Allanson *et al*., 1990).

Nutrient conditions indicated eutrophic conditions in all the study sites, however, one of the reference sites showed once a decrease in nutrient concentrations, namely towards a oligotrophic character (Figure 3.16). Phosphorus values of reference sites and production sites were similar, however, the main differences were in the spatial and temporal distribution thereof. In terms of temporal distribution, orthophosphate concentrations were higher in the winter months. In the summer months orthophosphate values were higher in deeper waters when compared to surface values. Oxygen depletion in deeper water could have been responsible for recycling of phosphates from the sediment, thereby raising the orthophosphate levels in deeper waters (Boyd & Musig, 1981; Kisand & Nõges, 2003). Depletion of orthophosphates by phytoplankton in surface water and a lack of mixing during summer stagnation could have resulted in the low phosphorus levels in surface water (Boyd & Musig, 1981; Hart, 1982). Winter values of orthophosphates remained high in deeper waters and showed an increase in surface concentrations (Hart, 1992). The increases in orthophosphates coincided with the period of continuous mixing suggesting that phosphorus rich waters were continuously brought up from the bottom, and enriched phosphorous depleted surface water (Hart, 1992). Increased runoff during the rainy season and fish farming activities could have contributed to higher phosphorus concentrations by adding phosphorus enriched particulate matter to the system (Heathwaite *et al*., 1996; Nhan *et al*., 2006).

Inorganic nitrogen levels appeared to be higher in winter and were almost entirely comprised of nitrate and ammonia. Vertical distribution of ammonia reflects higher concentrations in the lower portion of the water column and coincides with the summer stratification phase (Robarts *et al*., 1982). The stratification phase was accompanied by very low oxygen levels in bottom waters, indicating high rates of bacterial decomposition (Selong & Helfrich, 1998). The lack of oxygen would inhibit the conversion of ammonia to nitrate and the lack of mixing would cause ammonia to accumulate in the lower part of the water column. In winter, ammonia levels were more evenly distributed throughout the water column and this agrees with the annual mixing period. During this period ammonia is washed up from the hypolimnion where it replenishes depleted surface concentrations and is converted to nitrate (Hargreaves, 1998). The continuous resuspension of nutrients from the hypolimnion may be responsible for the replenishment of ammonia levels at the surface. This spatial and temporal distribution of ammonia observed in the farm dams is supported by a study from Robarts *et al*. (1982).

Nitrate distribution followed a similar trend at all the study sites with increases of nitrate in winter, which coincided with fish farming, in the production sites. However, significant differences in nitrate concentrations in one of the reference sites indicated that elevated nitrate levels was not limited to production dams. Studies by Stirling & Dey (1990), Selong & Helfrich (1998) and Azim *et al*. (2003) found similar results and indicated that fish farming affected nitrate levels to a lesser extent as opposed to ammonia levels. During summer stratification, nitrate was lower in surface water and could be attributed to consumption by phytoplankton and conversion of nitrate to atmospheric nitrogen in the well-oxygenated waters. Surface concentrations increased during winter - the possible result of continuous mixing of nitrogen depleted water with nutrient rich bottom water. Mixing of the water column and the absence of a thermocline would allow trapped ammonia to move upwards, to be converted to nitrate, increasing the amount of nitrate in surface water during winter. This increase in nitrate during the winter mixing period is supported by the findings of Robarts *et al*. (1982).

3.4 Conclusions

Seasonal variation in temperature and rainfall played an important role in regulating the mixing and stratification patterns in farm dams and ultimately in the distribution of nutrients. Reservoir depth, particularly the episodic extraction of water for irrigation, also proved to be an important factor in the stratification and mixing patterns of the dams.

The dams followed a holomictic, monomictic mixing regime in winter followed by a summer stratification period. The sudden extraction of large volumes of water for irrigation periodically changed two dams (reference site 2 and production site 2) into shallow, non stratifying systems, with their own patterns of mixing and nutrient distribution. Fluctuations in oxygen levels were primarily influenced by phytoplankton development and population crashes, decomposition rates, thermal stratification and mixing patterns (Chapter 4). Oxygen depleted conditions occurred in the hypolimnion during summer stratification and was indicative of high levels of enrichment in the sediment. Nutrients were more abundant in winter and could be ascribed to increased terrestrial runoff rich in inorganic particles, increased atmospheric deposition during rainstorms and the continuous mixing of the water column suspending nutrients from the sediment. Summer stagnation was characterised by lower nutrient conditions due to nutrients being trapped in the lower part of the water column and the uptake of nutrients by phytoplankton communities.

Nutrient conditions in production sites were however, fairly comparable to those measured in reference sites, indicating that aquaculture did not necessarily have direct a impact on nutrient levels over the course of the study. Other factors such as nutrient enrichment from terrestrial runoff, fertilisers used in the surrounding farmlands and incorrect management of surrounding farmlands, associated with these specific farm dams, can also contribute to eutrophication of the dams. Additionally, limnological processes that raise internal nutrient loading in the dams, occurs in winter and coincides with the fish farming season. This in fact disguised the effects of aquaculture on nutrient levels. However, no visible increase in nutrients, the extented period of oxygen depletion and significantly higher ammonia levels in production dams as opposed to reference sites, was indicative of severe organic enrichment in the bottom sediments. The addition of even more nutrients and organic waste by more than two consecutive years of aquaculture production would cause the continuous accumulation and resuspension of nutrients in the system. Additionally, the extraction of surface water during the summer stagnation phase, when nutrients are settled in the hypolimnion and the low flushing rates of these dams, would cause nutrients to remain in the system. Therefore, at present production levels, the impact of cage aquaculture on water quality was only marginally visible. However, production over a few consecutive years may damage the dam ecosystem in such a way that future production and utilisation of the resource will not be possible.

CHAPTER 4

RESPONSE OF PHYTOPLANKTON COMMUNITIES TO ENRICHMENT FROM CAGE PRODUCTION OF RAINBOW TROUT (*ONCORHYNCHUS MYKISS*) IN FOUR WESTERN CAPE FARM DAMS

4.1 Introduction

4.1.1 Importance of phytoplankton

Phytoplankton can be defined as plants lacking roots, vascular tissue and leaves. Phytoplankton forms a major component in the microbial food webs of aquatic ecosystems and aquaculture ponds. Their value lies particularly in modifying oxygen levels (Erez *et al*., 1990; Moriarty, 1997), maintenance of water quality by removing toxic metabolic end products (Moriarty, 1997), nutrient recycling and as nutrition for higher trophic levels (Moss, 1998; Wetzel, 2001). As primary producers they are accountable for the fixation of light energy into organic matter, which serves as a food source for grazing zooplankton and fish (Moss, 1998; Dodds, 2002; Brönmark & Hansson, 2005). Through the process of photosynthesis they form the net producers of dissolved oxygen (Erez *et al*., 1990; Hargreaves & Tucker, 2002). Phytoplankton are dispersed throughout the euphotic zone, either passively by turbulent diffusion and sinking, or actively by flagella and buoyancy regulation (Shapiro, 1990; Van Den Hoek *et al*., 1995; Moss, 1998). Phytoplankton cells have a range of adaptations that enable them to remain suspended in the upper water layers and to withstand grazing pressure from zooplankton and fish. The relatively small size of algal cells influences their surface-volume ratio, thereby reducing sinking rates (Reynolds, 1984a; Reynolds, 1984b; Wetzel, 2001).

4.1.2 Seasonal trends in phytoplankton abundance

Since phytoplankton assemblages undergo seasonal fluctuations, considerable time has been spent on describing the environmental forces responsible for their distribution. This has lead to the development of a model that describes the seasonal events that occur in phytoplankton and zooplankton community structure of an idealised standard impoundment (Reynolds, 1984b; Sommer *et al*., 1986). This model, however, applies particularly to water bodies in the Northern Hemisphere. Seasonal fluctuations of assemblages in the Southern Hemisphere are in agreement with this model but differ in terms of the seasonal scale (Ashton, 1985; Allanson *et al*., 1990). Northern Hemisphere sites experience biomass maximums during summer, whereas the phytoplankton assemblages in the Southern Hemisphere flourish in winter (Ashton, 1985; Allanson *et al*., 1990). This depends, however, on the type of phytoplankton. If it is cyanobacteria then they will form summer blooms. The factors regulating seasonal abundance and species composition in the Southern Hemisphere are the result of physical in-lake processes, brought on by climatic and hydrological conditions (Ashton, 1985, Imberger, 1985).

Various limnologists have studied both temporal and seasonal distribution of phytoplankton communities (Kalff & Knoechel, 1978; Goulder, 1979; Reynolds, 1984a; Figueredo & Giani, 2001; Roelke *et al*., 2004). Major factors influencing the spatial distribution of phytoplankton include physical factors (water temperature, thermal stratification, mixing of water column and turbulence, light penetration), chemical factors (nutrients, pH) and biotic factors (grazing and predation) (Hart, 1986; Robarts & Zohary, 1987; Shapiro, 1990; Philips *et al*., 1997; Litchman, 2000; Kobayashi & Church, 2003). Fluctuations in phytoplankton biomass are often described by a directional approach, namely bottom-up control and top-down control. The bottom-up hypothesis stipulates that the phytoplankton biomass is dependent on the availability of nutrients in the system (Leibold *et al*., 1997; Kobayashi & Church, 2003). If the supply of nutrients is not sufficient to sustain phytoplankton biomass, nutrient concentrations will decrease which will in turn limit the growth of phytoplankton (Tilman *et al*., 1982; Kobayashi & Church, 2003). In freshwater systems, phosphorus (P) is regarded as the limiting nutrient, whereas nitrogen (N) is the more common limiting nutrient in marine systems (Tilman *et al*., 1982; Dodds, 2002). Conditions that allow nutrient concentrations to become more readily available will in turn stimulate the growth of phytoplankton. As nutrients are essential for regulating phytoplankton biomass, N:P ratios are often used to indicate if phytoplankton growth is limited by phosphorous or nitrogen. The rationale behind using the N:P ratio as a parameter is that the atomic ratio of C:N:P in phytoplankton cells approximates 106:16:1 (42:7:1 by mass) (Redfield, 1958). When the N:P ratio significantly differs from the Redfield ratio, this indicates whether the system is P-limited or N-limited (Redfield, 1958). An atomic N:P ratio exceeding 16 indicates that the growth of phytoplankton may be limited by P, whereas an atomic ratio less than 16 indicates that phytoplankton growth may be limited by N (Redfield, 1958; Pieterse & Janse van Vuuren, 1997). Small, man-made dams are more likely to change between N-limiting and P-limiting systems due to their size and variation in nutrient loading rates, especially from surrounding farm lands. Phytoplankton biomass is also subjected to a top-down mechanism exerted by zooplankton and herbivorous fish. Top-down control stipulates that the density of phytoplankton biomass will be dependent on the abundance and grazing rate of zooplankton and herbivorous fish (Fussman, 1996; Dodds, 2002; Kobayashi & Church, 2003). Zooplankton communities that are dominated by large crustacean species such as e.g. *Daphnia* spp. are the most common consumers of phytoplankton biomass (Fussman, 1996). The relative effects of these two mechanisms therefore depends on the degree of nutrient limitation, stimulation by additional nutrient supply and the extent of zooplankton grazing.

4.1.3 Phytoplankton as indicators of resource quality

A reasonable knowledge of the phytoplankton assemblage that is sustained by the system gives us an immediate idea as to the quality and functioning of the system. Phytoplankton assemblages form the link between the physical and chemical environment and higher trophic levels, and prove therefore to be an appropriate measure of resource quality and health. This has given rise to several studies on the use of phytoplankton assemblages as a biological classification tool (Rawson, 1956; Carlson,

1977; Kalff & Knoechel, 1978; Hornström, 1981; Rosén, 1981; Willen, 2000). In South Africa extensive work has been done on the use of diatoms for the prediction of water quality conditions and is currently being implemented as part of national monitoring systems (Taylor *et al.*, 2005, Taylor *et al*., 2007a, Taylor *et al*., 2007b).

4.1.3.1 Species diversity and *biomass*

Oligotrophic systems characteristically support low species diversity and minimal biomass. Moderate enrichment will cause an increase in the plankton biomass accompanied by an increase in diversity. Representing groups include genera of Bacilliarophyta (diatoms), Chlorophyta (green algae), Cryptophyta, Dinophyta as well as Cyanophyta (blue-green bacteria). Changes in the planktonic community as a response to increased nutrient availability is marked with increasing biomass, an extended period of bloom forming, decreasing evenness and species richness, increased biomass of larger stress tolerant species and a changed structure among algal classes leading to a changed species composition (Robarts *et al*., 1992; Watson *et al*., 1997). Eutrophic systems with a high pH are able to sustain a high biomass and are often characterised by cyanobacteria dominance. Representing species composed of *Anabaena* spp., *Microcystis* spp., *Oscillatoria* spp. and also the Dinophyta species *Ceratium hirundinella* (Watson *et al*., 1997; Willen, 2000; Brönmark & Hansson, 2005).

4.1.3.2 Size structure within population

Size structures differ according to the reigning trophic status of the water body. Generally oligotrophic systems are dominated by small, edible algae whereas large inedible algal species dominate eutrophic waters (Watson *et al*., 1997; Willen, 2000). Large phytoplankton species grow slowly due to a limited capability for assimilating nutrients from nutrient poor conditions. This gives smaller species an advantage in dominating the assemblage under nutrient poor conditions. Small forms are, however, more susceptible towards grazing by herbivorous zooplankton, whereas large forms are often too large to be consumed (Sommer *et al*., 1986). During nutrient rich conditions large phytoplankton forms will replace smaller species, as they are more immune to grazing by zooplankton (Elser & Goldman, 1991; Conde-Porcuna *et al*., 2002).

4.1.4 Phytoplankton blooms

Ecological imbalances and physical disturbances can cause mass appearance of single phytoplankton species, known as “blooms”. Blooms characteristically have a low diversity with only one or two single species dominating the assemblage (Willen, 2000). Causative species that have been recorded belonged to the Chlorophyta, Dinophyta and Cyanophyta.

4.1.4.1 Driving forces behind bloom formation

Numerous studies have concluded that one of the main driving forces behind bloom formation is the nutrient condition in the water. During eutrophication the loading of phosphorus becomes one of the

major driving forces behind bloom formation in fresh water (Paerl, 1988). Nutrient enrichment often results in the dominance of species of cyanobacteria and Dinophyta (Pearl & Tucker, 1995; Vos & Roos, 2005). A report by Van Ginkel *et al*. (2000) addressed the distribution of cyanobacterial blooms in selected impoundments across South Africa and found that summer blooms were particularly dominated by the presence of *Microcystis* spp., *Anabaena* spp. and *Oscillatoria* spp. Water quality managers are able to predict the likelihood of cyanophyte blooms by using empirical models based on the phosphorus loading of the system. It has been determined that the tendency of cyanobacterial blooms to occur increases when TN:TP ratios fall below 29:1, whereas the occurrence tends to be infrequent or absent when these values are exceeded (Shapiro, 1990; Smith, 1983). The mass development and persistence of single species is not only dependent on the reigning nutrient conditions, but rather on a combination of environmental and hydrological factors. Factors leading to the formation of blooms involve enrichment by terrestrial runoff, water column turnover, thermal stratification, and temperature and light conditions (Harding & Paxton, 2001; Shapiro, 1990; Smith, 1983; Van Ginkel *et al*., 2000). Disruptions in the thermal characteristics of the water favour mass appearances of single species (Harding, 1997; Huisman *et al*., 1999). Chlorophyta blooms tend to appear during colder conditions, while cyanobacterial blooms are common in water temperatures exceeding 20 °C (Robarts & Zohary, 1987; Bucka, 1989; Harding & Paxton, 2001). The intensity and frequency of mixing events in the water column has a major impact on bloom development and on the species dominating the bloom (Huisman *et al*., 1999). Light availability is essential for the photosynthetic activity of phytoplankton and light availability also varies with depth. Therefore, the displacement of species due to mixing events will cause different growth rates and subsequent shifts in the dominance of the population. Species that are tolerant of low “critical light intensity” will dominate in environments with a high mixing frequency. At low or infrequent mixing rates, species with a low “critical light intensity” will be displaced. The growth of cyanobacteria is also influenced by the stability of the water column (Harding & Paxton, 2001; Huisman *et al*., 1999). Thermal stratification of the water column ultimately results in nutrient stratification. During nutrient stratification the upper layers are poor in nutrients while nutrient levels from deeper layers are replenished from the sediment. Despite the nutrient rich conditions in deeper layers, low light penetration will limit cyanobacterial growth to surface water. In shallow, warm lakes that are subjected to a pattern of regular mixing, cyanobacterial biomass production will be stimulated (Harding, 1997).

4.1.4.2 Competitive advantages of cyanophytes

Certain bloom forming species have physiological mechanisms that give them a competitive advantage to out-compete other algae under environmental stress conditions. During periods of high water column stability, nutrients tend to be more concentrated in bottom waters. Under these conditions, the ability of cyanobacteria to regulate their vertical position between well-lit surface waters and nutrient rich bottom waters, gives them a competitive advantage over other non-motile plankton species. Cyanobacteria manage to regulate their buoyancy by controlling the collapse and reformation of intracellular gas vacuoles (Shapiro, 1990; Pearl & Tucker, 1995). Loss of buoyancy is created by the production of dense polysaccharides that enables them to sink to the sediments where

phosphorus is more readily available. The formation of gas vacuoles by suboptimal photosynthesis enables them to rise to the surface where CO_2 and nitrate are more available (Pearl & Tucker, 1995). Diurnal variation, to obtain optimal light and nutrient conditions, causes cyanobacterial cells to accumulate at the surface during daytime and in deeper layers during night time. Cyanobacteria comprise the ability to out-compete phytoplankton communities under poor light conditions by the formation of supplementary pigments and by increasing their cellular chlorophyll content (Bucka, 1989; Shapiro, 1990). Cyanobacteria contain mechanisms that allow them to assimilate atmospheric nitrogen and maintain high growth rates in conditions with low levels of inorganic nitrogen (Smith, 1983; Pearl & Tucker, 1995).

Cyanobacteria are able to out-compete other algae during periods when pH values are high and carbon dioxide concentrations are low, by intercepting carbon dioxide from the atmosphere (Paerl, 1988; Shapiro, 1990; Harding & Paxton, 2001). Cyanobacteria are adapted to a wide range of temperatures, causing high biomass densities both during winter and summer. They are especially well adapted to survive high water temperature when growth of other algae is limited (Robarts & Zohary, 1987). Some species have a competitive advantage over the rest of the assemblage pertaining predation by herbivorous zooplankton. Cyanobacteria prove to be of poor nutritional quality for filter feeding zooplankton. Their size is too large to be effectively consumed and digested (Pearl & Tucker, 1995). They are however grazed upon by rotifers and protozoa, leaving them more available for herbivorous zooplankton (Pearl & Tucker, 1995).

4.1.4.3 Implications to resource health

Phytoplankton blooms will ultimately result in three nuisance categories which can be distinguished as: the deterioration of visible water quality, health hazards and the loss of aesthetical and recreational significance of the resource (Paerl, 1988; Smith, 1988). Perceptible water quality is the most common measure by which consumers judge the quality of the resource. Unmanaged phytoplankton blooms can impair visible water quality by causing severe fluctuations in dissolved oxygen levels and pH, taste odour problems and by releasing toxic components from living cyanobacterial cells (Boyd *et al*., 1975).

During night time and low light conditions, photosynthesis and the production of oxygen ceases. At the same time phytoplankton and other aquatic organisms continue to consume large amounts of dissolved oxygen for respiration, causing dissolved oxygen concentrations to drop below critical levels (Paerl, 1988; Brunson *et al*., 1994; Pearl & Tucker, 1995). Photosynthetic activity during dense phytoplankton blooms also cause severe carbon dioxide depletion and are associated with raising the pH of the water resource (Talling, 1976; Erez *et al.*, 1990). Competition for light and nutrients can lead to sudden die-offs of a phytoplankton bloom. After the collapse of a bloom, dead algal material settles down to the sediment where bacterial decomposition takes place. Bacterial decomposition requires large amounts of oxygen, causing the hypolimnetic waters to become anoxic (Boyd *et al*., 1975; Boyd *et al*., 1978; Erez *et al*., 1990). Without the buffering capacity of oxygen in hypolimnetic waters, toxic substances such as nitrite, hydrogen sulphide and ammonia are released back into the water column.

Decomposition of dead algal blooms cause ammonia levels to rise, which even in small concentrations can be lethal to the culture fish species (Mortimer, 1941). Sudden die-offs of a bloom also result in the release of nutrients from decaying algal material, adding to the internal nutrient load of the dam.

4.1.5 Fish farming and eutrophication

4.1.5.1 Role of fish farming in eutrophication

Fish farming wastes, such as metabolic products and unconsumed feed, are released directly into the aquatic environment and assimilated by the water column and the underlying sediment. Additional input of organic and soluble wastes increase the rate at which nutrients are added and accumulate within the system (Stirling & Dey, 1990; Pecher, 2000; Diaz *et al*., 2001). Phytoplankton growth is directly influenced by the reigning nutrient conditions of the water source (Wetzel, 2001). Impacts of fish farming on water quality can therefore be reflected in the response of the phytoplankton communities to enrichment. This can be measured by changes in the species composition and biomass of the phytoplankton populations. To date only a few studies have focussed on phytoplankton assemblages forming mass appearances in aquaculture ponds. The most common species to form blooms in aquaculture ponds have been identified as dinoflagellates (*Ceratium hirundinella, Gymnodinium* spp.) and cyanobacteria (*Microcystis* spp.*, Anabaena* spp., *Oscillatoria* spp.) (Boyd *et al*., 1978; Brunson *et al*., 1994; Pearl & Tucker, 1995). Dinoflagellate blooms of *Gymnodinium* spp. have been found to coincide with high levels of urea, as they are able to utilise urea as a source of nitrogen (Bucka 1989; Gilbert & Terlizzi, 1999). A comparative study by Diaz *et al*. (2001) marked an increase in the biomass and density of phytoplankton after the introduction of a fish farm enterprise. The increased biomass and densities were ascribed to a shift in species dominance within the assemblage and were caused by an increase of cyanobacteria (*Anabaena spiroides*) and a decrease in nanoplanktonic (*Rhodomonas lacustris*) abundance. Other evidence of eutrophication such as *Ceratium hirundinella* blooms were also recorded in reservoirs downstream from the fish farm (Diaz *et al*., 2001). A similar study by Stirling & Dey (1990) found that the phytoplankton assemblage was dominated by *Microcystis aeruginosa*, which is also indicative of highly eutrophic systems.

The central investigative theme of this chapter pertains to phytoplankton communities within man made farm dams that are used for aquaculture. The chapter provides 1.5 year monitoring results of four Western Cape farm dams, of which two were used for net cage production of rainbow trout (*Oncorhynchus mykiss*) and two that did not harbour aquaculture. The objective of the monitoring was to observe seasonal variation pertaining to phytoplankton structure (composition, abundance, richness and diversity) and functioning (response to environmental change) of these aquatic ecosystems.

4.2 Materials and methods

Four farm dams were selected for the purpose of the study of which two contained net cage culture of rainbow trout (*Oncorhynchus mykiss*). The remaining two sites, also referred to as reference sites, contained no aquaculture and had no previous history of aquaculture.

To study the phytoplankton succession, whole water samples were collected with a water sampler every two weeks from June 2005 to November 2006. Samples were taken at three depths within the euphotic zone (0 m, 3 m, 6 m) and immediately fixated with Lugol-acetic solution for preservation and dyeing. Phytoplankton identification and counts were analysed with an inverted microscope according to the inverted microscope technique of Utermöhl (Lund *et al*., 1958). Cell biovolumes of individual species were calculated from appropriate geometric shapes (Hillebrand *et al*., 1999). Phytoplankton biomass was then estimated from density (assuming density of fresh algae to be 1 g/cm^3) and measurements of cell volume (Wetzel & Likens, 2000). Diversity analysis measurements included species richness and species diversity (Shannon & Weaver, 1949). Biomass was used for the diversity calculation (Shannon & Weaver, 1949; Wilhm, 1968; Figueredo & Giani, 2001).

Other related environmental variables included: temperature, water transparency, total suspended solids, pH, dissolved oxygen, dissolved inorganic nitrogen (NO_2 –N + NO_3–N + NH_3-N) and orthophosphate (as P). An Oxyguard MKIII (OxyGuard International) was used to measure dissolved oxygen (mg/L) and water temperature (ºC). The pH was recorded with a Hanna pH 211 Microprocessor. Turbidity was quantified using a standard sechhi disc (Wetzel & Likens, 2000). See Table 2.3 and 2.4 (Chapter 2) for complete methodologies applied. For the present study, the atomic ratios of N:P were calculated for dissolved inorganic nitrogen and soluble reactive phosphorous. Dissolved inorganic nitrogen (DIN) was calculated as ammonia nitrogen (NH_3-N) plus nitrate nitrogen (NO_3-N) and nitrite nitrogen (NO_2-N). The ratios obtained were then compared to the Redfield ratio to determine nutrient limitation (Redfield, 1958).

4.3 Results

4.3.1 Species identification

The total number of phytoplankton species identified between June 2005 and November 2006 was represented by 43 genera within 15 orders, 9 classes and 6 divisions. Table 4.1 lists the phytoplankton taxa identified. The Chlorophyta were the most significant group in terms of species number (72.1%), followed by Cyanophyta (16.2%) and Heterokontophyta (18.6%). Of the genera identified, 9.3% belonged to the division Cryptophyta, 9.3% belonged to the division Euglenophyta and 6.9% belonged to the division Dinophyta. Of the 15 orders identified, Chlorococcales was the most diverse with representatives from 11 genera.

Table 4.1: A list of the phytoplankton taxa identified in the reference and production sites from June 2005 to November 2006

Division	Class	Order	Genus	Species
CHLOROPHYTA	Chlorophyceae	Chlorococcales	*Ankistrodesmus*	*convolutus*
			Ankistrodesmus	*spiralis*
			Coelastrum	spp.
			Crucigenia	*quadrata*
			Crucigenia	*apiculata*
			Kirchneriella	spp.
			Kirchneriella	spp.
			Nephrocytium	spp.
			Oocystis	spp.
			Oocystis	spp.
			Pediastrum	*biradiatum*
			Pediastrum	*duplex*
			Quadrigula	*lacustris*
			Scenedesmus	*bijuga*
			Scenedesmus	*acuminatus*
			Scenedesmus	*cavinatus*
			Scenedesmus	*arcuatus*
			Schroederia	*spiralis*
			Selenastrum	spp.
			Tetraedron	*caudatum*
		Volvocales	*Chlamydomonas*	spp.
			Eudorina	spp.
			Pseudosphaerocystis	spp.
		Tetrasporales	*Elakatothrix*	*gelatinosa*

	Zygnematophyceae	Desmidiales	*Closterium*	spp.
			Cosmarium	spp.
			Staurastrum	*gracile*
			Staurastrum	*gemelliparum*
			Micrasterias	spp.
		Zygnematales	*Spondylosium*	spp.
CYANOPHYTA	Cyanophyceae	Chroococcales	*Aphanocapsa*	spp.
			Merismopedia	spp.
			Merismopedia	*elegans*
			Microcystis	spp.
		Oscillatoriales	*Oscillatoria*	spp.
			Oscillatoria	*limnetica*
		Nostocales	*Anabaena*	*circinalis*
CRYPTOPHYTA	Cryptophyceae		*Chroomonas*	spp.
			Cryptomonas	*erosa*
			Cryptomonas	*ovata*
			Rhodomonas	*lacustris*
DINOPHYTA	Dinophyceae	Peridiniales	*Ceratium*	*hirundinella*
			Peridinium	spp.
		Gymnodiniales	*Gymnodinium*	spp.
HETEROKONTOPHYTA	Xanthophyceae	Tribonematales	*Tribonema*	*affine*
	Bacillariophyceae	Pennales	*Nitzschia*	spp.
			Fragilaria	spp.
			Frustulia	spp.
			Navicula	spp.
		Centrales	*Cyclotella*	*stelligera*
			Aulacoseira	*granulata*
	Chrysophyceae	Chromulinales	*Dinobryon*	*divergens*
EUGLENOPHYTA	Euglenophyceae	Euglenales	*Euglena*	*oxyuris*
			Phacus	*platalea*
			Trachelomonas	spp.
			Trachelomonas	spp.

Following Van Den Hoek *et al*., 1995

Based on monthly mean biomass at all depths, production site 2 showed the highest biomass (x = 2.56 mg/L, n = 18), followed by production site 1 (x = 2.07 mg/L, n = 18) and reference site 1 (x = 1.4 mg/L, n = 18). Reference site 2 sustained the lowest biomass (x = 0.60 mg/L, n = 18) for the duration of the study.

4.3.2 Seasonal distribution of biomass and species composition

4.3.2.1 Production site 1 (Nietvoorbij Dam)

The seasonal variation and abundance of the phytoplankton community in production site 1 is shown in Figure 4.1. The highest values for biomass in production site 1 was measured in April 2006 (9.2 mg/L) and the lowest biomass (0.2 mg/L) was measured in July 2005. During periods of high biomass the phytoplankton biomass was co-dominated by *Ceratium hirundinella* and *Gymnodinium* spp. The most dominant group, in terms of contribution to total biomass, was the dinophytes (40%). Dinoflagellates (mostly of the genera *Ceratium* and *Gymnodinium*) were low in numbers, but due to their large size, were major contributors to the total phytoplankton biomass. The second most important group was the cyanophytes which contributed 26 % of the total biomass. The chlorophytes were the most diverse group (*Staurastrum gracile, Oocystis* spp.*, Scenedesmus bijuga, Nephrocytium* spp.*, Elakatothrix gelatinosa, Eudorina* spp.*, Coelastrum* spp.*, Dinobryon divergens, Ankistrodesmus convolutus*) and contributed 14% to the total biomass. Diatoms were never dominant and accounted for only 1% of the total biomass. Cyanophyta genera (*Anabaena* spp.*, Microcystis* spp.*, Merismopedia* spp., *Spirulina* spp.) were dominant on occasions, particularly in late spring and summer. As a whole, the phytoplankton biomass of production site 1 decreased with an increase in depth. In terms of contribution to the total phytoplankton biomass, the cyanophyte contribution increased with an increase in sampling depth. In surface samples their contribution to the total biomass comprised 21%, whereas in the deeper layers their contribution was 34%. The increase in the contribution of cyanophytes to total biomass was due to an increase in the occurrence and abundance of *Microcystis* spp. and *Anabaena* spp. Euglenophytes appeared to be more abundant with an increase in depth and were primarily represented by *Trachelomonas* spp.*, Phacus longicauda* and *Euglena oxyuris.* In surface water, euglenophyte biomass contributed to 3.5% of the total biomass and at deeper sampling stations, 17.3%. The contribution of the remaining groups (cryptophytes, chlorophytes, dinophytes, and heterokontophytes) to the total biomass decreased with an increase in sampling depth.

The total phytoplankton biomass of production site 1 indicated two separate peaks during the study period (Figure 4.1). At the beginning of the study, the phytoplankton assemblage of production site 1 was numerically and in terms of biomass dominated by the presence of *Ceratium hirundinella* and *Gymnodinium* spp. These two genera made up 0.72 mg/L of the total biomass measured in June 2005. With the onset of spring in September 2005, representatives of the group Cyanophyta (0.5 mg/L) and Chlorophyta (0.4 mg/L) increased and became the major contributors to the diverse spring peak in October 2005 (R_s = 0.80; p = 0.0001) and (R_s = 0.74; p = 0.0007) respectively. Another important contributor was the euglenophytes which dominated samples at deeper sampling depths. Cyanophyte contribution to biomass (60%) exceeded that of other groups during December 2005. Other important cyanobacteria included *Merismopedia elegans* and *Spirulina* spp. Cyanobacteria was most abundant at surface layers and at 6 m.

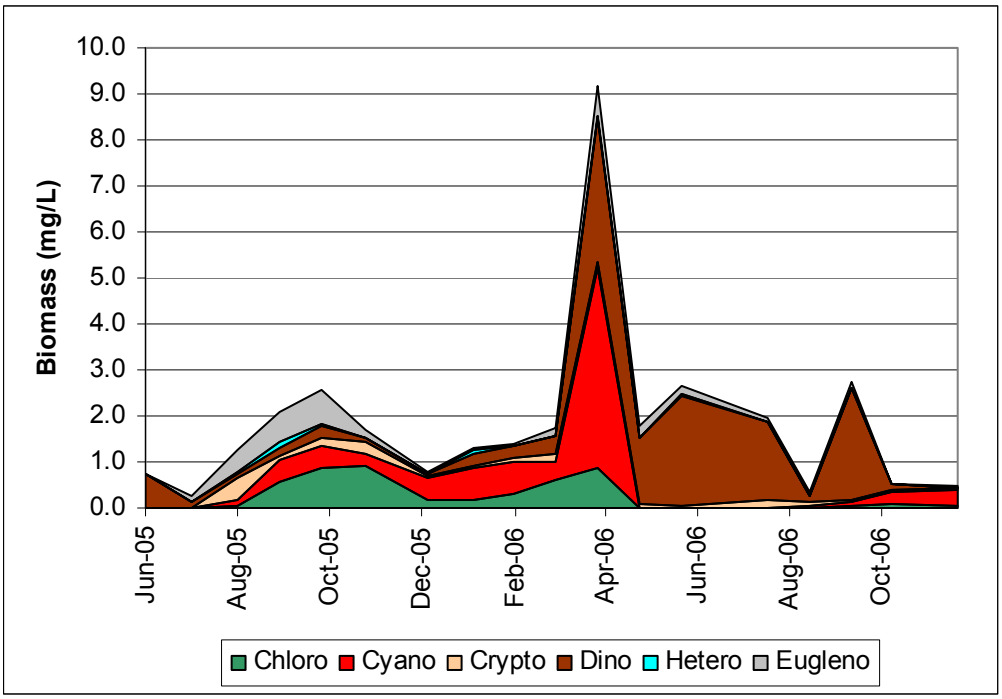


Figure 4.1: Seasonal changes in composition of the phytoplankton populations in production site 1 from June 2005 to November 2006. (Chloro = Chlorophyta; Cyano = Cyanophyta; Crypto = Cryptophyta; Dino = Dinophyta; Hetero = Heterophyta; Eugleno = Euglenophyta)

The second peak in chlorophyte biomass was observed in April 2006 with the desmid, *Staurastrum gracile,* being the most significant contributor, numerically (23 cells/mL) and in terms of biomass (1.29 mg/L). During April 2006, cyanophyte biomass increased to reach a yearly maximum (4.3 mg/L). The cyanophyte biomass was dominated by *Microcytis* spp. and *Merismopedia elegans* and was most abundant at 3 m. During the winter period the most dominant phytoplankton species, both numerically and in terms of contribution to biomass, was *Ceratium hirundinella* (3 mg/L). This coincides with a period of P-limitation during July and August. The dominance of *Ceratium hirundinella* persisted throughout the winter until September 2006. *Ceratium hirundinella* populations were evenly distributed at surface and deeper layers. In October 2006, following a decline in *Ceratium hirundinella* numbers, cyanophyte species increased. Cyanophyte species present included *Anabaena* spp., *Microcystis* spp. and *Merismopedia* spp. Euglenophytes were present during most of the study period, especially in the deeper water layers. The genus, *Trachelomonas*, was numerically the most abundant (2 – 59 cells/mL) of the euglenophytes. During autumn and spring the large euglenophyte, *Euglena oxyuris,* was present with biomass ranging between 0.03 mg/L – 0.18 mg/L.

Seasonal patterns were observed for some individual species and genera. *Staurastrum gracile* and *Coelastrum* spp. increased in both years during the onset of spring, after being absent for the duration of winter. The major contributors to biomass, *Ceratium hirundinella* and *Gymnodinium* spp. exhibited seasonality, with an increase in abundance during autumn, a peak abundance during winter (June,

July, August) and a decline in September. Cyanobacteria so displayed a seasonal pattern of development in spring with the onset of stratification, reaching maximum values in December.

4.3.2.2 Reference site 1 (Poplar Dam)

Figure 4.2 shows the seasonal succession of the main groups of phytoplankton identified in reference site 1 from June 2005 to November 2006. Total phytoplankton biomass reached its highest value during September 2005 (6.6 mg/L) and lowest value in November 2006 (0.02 mg/L). The dinophytes appeared to be the most dominant group and accounted for 47% of the total biomass. Within the dinophytes, the most abundant genus was *Peridinium.* Chlorophytes, the second most abundant group, comprised 24% of the total biomass, with representatives of the genera *Tetraedron* spp.*, Eudorina* spp.*, Scenedesmus* spp.*, Nephrocytium* spp.*, Staurastrum* spp.*, Pediastrum* spp.*, Elakatothrix* spp.*, Dinobryon* spp.*,* and *Spondylosium* spp. Other contributors to phytoplankton biomass were cyanophytes (11%), heterokontophytes (2%) and cryptophytes (3%). In temrs of biomass, the most abundant cyanophyte species were *Oscillatoria* spp. and *Anabaena* spp. The abundance of the Euglenophyta increased with an increase in depth. In surface layers, Euglenophyta biomass accounted for 1.5% compared to 41% in samples from deeper depths. Euglenophytes were represented by *Euglena oxyuris, Trachelomonas* spp. and *Phacus platalea*.

At the start of the study, the phytoplankton biomass of references site 1 was dominated by a high dinophyte biomass (1.3 mg/L), with the genus *Peridinium* the most abundant (Figure 4.2). *Peridinium* were most abundant at the surface and decreased with an increase in depth. After a slight decline in *Peridinium* biomass, dinophytes continued to dominate the system during October 2005. The spring chlorophyte assemblage started to develop in August 2005 and reached a yearly maximum biomass during September 2005 (2.5 mg/L). The chlorophyte biomass was more abundant at 3 m of depth than at the surface and its lowest abundance was measured at 6 m depth. The concentration of chlorophytes at the surface coincided with alkaline pH values (> 8.3). In terms of biomass, the most important species was the colonial *Eudorina* spp. and the desmid *Staurastrum gracile.* At the same time, cyanophytes co-dominated the system with *Anabaena* spp. as the most abundant. DIN:SRP ratios indicated that reference site 1 was strictly N-limited (DIN:SRP = 2) during this period. The ability of nitrogen fixing cyanophytes, especially *Anabaena* spp.*,* to overcome N-deficit conditions together with increasing water temperatures favoured the growth of cyanophytes. However, biomass corresponded more significantly to an increase in water temperature (R_s = 0.47; p = 0.05). Only one peak in cyanophyte biomass was observed in September 2005 and contributed 36% of the total biomass. In terms of biomass, the major species responsible was the nitrogen fixing *Anabaena circinalis* (3.7 mg/L). During the cyanophyte peak, they were most abundant in the surface layers and at a depth of 3 m, as opposed to 6 m of depth. At the same time, samples from deeper water layers were dominated by the presence of euglenophytes such as *Euglena oxyuris, Trachelomonas* spp. and *Phacus platalea*. Chlorophytes increased to dominate the assemblage during November 2005. From December 2005 to March 2006 the total biomass remained low and ranged between 0.4 mg/L and 0.6 mg/L. During this period the assemblage was occasionally dominated by euglenophytes and

dinophytes. Euglenophytes (*Trachelomonas* spp.*; Euglena* spp.) were spatially more abundant at a depth of 6 m, whereas dinophytes (*Peridinium* spp.) dominated in surface samples. The latter dominance remained until February 2006, although total biomass decreased. In April 2006, the dinophytes re-emerged in the assemblage. The high dinophyte biomass after turnover in May, corresponded with P-limiting conditions (DIN:SRP > 16) during June, July and August. They remained dominant until June 2006, during which time they reached a yearly maxima of 3.6 mg/L. The dinophytes were spatially the most abundant at 0 m and decreased towards deeper layers. In August 2006, following a population crash, they remained low ranging between 0.01 mg/L and 0.1 mg/L. During June and July 2006, the phytoplankton community was co-dominated by cryptophytes, most notably *Cryptomonas erosa*. *Cryptomonas erosa* was most abundant in samples from 3 m of depth. From August 2006 onwards the phytoplankton population mainly constituted of members of all the divisions, except cyanophytes. The most important contributors were the euglenophytes and chlorophytes.

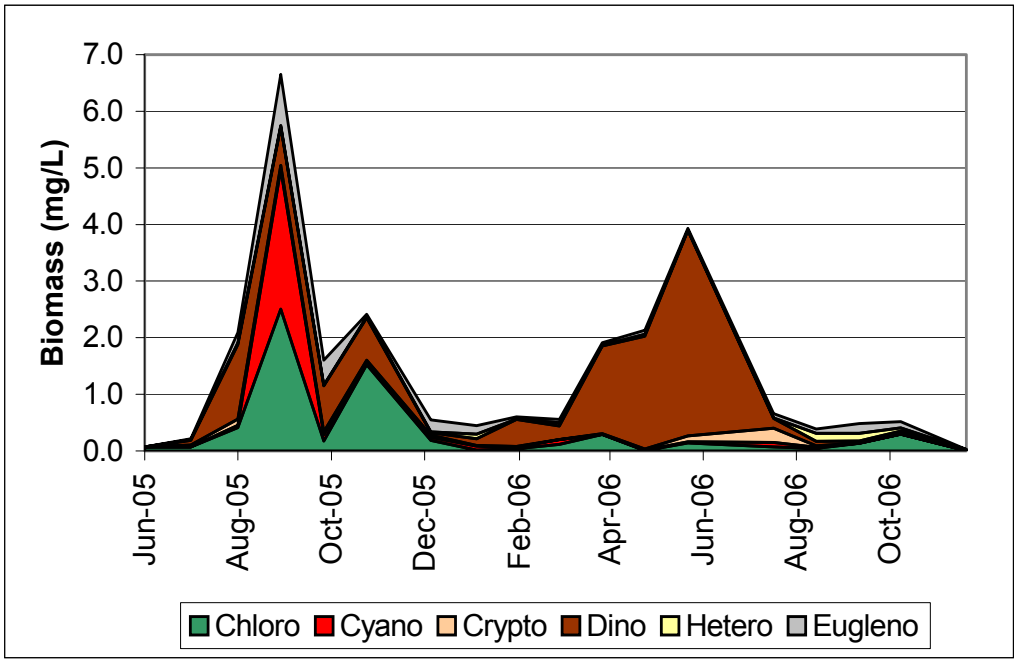


Figure 4.2: Seasonal changes in composition of the phytoplankton populations in reference site 1 from June 2005 to November 2006 (Chloro = Chlorophyta; Cyano = Cyanophyta; Crypto = Cryptophyta; Dino = Dinophyta; Hetero = Heterophyta; Eugleno = Euglenophyta)

4.3.2.3 Production site 2 (John Smith Dam)

The changes in seasonal succession and abundance of the phytoplankton populations in production site 2 are presented in Figure 4.3. The phytoplankton biomass of production site 2 ranged between 0.1 mg/L and 6.4 mg/L, with the highest biomass observed during April 2006. The total phytoplankton biomass in production site 2 decreased with an increase in sampling depth. In terms of biomass, cyanophytes were clearly the most important taxonomic group contributing 47% of the total biomass.

Among the cyanophytes, the two most abundant genera were *Microcystis* and *Anabaena*. The cryptophytes (mainly *Cryptomonas erosa*) represented 15% of the total biomass. The heterokontophytes contributed a mere 7% of the total biomass. The chlorophytes were the most diverse with major contributor species from the genera *Scenedesmus, Eudorina, Cosmarium, Staurastrum, Schroederia* and *Coelastrum*. All the taxonomic groups, with the exception of Heterokontophyta and Euglenophyta, showed a decrease in their contribution to total biomass with an increase in sampling depth. Euglenophyta species of the genera *Trachelomonas, Phacus* and *Euglena* were found in higher densities in deeper sampling layers and comprised 5% of the total biomass.

Cryptophytes, in particular *Cryptomonas erosa,* dominated the winter phytoplankton assemblage in production site 2 (Figure 4.3). Their dominance persisted from June 2005 (97%) to July 2005 (89%) with a sudden decline in biomass contribution during August 2005. The high abundance of cryptophytes was restricted to the surface layers. The spring assemblage of 2005 was dominated by chlorophytes (2.2 mg/L), which contributed 37% to the total phytoplankton biomass. Among the spring chlorophyte species, the most significant was the colonial *Eudorina* spp. The highest concentration of chlorophytes was found at 3 m. The spring assemblage of chlorophytes was co-dominated by cyanophytes (3.0 mg/L) of the genera *Microcystis* and *Anabaena.* Temperature correlations indicated a significant positive relationship between cyanophyte biomass and an increase in water temperature (R_s = 0.74; p = 0.0007). The cyanophyte biomass increased steadily from August 2005 to a sudden decline in October 2005. At the same time, a strong peak in biomass (1.8 mg/L) of the Heterokontophyta was observed. This group dominated the assemblage during October 2005 with the most significant contribution by the genus, *Tribonema* (*Tribonema affine*). A rapid increase in cyanophyte biomass to reach maximum values (5.3 mg/L) in December 2005 took place, followed by a sudden crash of the population. In December 2005, production site 2 was strictly N-limited (DIN:SRP = 1.08) and as a result favoured the growth of cyanophytes. A significant positive correlation between % contribution to biomass and temperature was detected (R_s = 0.5; p = 0.03). The correlation between DIN:SRP and cyanophyte biomass was negative, however not significant (*Rs* = -0.19; p = 0.46). Thus, as DIN:SRP ratios decreased, cyanophyte biomass increased. During this cyanophyte peak, the cyanophyte biomass was found in the surface layers where they were able to fixate atmospheric nitrogen. In February 2006, the biomass values were generally very low ranging between 0.05 mg/L and 0.17 mg/L. During this time the Dinophyta (mainly *Gymnodinium* spp.) made an appearance and dominated the assemblage. The chlorophytes once again peaked during April 2006, making up 50% (3.2 mg/L) of the total biomass. This peak was due to an increase in the abundance of the chlorophyte genus, *Cosmarium* which was most abundant at 3 m. During March and April 2006, the euglenophytes (*Phacus platalea, Euglena oxyuris,* and *Trachelomonas* spp.) and cyanophytes (*Anabaena* spp.) also became important contributors until the replacement of both groups by cryptophytes. Euglenophytes were more evident at 3 m and remained present from April 2006 until June 2006. This can possibly be explained by the decrease in water depth (< 3 m) during the same period. *Cryptomonas erosa* again became the dominating species during June 2006 (84%) and July 2006 (71%), with highest

concentrations at the surface. A correlation analysis supported the increase in *Cryptomonas erosa* abundance with a decrease in water temperatures (R_s = -0.62; p = 0.008). Alkaline pH values were also recorded in surface waters that correlated with the concentration of *Cryptomonas erosa* biomass at the surface (R_s = 0.50; p = 0.04). The abundance of Cryptophyta decreased during August 2006. As water temperatures increased in early spring, cyanophytes became more evident (1.3 mg/L-1.8 mg/L) and dominated the assemblage from September 2006 onwards (R_s = 0.74; p = 0.0007).

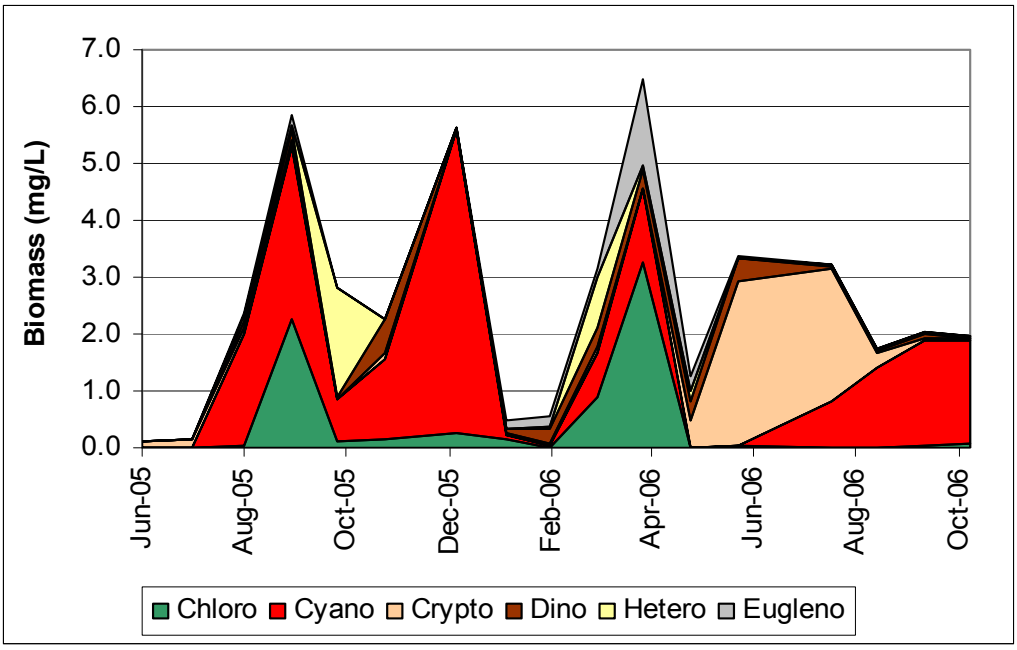


Figure 4.3: Seasonal changes in composition of the phytoplankton populations in production site 2 from June 2005 to November 2006 (Chloro = Chlorophyta; Cyano = Cyanophyta; Crypto = Cryptophyta; Dino = Dinophyta; Hetero = Heterophyta; Eugleno = Euglenophyta)

Cryptomonas erosa exhibited a seasonal pattern as they dominated the assemblage during both the winter periods. Other individual species and genera that showed a seasonal occurrence included: cyanophytes of the genera *Microcystis* spp. with co-domination by *Anabaena* spp. in August and September of both 2005 and 2006.

4.3.2.4 Reference site 2 (Garden Dam)

Reference site 2 had the lowest biomass throughout the entire study with values ranging between a minimum total biomass of 0.017 mg/L during June 2005 and a maximum of 1.89 mg/L in July 2006 (Figure 4.4). The major contributor to the total biomass in reference site 2 was the taxonomical group Dinophyta (34%), primarily represented by the genera, *Peridinium* and *Ceratium.* Periodically, heterokontophytes (19%) and cyanophytes (4.6%) were numerous. Representative genera included *Merismopedia, Oscillatoria, Anabaena, Tribonema, Fragilaria* and *Cyclotella.* The most diverse group, the chlorophytes contributed 13% to the total phytoplankton biomass and representative genera included *Staurastrum, Micrasterias, Scenedesmus, Nephrocytium, Dinobryon, Spondylosium,*

Schroederia, Crucigenia, and *Ankistrodesmus*. The contribution to total biomass of most taxonomic groups decreased in deeper water layers. Only members of the Euglenophyta increased numerically and in terms of biomass, with an increase in sampling depth. Among the euglenophytes the most important species included *Euglena caudata* and *Trachelomonas* spp. Their contribution to total biomass increased from 9% in surface samples to 31% in deeper samples.

During the course of the study, total phytoplankton biomass of reference site 2 showed several distinct peaks. At the start of the study the phytoplankton assemblage was dominated by dinophytes, particularly species of the genera *Peridinium (Peridinium* spp.*; Ceratium hirundinella).* Dinophyte biomass was the most abundant in surface samples (0.9 – 0.6 mg/L) and least abundant at 6 m (0.08 mg/L). These species dominated the system until a sudden decline in October 2005 and remained in low abundance (0.003 – 0.1 mg/L) until May 2006. From October onwards, the heterokontophytes became more important contributors to total biomass. The high Heterokontophyta biomass was caused by an increase in the biomass of species belonging to the class Chrysophyceae (*Dinobryon divergens).* The Heterokontophyta biomass decreased from being most abundant in surface samples to its lowest abundance at a depth of 6 m. Occasional biomass peaks by *Cryptomonas erosa* were observed during September and October 2005. Cryptophyta biomass was evenly distributed over all depths. The summer assemblage in December 2005 was co-dominated by euglenophytes (0.34 mg/L) and heterokontophytes (0.27 mg/L). The most significant species was from the genera *Nitzschia, Dinobryon (Dinobryon divergens), Fragilaria, Cyclotella* (*Cyclotella stelligera*) and *Tribonema* (*Tribonema affine*). The Heterokontophyta dominance was followed by a small increase in the biomass of cyanophytes (*Merismopedia elegans, Oscillatoria* spp.*, Anabaena* spp.). The euglenophyte population (*Trachelomonas* spp.*, Euglena* spp.) returned for a second peak in March 2006 (0.8 mg/L). Euglenophytes are usually more prominent in bottom waters. The increased abundance of euglenophytes in reference site 2 can partly be ascribed to the decrease in water depth (< 3m). Chlorophyte biomass remained low for the duration of the study, but increased to reach its highest biomass values during May, June and July 2006. For much of March, June and July 2006, the abundance of cyanophytes was greater at 3 m than at either the surface or at 6 m (0.2 – 0.3 mg/L). During May, June and July the assemblage was dominated by *Ceratium hirundinella* and *Peridinium* spp. These species dominated the surface and 3 m depth samples until a decline in August 2006. From May 2006 onwards, the heterokontophytes re-emerged and continued to be an important contributor to the total biomass with values ranging between 0.1 mg/L and 0.3 mg/L.

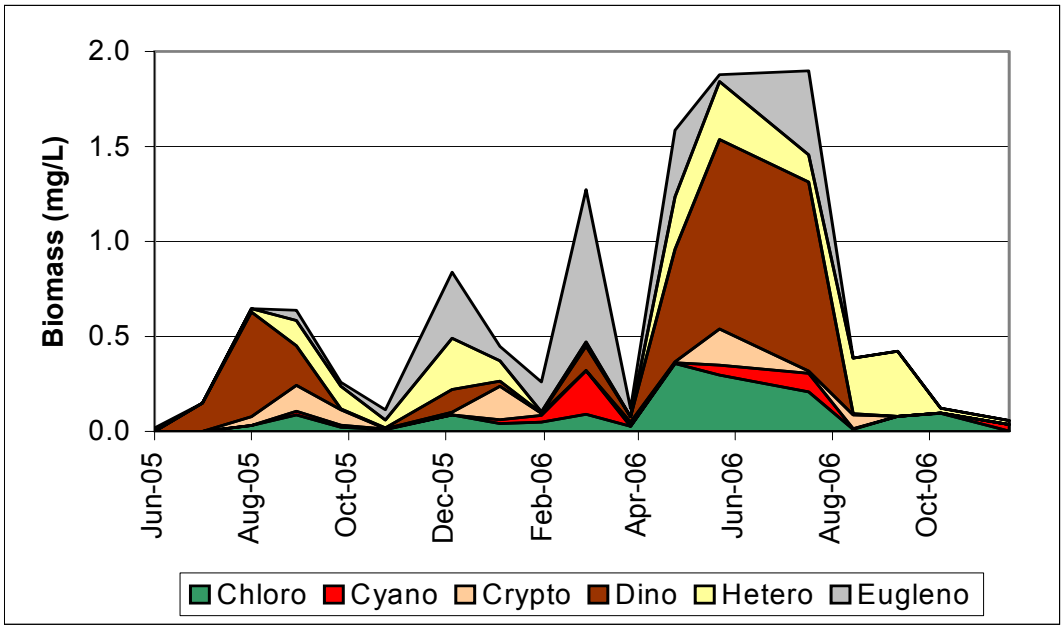


Figure 4.4: Seasonal changes in composition and biomass of the phytoplankton populations in reference site 2 from June 2005 to November 2006 (Chloro = Chlorophyta; Cyano = Cyanophyta; Crypto = Cryptophyta; Dino = Dinophyta; Hetero = Heterophyta; Eugleno = Euglenophyta)

Only one group of species exhibited a seasonal pattern of occurrence. The dinophyte species, *Ceratium hirundinella* and *Peridinium* spp., appeared during the winter months of both 2005 and 2006. Their populations started to develop in June, increased to reach a maximum density in August and persisted until a break down in September.

4.3.3 Nutrient limitation

The atomic ratios of dissolved inorganic nitrogen and soluble reactive phosphorous varied between 0.6 and 34. According to the literature, freshwater systems are N-limited when N:P ratios fall below 16, whereas phosphorus is the limiting nutrient when ratios exceed 16 (Moss, 1998; Wetzel, 2001). Based on comparisons with the Redfield ratio, the data from both production and reference sites showed either N or P limitation of phytoplankton growth. Based on DIN:SRP ratios, nitrogen was the most limiting nutrient in all the dams, especially during the summer months. P-limitation appeared more frequently during winter. High N:P levels could be ascribed to the high DIN levels in winter, because total phosphorus values measured were already indicating eutrophic conditions. Nitrate and nitrite levels were fairly low but ammonia increased throughout the water column during continuous mixing in winter months (Chapter 3). Although mostly N-limited, the incidence of P-limitation was higher in reference site 2 and production site 2.

4.3.4 Species richness and species diversity

Species richness and Shannon-Weaver diversity indices were calculated following the inspection of phytoplankton samples. Results signified that production site 1 presented the highest species richness (10.3 ± 5.9, *n* = 17), followed by production site 2 (7.2 ± 3.4, *n* = 17) (Figure 4.5). The species richness values in production site 1 were lowest during the winter months of both 2005 and 2006. This partly coincides with the period of fish farming. During the same period, the phytoplankton community was co-dominated by two Dinophyta species, *Ceratium hirundinella* and *Gymnodinium* spp. (high dominance values and low species richness). A significant negative relationship (R_s = -0.79; p = 0.0002) was found between the contribution of Dinophyta and species richness, thus supporting the high dominance values and low species richness in production site 1. However, species richness did not remain low for the entire production season as other species started to appear towards October and during turnover in May 2006. The maximum number of species in production site 1 was identified in late spring of 2005 (November) and included species from 15 genera. The species diversity in production site 1 also indicated a highest calculation of 1.81 during this period (Figure 4.6) with the group Chlorophyta being the most diverse. The lowest species diversity was estimated during August 2005, when the population was largely dominated by *Anabaena circinalis*. This finding is supported by a low species richness calculated for June 2005.

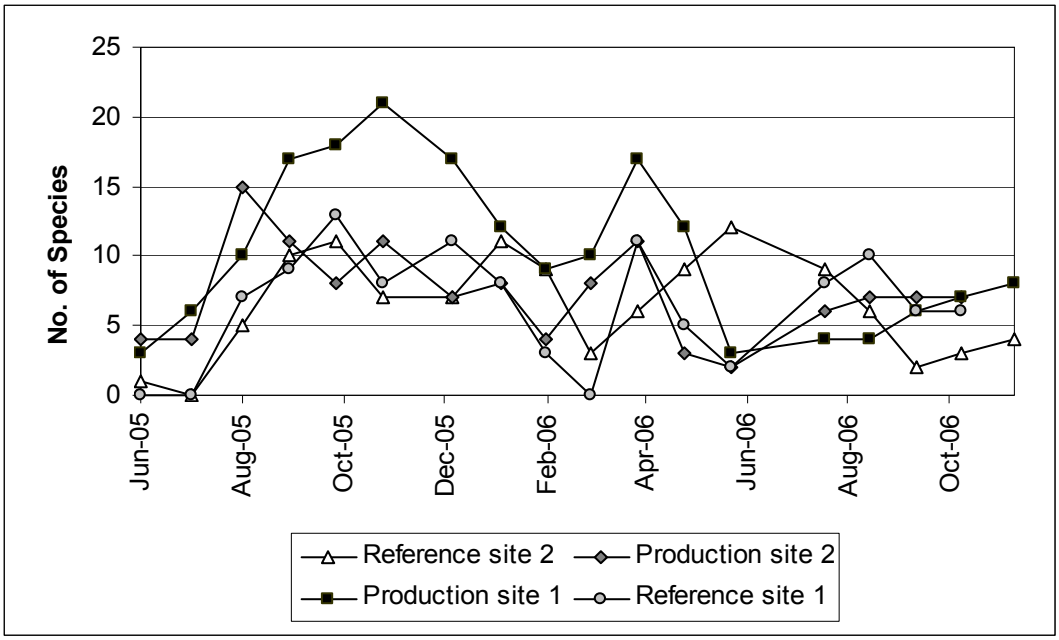


Figure 4.5: **Number of species identified in the reference and production sites from June 2005 to November 2006**

The species richness measured in production site 2 showed a similar pattern to that of production site 1. The months of June and July 2005 were characterised by low species richness and low species diversity, during which the population was dominated by the cryptophytes (Figure 4.6). The production season in 2006 also coincided with low species richness and concurred mainly with a mass appearance of *Cryptomonas erosa* and the cyanophyte *Microcystis* spp. (high dominance values and low diversity). A significant negative correlation (R_s = -0.63; p < 0.05) was calculated between species richness and *Cryptomonas erosa* biomass contribution, thus supporting the low species richness during periods of high dominance. Species diversity calculations in production site 2 were higher during the recovery periods (October through March). The highest species diversity was measured during January 2006 with species from 8 genera identified (Figure 4.6).

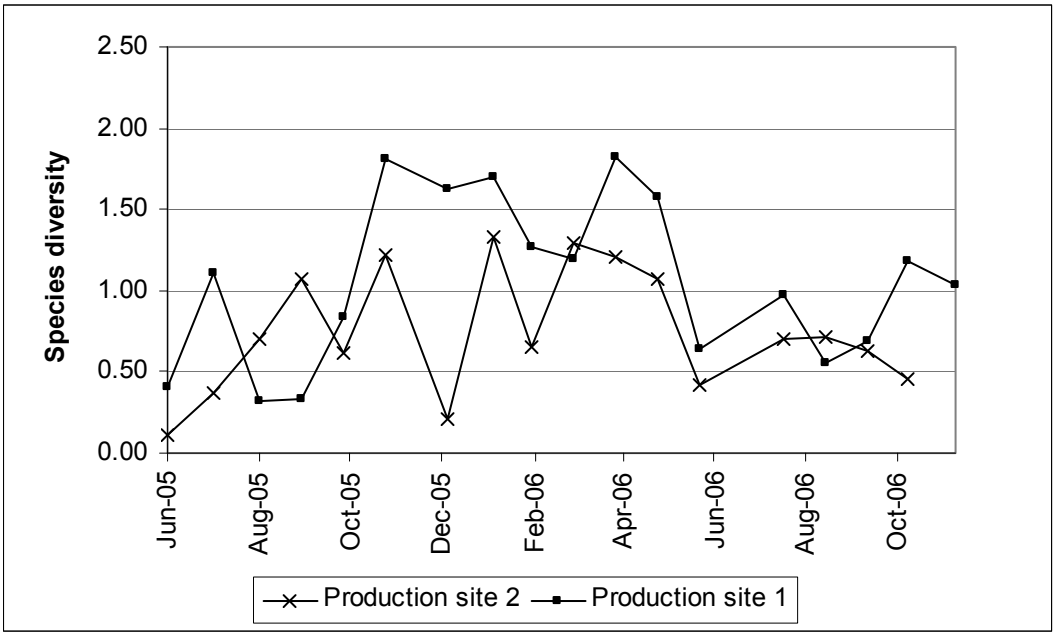


Figure 4.6: **Shannon diversity (based on biomass as unit) in production sites between June 2005 and November 2006**

Reference site 1 (6.5 ± 3.7, n = 17) and reference site 2 (5.9 ± 4.3, n = 17), showed the lowest species richness (Figure 4.7). Reference site 2 showed its lowest richness and diversity during September 2006, during which time the population biomass was dominated by the chlorophyte *Dinobryon divergens*. The highest species richness in reference site 2 was measured during June 2006, with most of the represented genera from the division Chlorophyta present (R_s = 0.36; p > 0.05). The highest species diversity was however, found in April 2006. No visible increase or decline in species numbers and species diversity was observed during the production period in 2005 and 2006. Reference site 1 displayed its lowest species richness and species diversity in June 2006 during which time the assemblage biomass was dominated by *Gymnodinium* spp. (high dominance values and low diversity) (R_s = -0.15; p > 0.05). Reference site 1 showed a maximum number of species in October 2005, however, the highest species diversity was observed during August 2006 (Figure 4.7).

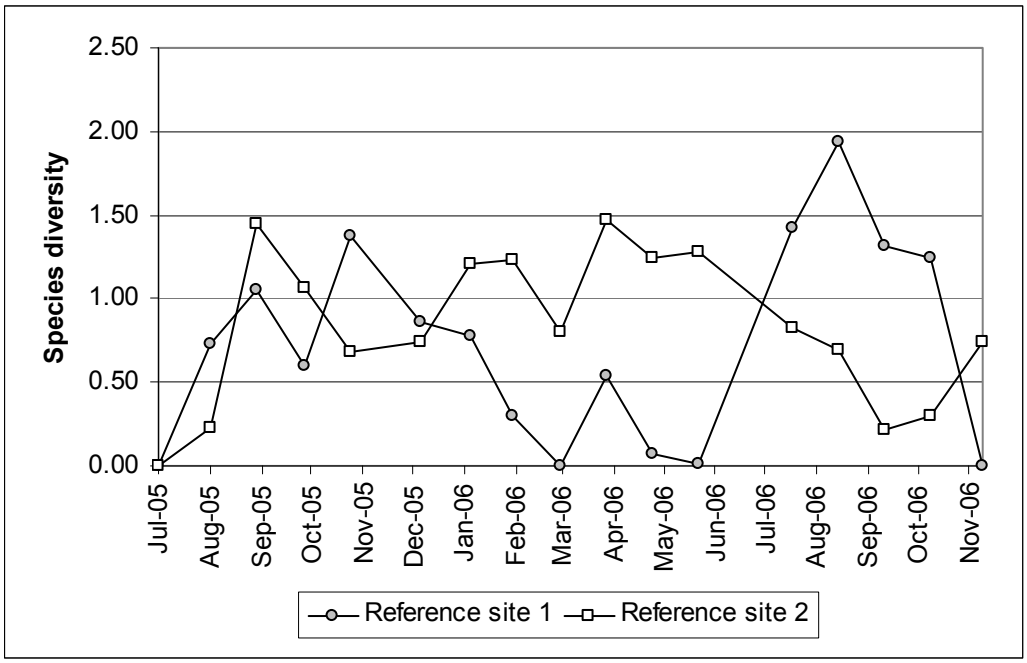


Figure 4.7: **Shannon diversity (based on biomass as unit) in reference sites between June 2005 and November 2006**

4.4 Discussion

One of the goals of this study was to assess species composition and abundance of phytoplankton in farm dams used for aquaculture. The following discussion considers taxa enumeration of species, species richness and diversity, phytoplankton abundance in terms of biomass and seasonality of dominant taxa.

The study identified phytoplankton species from a total of 43 genera and six classes of which the group, Chlorophyta was the most diverse. Species richness and diversity calculations of production sites pointed out that a decline of these indices concurred partly with the fish farming season, especially in mid winter (June, July, August). Determination of species richness and diversity in reference sites did not demonstrate a consistent decline or increase between winter and summer. In most of the dams low species richness and diversity were evident. In production sites low, species richness and diversity were ascribed to massive developments of *Microcystis* spp.*, Cryptomonas erosa, Ceratium hirundinella*, *Gymnodinium* spp. and *Anabaena circinalis*. In reference sites, single species dominance was caused by *Dinobryon divergens*, *Peridinium* spp. and *Ceratium hirundinella*. All the dams, with the exception of reference site 2, showed a high degree of species diversity during May. This coincided with the annual holomictic mixing event during which the whole water column mixes and nutrients are recycled from the enriched bottom waters (Ashton, 1985; Imberger, 1985; Allanson *et al*., 1990). Adequate phosphorus and inorganic nitrogen concentrations could have favoured a change in the species composition by shifting the population back to an earlier stage of succession with a higher degree of diversity (Ashton, 1985).

In terms of biomass in the farm dams studied, the highest biomass was found during the winter months. Seasonal trends pertaining to high phytoplankton biomass in winter has been widely documented in studies on large reservoirs in Southern Africa (Ashton, 1985; Allanson *et al*., 1990; Hart, 2006). Biomass started to increase after the annual holomictic overturn in May and remained high for the rest of winter. The winter monomictic regime of these reservoirs (Chapter 3) is responsible for a continuous upward movement of nutrients to the well-lit surface, creating favourable conditions for development and sustaining of high algal biomass (Haynes, 1973; Tilman *et al*., 1982; Imberger, 1985). In references sites winter maximums were caused by dinophytes, especially *Ceratium hirundinella* and *Peridinium* spp. In production sites, winter populations were dominated by species of the dinophytes, cyanophytes and cryptophytes. Summer months were characterised by lower biomass values, with the exception of production site 2 which experienced high cyanophyte biomass. The persistence of cyanophytes in this dam was influenced by a combination of fish farming activities as well as hydrological changes in the dam. Despite the observed seasonal trend in all the dams studied, the phytoplankton production was almost twice more abundant in production sites as opposed to the reference sites. The introduction of additional nutrients by fish farming provided the resident phytoplankton assemblages with sufficient nutrients to sustain biomass of a higher magnitude. This

response of higher biomass, particularly in the vicinity of fish farming, has previously been observed and is well documented (Smith, 1988; Sterling & Dey, 1990; Guo & Li, 2003).

Seasonality of individual species was observed at all the sites. The primary species that illustrated a seasonal pattern were *Ceratium hirundinella* in winter and spring, *Cryptomonas erosa* in winter and *Staurastrum gracile* with the onset of spring. *Ceratium hirundinella* is a unicellular dinoflagellate that is widespread throughout fresh water ecosystems and has been recorded in numerous South African reservoirs (Van Ginkel *et al*., 2001; Van Ginkel *et al*., 2007). Light intensity, growth temperature (5°C to 30°C) and sufficient nutrients were cited as the most important factors controlling bloom formation. *Ceratium hirundinella* has been recorded to be a predominately summer species, however, results from this study indicated its dominance appeared after the annual turnover of the water column and remained during winter and spring. Investigations of the Hartbeespoort Dam found a similar dominance by *Ceratium hirundinella* in winter (Whittington *et al.*, 2000; Van Ginkel *et al*., 2001). Van Ginkel *et al*. (2001) suggested that bloom formation was due to the adequate availability of inorganic nitrogen during winter months. According to Rosén (1981), *Ceratium hirundinella* is commonly found in mesotrophic to eutrophic systems. In the winter months, production site 1 and reference site 2 were eutrophic with TP concentrations of 0.1 mg/L (DWAF, 1996c). *Ceratium hirundinella* is able to utilise both inorganic and organic phosphorus thus enabling the dinoflagellate to grow when inorganic phosphorus levels are inadequate for other species (Bucka, 1989; Moss, 1998). According to DIN:SRP calculations, production site 1 and reference site 2 were P-limited (DIN:SRP > 16) during the winter months (June, July and August) which coincided with high *Ceratium hirundinella* biomass in both reservoirs. These findings are supported by Whittington *et al*. (2000) and Van Ginkel *et al*. (2001).

Cyanophytes occur most often in summer, during stratification conditions and are typical of enriched water bodies. It is well-documented that cyanophytes are able to dominate a phytoplankton population when nitrogen is the limiting nutrient and in warmer water temperatures (Bucka, 1989; Shapiro, 1990). During phases of low N:P ratios, cyanophytes have an advantage over other groups as they are able to fixate atmospheric nitrogen and will become more abundant (Smith, 1983; Pearl & Tucker, 1995). Cyanophyte biomass was of a higher magnitude and more frequently observed in production sites compared to reference sites. In production site 2, cyanophytes formed a major part of the biomass in September 2005, December 2005 and September 2006. The cyanophyte dominance during the summer months of 2005 agreed with a N-limitation situation in production site 2. In production site 1, cyanophytes were present from the onset of stratification in spring and remained in low abundance until dominating the association in April. Higher water temperatures (± 23 °C) and N-limiting conditions in production site 1 favoured the growth of cyanophytes.

4.5 Conclusions

While the success of cage fish farming is a function of water quality and the phytoplankton assemblage of the water resource. Poor management of nutrient input during fish farming will have detrimental effects on phytoplankton ecology and associated problems, such as oxygen depletion, pH fluctuations and increasing ammonia levels, can ultimately terminate fish farming.

The phytoplankton of production and reference sites comprised of species from six taxonomic classes, with Chlorophyta as the most diverse group. In general, species diversity across all the reservoirs was very low. However, production dams demonstrated a more evident decline of species diversity during the winter months. Low species richness and diversity in production dams was ascribed to single species domination, by species typical of highly eutrophic systems and represented by the genera *Ceratium, Microcystis* and *Anabaena*.

Seasonal abundance of phytoplankton was largely influenced by the onset and duration of stratification, the mixing regime and nutrient availability in the dams. With highest biomass concentrations in winter, the continuous mixing regime of the dams was central in providing favourable nutrient conditions to sustain high algal biomass. Both reference and production sites exhibited similar patterns of algal biomass in winter. However, the main impacts of cage aquaculture were in terms of magnitude and dominant species. Biomass peaks in production sites were twice more than in reference sites. The increase in biomass in production sites can be damaging to the fish farmer as dense phytoplankton densities can raise pH levels that can directly or indirectly through ammonia toxicity, affect fish growth and health. High phytoplankton biomass can also cause fluctuations in oxygen levels that can induce additional stress on rainbow trout. Species dominating the peaks in production dams included *Ceratium hirundinella, Anabaena* spp. and *Microcystis* spp. all of which are indicators of highly eutrophied systems. The dominance of assemblages in production sites by *Microcystis* spp. and *Anabaena* spp. towards the end of the production season, may raise concern for the fish farmer as these species are known to cause off-flavours. At the current production levels, the impact of cage aquaculture on the phytoplankton assemblages was evident from an increase in biomass and a higher occurrence of cyanophyte dominance.

CHAPTER 5

SEASONAL ABUNDANCE AND SPECIES COMPOSITION OF ZOOPLANKTON COMMUNITIES IN FARM DAMS USED FOR AQUACULTURE

5.1 Introduction

The formation of man made reservoirs (e.g. storage reservoirs, farm dams) has greatly increased the availability of habitats for aquatic biota. Investigations into zooplankton colonisation traits (composition, biomass, and size structure) have been restricted primarily to larger reservoirs and natural pans in South Africa, but data on zooplankton associations in smaller reservoirs and farm dams are lacking. Findings indicate that southern African continental waters are relatively low in species diversity and usually consist of between one and two calanoid copepods and two to six cladocerans (Hutchinson *et al*., 1932; Allanson *et al*., 1990). However, species diversity varies greatly between different water bodies, between geographical locations and over time (Hutchinson *et al*., 1932; Allanson *et al*., 1990).

5.1.1 Factors influencing species composition and abundance

Species composition and abundance of zooplankton populations are explained as a response to a combination of biotic and abiotic components. The effects of abiotic factors such as the circulation regime of the system (Hart & Hart, 2006), wind action, food availability (Lampert, 1988; Moss, 1998), turbidity gradients (Hart, 1986, 1990, 1991, 1999), water temperature (Hart & Rayner, 1994) and reservoir productivity (Bays & Crisman, 1983; Ostojic, 2000; Seda & Devetter, 2000) have been cited as key factors responsible for the patchy distribution of zooplankton. Succession is also dependent on interactions with other biotic components, namely phytoplankton, as a primary food source (bottom-up) and predation by zooplanktivorous fish species (top-down) (Jakobsen & Johnsen, 1987; Winder *et al*., 2003; Yoshida *et al*., 2003) as well as competition (Goulder 1979; Hart & Rayner, 1994). Vertical migration of zooplankton enables them to follow phytoplankton reserves and to reach optimal temperatures and dissolved oxygen conditions (Stich & Lampert, 1981; Gliwicz, 1986; Allanson *et al*., 1990). Zooplankton does not always follow the suggested diel vertical migration pattern and in some cases reversed vertical migration has been observed. This behaviour is speculatively related to the vertical distribution of phytoplankton in the water column (Fejes *et al*., 2003), competition, as well as extremely high alkaline conditions (Connel, 1978).

5.1.1.1 Turbidity

Zooplankton assemblages are highly susceptible to the level of sediment-related or inorganic turbidity. High suspended solid levels can have impacts on zooplankton species composition and therefore, community structure, by affecting their feeding biology. Firstly, the shading effects of high suspended solids inhibit the autotrophic photosynthetic action of phytoplankton, thereby reducing the primary food

resource of zooplankton. Secondly, high amounts of non-nutritive suspended solids affect the food-collecting abilities of different species (Allanson *et al*., 1990; Hart, 1992). Studies at Lake Le Roux, South Africa, have found that the contribution of *Daphnia* spp. towards the total zooplankton biomass decreased during the periods of high turbidity (Hart, 1986). The filter feeding nature of *Daphnia* is impaired by large amounts of fine suspended solids and a non-filter feeder would replace it. *Daphnia pulex* and *Daphnia longispina* are more adapted to clear-water conditions, whereas *Daphnia barbata* and *Daphnia gibba* are more adapted to turbid waters (Hart & Hart, 2006). Other zooplankton taxa such as copepods are more adapted to occupy turbid waters (Hart & Hart, 2006). The raptorial feeding behaviour of these taxa enables them to selectively capture their prey and they therefore have a competitive advantage during times of high turbidity (Moss, 1998). On the other hand, high turbidity levels also favour the growth of zooplankton, as it creates a visual barrier that reduces predation by zooplanktivorous fish species (Hart, 1986; Wetzel, 2001). Large bodied cladocerans are generally more vulnerable to predation by zooplanktivorous fish, although studies have indicated co-existence between large bodied species and zooplanktivorous fish (Hart, 1986).

5.1.1.2 Eutrophication

The trophic status of a reservoir is another factor that has a significant influence on the structure of zooplankton assemblages, as it affects the quality and quantity of phytoplankton biomass. An increase in the trophic status of a water body is associated with increased numbers and biomass of zooplankton as their primary food source becomes more available (Bays & Crisman, 1983; Pace, 1986; Canfield & Jones, 1996; Moss, 1998). Changes in zooplankton succession are also largely determined by their primary food resource. The phytoplankton of highly eutrophicated systems is generally dominated by species that are associated with toxin production and are often too large (> 40 µm) to be consumed by zooplankton. This often results in the disappearance or replacement within copepod and cladoceran communities (Nielsen, 1991; Matveev *et al*., 2000). Certain *Bosmina* spp. and *Brachionus* spp. are able to feed on colonial cyanobacteria during blooms (Fulton & Paerl, 1987). Cyclopoid copepods are more successful in eutrophic systems than calanoid copepods and this can be ascribed to their ability to utilise large particles as food (Pace, 1986). This enables them to consume colonial and filamentous cyanobacteria associated with elevated eutrophication. An increase in trophic status is also reflected in changes within the size-structure of zooplankton populations. Inhibition of filtering rates through filamentous and colonial cyanobacteria can cause large bodied organisms to be replaced by smaller forms, particularly small cladocerans, ciliated protozoa (< 20 µm), rotifers and copepod nauplii (Porter *et al*., 1979; Pace & Orcutt, 1981; Bays & Crisman, 1983). Small-bodied ciliates prey on bacteria and are more abundant in eutrophic systems that characteristically consist of higher bacterial numbers. Rotifers are able to consume a wider range of food particles (5 – 100 µm) because of their raptorial feeding habit. Therefore, they are able to reduce the biomass of phytoplankton even when larger species are dominant (Moss, 1998). In terms of protozoan abundance, eutrophic conditions are characterised by members of the Scuticociliatida, Oligotrichida, and Haptorida, whereas, oligotrophic lakes are dominated by Oligotrichida (Beaver & Crisman, 1982). Zooplankton are also currently used as indicators of environmental conditions in freshwater

ecosystems (Joska *et al*., 2005). Zooplankton, however, occur in a wide range of water quality conditions and their use as indicators of environmental conditions is therefore limited to extreme conditions. Their value lies in their high turnover rates that make them more ready to respond to changes in environmental conditions. Rotifers, especially members of the genera *Brachionus*, *Colurella*, *Trichocerca*, *Monostyla, Euchlania* and *Rotaria*, are most commonly used as indicators of highly eutrophic conditions (Gannon & Stemberger, 1978; Sládeček, 1983; Conde-Porcuna *et al*., 2002).

5.1.2 Seasonal succession of zooplankton communities

Zooplankton appeared to be most abundant during the warmer, summer months in stratifying systems, whereas maximum biomass were recorded during winter months in non-stratifying systems (Hart, 1985; Wetzel, 2001). Seasonal succession of zooplankton populations in southern hemisphere freshwater systems has been extensively reviewed by Hart (1985). He concluded that seasonal succession is primarily related to the availability and quality of the primary food source, temperature and water transparency. Despite different regulators, the studied dams showed a similar succession pattern. During spring and early summer the availability of edible green algae and rising temperatures stimulated growth of *Bosmina* spp. and *Daphnia* spp. (Hart, 1985; Allanson *et al*., 1990). Another incidence of early winter increase, following annual circulation was observed in the highly eutrophic Hartbeespoort Dam (Hart, 1985). The early summer increase in Lake Le Roux was ascribed to the increasing surface temperature as summer approached (Hart, 1985). Increased grazing pressure on small edible phytoplankton forms favoured the growth of large inedible species such as cyanophytes and dinophytes and in response to this the zooplankton populations declined.

The development of zooplankton assemblages differs among dams and is greatly affected by the age of the dam, morphometry, water quality, geological formations, meteorological conditions (wind & precipitation) and vegetation in and surrounding the water body. The main effect of rainfall is visible in the increased runoff and subsequent increased suspended solid content during the rainy seasons. Older dams were also found to have a higher incidence of larger species of copepods and cladocerans (Zimba *et al*., 2003). A marked difference has been found between vegetated and unvegetated reservoirs, with the former being rich in zooplankters and the latter being poor (Richard *et al*, 1985). Farm dams differ from lakes in terms of their hydrological aspects and how the ecosystems cope with changes. Water is artificially extracted for domestic, industrial and agricultural purposes which leads to fluctuations in water level and the retention time of the dams. Findings by Naselli-Flores and Barone (1994) showed that the dominance of cladocerans was dependent on fluctuations in water levels. These findings could have resulted from changes in the bottom-up (phytoplankton availability) and top-town (predation) forces in the water bodies. Fluctuations in water levels can also indirectly influence zooplankton communities by inhibiting or preventing the growth of vegetation that provides refuge against zooplanktivorous fish (Hart & Hart, 2006). Species diversity within rotifer populations and dominance were found to be closely related to retention time of dams (De Manuel & Armengol,

1993). Another consideration regarding these systems is the nature of the outflow, with particular reference to the depth of water extraction.

5.1.3 Implications of cage aquaculture

Man made dams in South Africa generally show high turbidity levels or are culturally eutrophied (Toerien *et al*., 1975; Grobler & Silberbauer, 1984). From the literature, it is clear that these two factors have a tremendous effect on the succession pattern of zooplankton populations. Additionally, the introduction of cage aquaculture into farm dams can have detrimental affects on the aquatic ecosystem. High stocking densities and feed inputs result in large quantities of suspended material entering the water column. Nutrients from fish feed and metabolic waste may account for increased eutrophication and higher suspended solid content in the water, ultimately affecting the colonisation characteristics (size structure, species composition, species abundance) of zooplankton.

The main objective of this chapter was to describe the impacts of cage aquaculture on species composition, and abundance (biomass) of the primary zooplankton groups, from November 2005 to November 2006. For the purpose of this chapter the author studied the distribution of zooplankton in two farm dams containing cage production of rainbow trout (*Oncorhynchus mykiss*) and in two farm dams without any fish farming.

5.2 Materials and methods

Four farm dams were selected for the purpose of the study of which two contained net cage culture of rainbow trout (*Oncorhynchus mykiss*). The remaining two sites, also referred to as reference sites, contained no aquaculture and had no previous history of aquaculture.

Macrozooplankton (cladoceran & copepoda) samples were collected on a fortnightly basis from November 2005 to November 2006 using a Schindler-Patalas plankton trap. Collection took place from a fixed station in the deepest part of each dam. Samples were fixated in a final concentration of 5 % formalin for subsequent counting and identification (Wetzel & Likens, 2000). Identification and measurements of individual species were made using a compound microscope with an ocular micrometer. Cell counts were performed under a stereomicroscope. Dry weight biomass was derived from length-weight relationships determined by Hall *et al*. (1970), Dumont *et al*. (1975), Culver *et al*. (1985) and Wetzel & Likens (2000).

Unconcentrated water samples for microzooplankton (rotifera & protozoa) analysis were collected with a water sampler. Samples were fixated with Lugol's acetic solution for preservation and dyeing of cells. In the laboratory, samples were assessed under an inverted microscope (Lund *et al*., 1958). Dry weight biomass of rotifera taxa was derived from Hall *et al*. (1970) and Foissner & Berger (1996). Biovolumes of protozoan taxa were calculated from appropriate geometric shapes and then converted to biomass (Beaver & Crisman, 1982).

Additional environmental variables were measured and included: water temperature (°C); water transparency (as secchi disk depth in meters); total suspended solids (mg/L); pH; dissolved oxygen (mg/L); inorganic nitrogen (NO_2–N + NO_3–N + NH_3-N) and total phosphorus (as P). Tables 2.3 and 2.4 provide a complete description of the analyses that were performed (Chapter 2).

5.3 Results

5.3.1 Species identification and composition

A total number of 22 zooplankton genera were identified in the various dams between November 2005 and November 2006 (Table 5.1). The number of species found in the production sites (x = 23) was slightly higher than the total number of species identified in the reference sites (x = 18). In terms of number of species, the protozoa were the most abundant (48.2 %) followed by the rotifera (22.2 %), the cladocera (18.5 %) and the copepods (11.1 %). The identification of copepod species presented difficulties and they were only classified according to calanoid or cyclopoid characteristics.

Table 5.1: Species of zooplankton identified in production site 1 (PR 1), production site 2 (PR 2), reference site 1 (RS 1) and reference site 2 (RS 2) from November 2005 to November 2006 (+ = present; - = absent).

	PS 1	PS 2	RS 1	RS 2
CLADOCERA				
Bosmina longirostris	+	+	+	+
Daphnia longispina	+	+	+	+
Daphnia pulex	+	+	+	+
Daphnia barbata	+	+	+	+
Daphnia spp.	+	+	+	+
COPEPODA				
Cyclopoida	+	+	+	+
Calanoida	+	+	+	+
Nauplii	+	+	+	+
ROTIFERA				
Brachionus urceus	-	-	-	+
Colurella spp.	-	-	+	-
Keratella cochlearis	+	+	-	+
Keratella quadrata	-	-	+	+
Keratella valga	+	-	-	-
Trichocerca spp.	-	+	-	-
PROTOZOA				
Acanthocystis spp.	+	-	+	+
Euplotes eurystomus	-	-	-	+
Halteria spp.	+	+	+	+

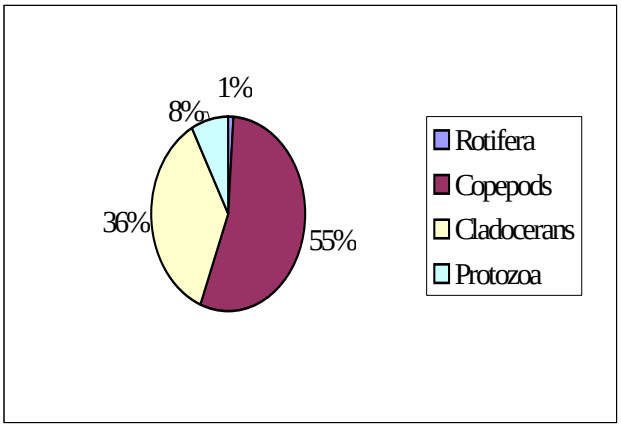
Holophrya spp.	+	-	-	-
Paramecium spp.	-	+	-	-
Prorodon spp.	-	-	+	-
Quadrulella globulosa	-	-	-	+
Strombidium viride	+	+	+	+
Tinema lineare	-	+	+	-
Tintinnopsis spp.	-	+	+	+
Urotricha spp.	+	-	+	-
Vorticella monilata	+	+	+	+

5.3.1.1 Cladocera

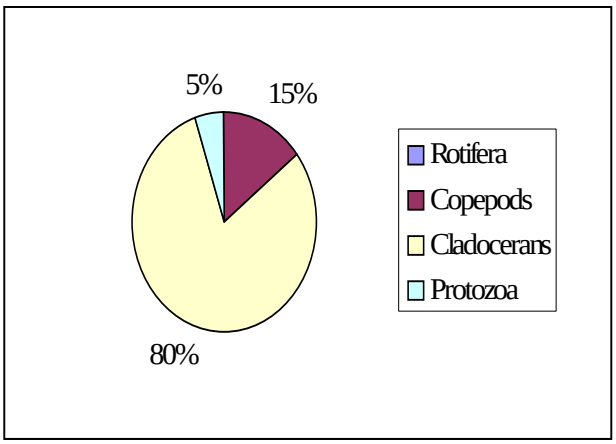
Cladocerans were found to be the most important contributors to the total biomass, even though they were only presented in small numbers. This can be explained by their large body size compared to the size of other zooplankton species. Cladocerans were ten times more abundant in production site 1 (x = 186 mg/L) than in reference site 1 (x = 17.2 mg/L) and three times more abundant in production site 2 (x = 104 mg/L) than in reference site 2 (x = 34.2 mg/L). The spatial distribution of cladocerans was characterised by an increase in numbers with an increase in sampling depth. In production site 1, however, numbers decreased towards the bottom waters. The most dominant cladoceran species included representatives from the genera *Daphnia (Daphnia duplex, Daphnia barbata; Daphnia longispina)* and *Bosmina (Bosmina longirostris).* Species from these genera were present in both production and reference sites (Figure 5.1).

5.3.1.2 Copepoda

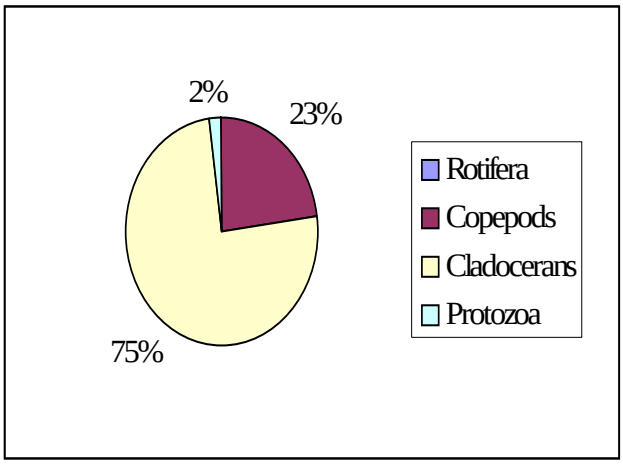
As with the cladocerans, copepods were more abundant at production sites than at reference sites. Copepod populations in production site 1 were 1.5 times higher than in reference site 1, while the populations in production site 2 were nine times more numerous than in reference site 2 (Figure 5.1). Both calanoid and cyclopoid were recorded in all the study sites. Further observations of the ratio between calanoid and cyclopoid species indicated that the production sites were primarily colonised by cyclopoid species. Although calanoid and cyclopoid copepods were evenly abundant in reference sites, the growth of calanoid species was slightly favoured.



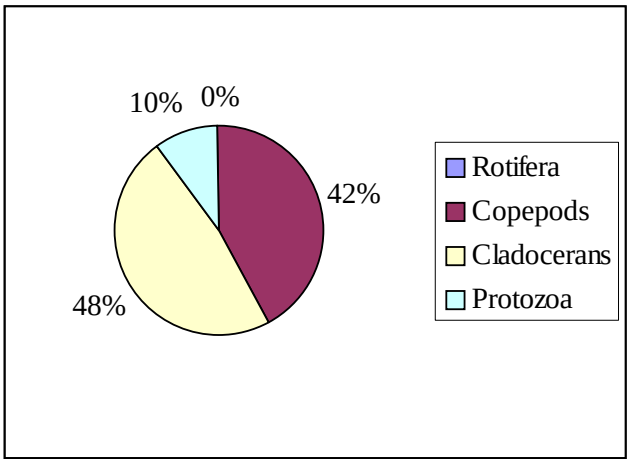
(a) Reference site 1



(b) Production site 1



(c) Reference site 2



(d) Production site 2

MAAK LADSCAPE BL NB PG NUMBERS

5.3.1.3 Rotifera

Rotifera were numerically the most abundant group, but due to the small size of the genera it did not make a marked difference in total biomass (Figure 5.1). Rotifera species were regularly detected in three of the sites, especially the cosmopolitan species *Keratella cochlearis.* However, in production site 2, rotifers only occurred occasionally and in very low numbers. Rotifera representatives of the genera *Brachionus, Colurella* and *Keratella* were only detected in reference sites, while *Keratella valga* and *Trichocerca* were only found in production sites.

5.3.1.4 Protozoa

Protozoa were the most diverse group in all the dams with species from the genera *Strombidium, Halteria, Tintinnopsis, Vorticella, Prorodon, Paramecium, Holophrya, Acanthocystis, Euplotes, Quadrulella, Tinema* and *Urotricha*. For this part of the study, protists consisted only of protozoa, as heterotrophic flagellates are discussed as phytoplankton (Chapter 4). Genera that were present in all the dams included *Strombidium*, *Halteria* and *Vorticella (Vorticella monilata)*.

5.3.2 Seasonal abundance

The total biomass measured in production sites was significantly higher than biomass values from the reference sites ($p < 0.009$). Biomass values in production sites fluctuated between 0.006 mg/L and 2.4 mg/L, while the total zooplankton biomass in reference sites ranged between 0.0002 mg/L and 0.3 mg/L.

5.3.2.1 Production site 1 (Nietvoorbij Dam)

Production site 1 had the highest biomass throughout the entire study period with biomass ranging between 0.3 mg/L and 2.4 mg/L. Figure 5.2 indicates two distinct peaks of which one occurred during April 2006 followed by a winter peak in June 2006. At the beginning of the study the population was dominated by cladocerans. This dominance remained until the peak in April 2006 during which time cladocerans dominated the biomass. The most significant contributor was the large bodied cladoceran, *Daphnia* spp. During this phase a small number of the rotifer, *Keratella cochlearis,* were also identified, but due to their small size, there was no marked contribution to total biomass. In April, the phytoplankton biomass of production site 1 showed a peak with the most important contributors being dinophytes and cyanophytes. Despite the inedible nature of these species, cladocerans are able to utilise them, either directly or indirectly as algal debris (Schoenberg & Carlson, 1984; Fulton & Paerl, 1987). During May, a decline in cladoceran numbers was observed and was followed by a winter peak in June. The highest zooplankton biomass at all the study sites was recorded during this peak, with values as high as 2.4 mg/L. Regarding the cladoceran *Daphnia* spp., dominated the population, however, numerically the rotifer, *Keratella cochlearis* was most abundant. This peak was also characterised by contributions from copepod nauplii and *Keratella cochlearis*.

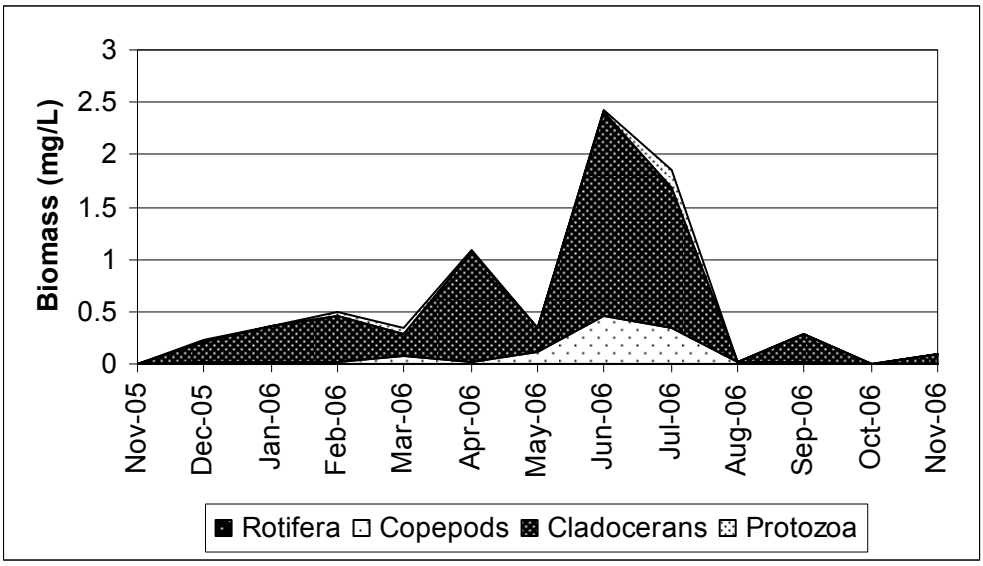


Figure 5.2: **Seasonal distribution of zooplankton biomass in production site 1 between November 2005 and November 2006**

5.3.2.2 Reference site 1 (Poplar Dam)

The seasonal distribution of zooplankton biomass in reference site 1 showed three distinct peaks. One occurred in April 2006, followed by the second peak in June 2006 and a third in September 2006 (Figure 5.3). During the first peak, members of cladocera dominated the zooplankton population. The most important contributor to cladoceran biomass was *Bosmina longirostris*, with a total biomass of 0.36 mg/L. The occurrence of *Bosmina longirostris* increased with depth and a maximum biomass was measured in samples of near bottom water. The peak in April was followed by a sudden decline in population biomass and during that phase protozoa dominated the system. Zooplankton biomass gradually increased throughout the winter months until a second peak was reached in June 2006. In terms of biomass, members of the cladocera (0.25 mg/L) and copepoda (0.16 mg/L) dominated this peak. The bulk of copepoda were calanoid copepods and copepod nauplii. The winter zooplankton peak corresponded with a high phytoplankton biomass which was dominated by *Peridinium* spp. and a small number of *Cryptomonas erosa,* which are grazed upon by *Daphnia* spp. (Bergquist *et al*., 1985; Edgar & Green, 1994). The abundance of both cladoceran and calanoid copepods was found to increase with depth. Due to the small size, rotifera and protozoa did not contribute significantly to the overall biomass. However, members of the group rotifera numerically dominated the peak in September 2006 with the most abundant species being *Keratella cochlearis*. Zooplankton biomass was lowest in the summer months, corresponding to the low phytoplankton biomass. Members of the group protozoa were perennial species in the system and were present at all sampling events. Although they did not contribute significantly to total biomass, numerically the most important genera were *Acanthocystis*, *Strombidium* (e.g. *Strombidium viride*) and *Halteria.*

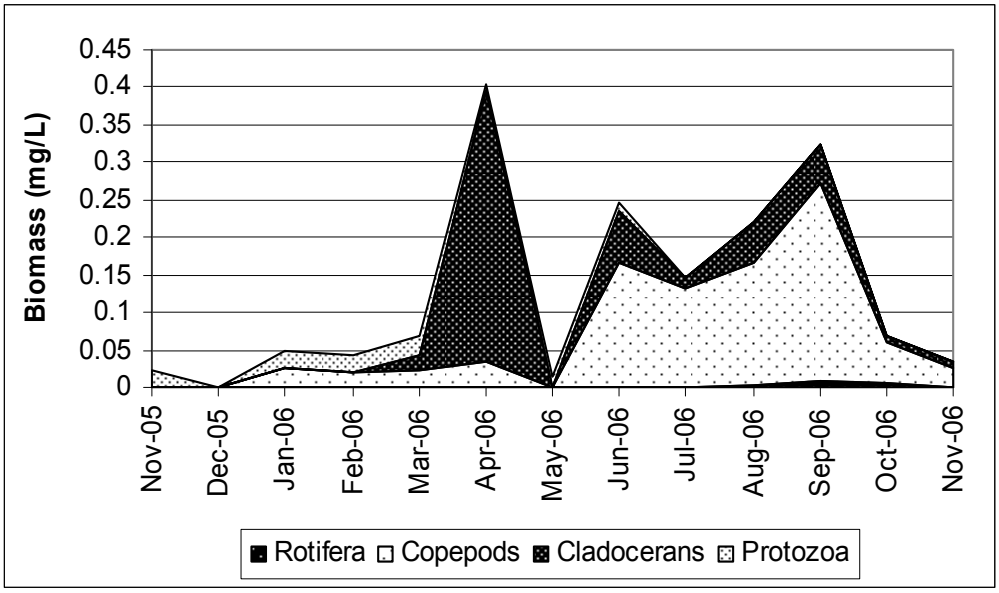


Figure 5.3: **Seasonal distribution of zooplankton biomass in reference site 1 between November 2005 and November 2006**

5.3.2.3 Production site 2 (John Smith Dam)

The zooplankton biomass measured in production site 2 was about five times higher than that recorded in the two reference sites. Figure 5.4 clearly indicates that the zooplankton biomass peaked on three occasions. The summer peak was dominated by the cladocerans (1.8 mg/L) of which the large bodied *Daphnia* spp. were the most important (biomass and cell numbers). The number of cladocerans increased in samples from deeper waters. Following a decline in January, biomass increased gradually until a second peak in April 2006. Copepods were the major group in this peak and contributed 98 % of the total biomass. The bulk of the copepods identified belonged to the cyclopoid. Although in lower abundance, the population remained dominated by copepods until May, when the number of the small-bodied cladoceran *Bosmina longirostris* started to increase. This was followed by the development of the winter peak in June 2006. In terms of biomass, this peak was co-dominated by copepods (cyclopoida & nauplii) and cladocerans (*Bosmina longirostris* & *Daphnia* spp.). Cyclopoid copepods were numerically the most abundant during this peak. During this period the phytoplankton biomass peaked due to the presence of cryptophytes (Chapter 4). The dominant phytoplankton species during that peak was *Cryptomonas erosa*, which has been associated with high reproduction rates in *Daphnia* (Infante & Litt, 1985). From July 2006 onwards, the peak in zooplankton biomass declined until October. The rotifera group was rarely present in this dam.

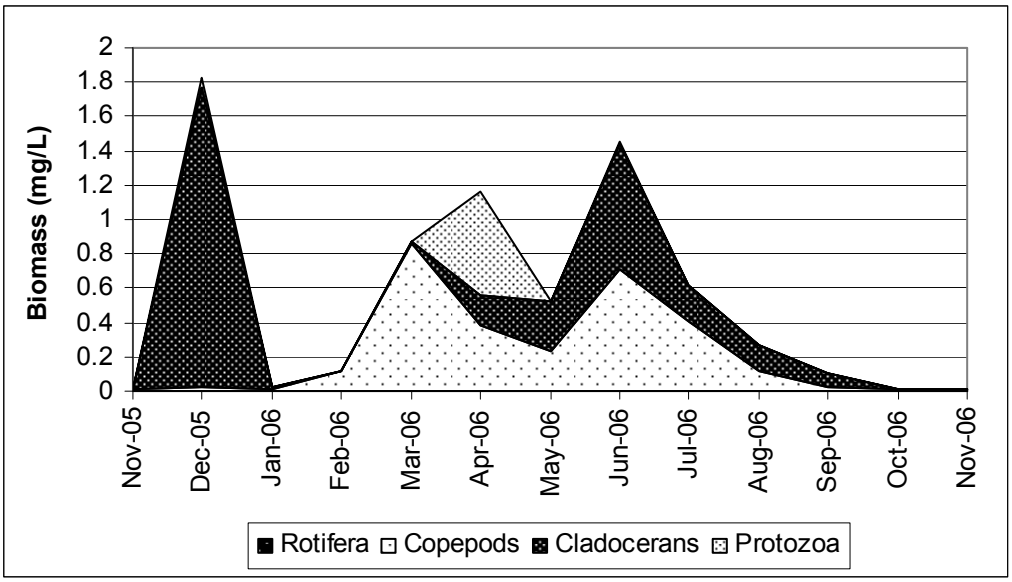


Figure 5.4: **Seasonal distribution of zooplankton biomass in production site 2 between November 2005 and November 2006**

5.3.2.2 Reference site 2 (Garden Dam)

In reference site 2 the zooplankton biomass fluctuated between 0.007 mg/L and 0.36 mg/L (Figure 5.5). Copepods were the main zooplankton group in summer, making up 72 % of the total biomass. Almost all identified copepods were in nauplii stages with a few calanoida individuals. Protozoa also reached its highest biomass during this phase with *Euplotes eurystomus* as a major contributor, partly due to its large size.

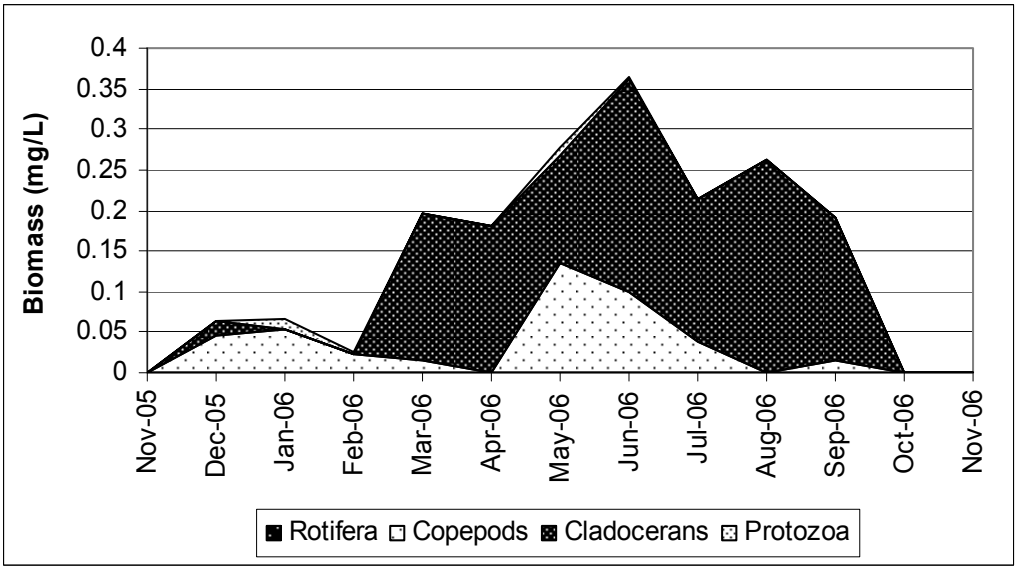


Figure 5.5: **Seasonal distribution of zooplankton biomass in reference site 2 between November 2005 and November 2006**

Following the small peak in December 2005, the zooplankton biomass declined and only reappeared in March 2006. The winter peak was higher in biomass compared to the summer biomass values, with a biomass reaching 0.36 mg/L. This peak was co-dominated by calanoid copepods and species of the cladoceran genus, *Daphnia.* Both cladocerans and copepods increased in biomass with an increase in depth. Numerically both, copepod nauplii and the rotifera *Keratella cochlearis*, dominated the peak. Despite the numerical abundance, the presence of the large bodied *Daphnia* favoured the total cladoceran contribution to total biomass. The phytoplankton biomass during the winter months consisted largely of edible green algal genera (*Scenedesmus, Crucigenia,* and *Kirchneriella*) that are readily consumed by cladocerans. The winter peak gradually declined until it disappeared during October 2006. The rotifera were absent for most of the sampling period, except for high numbers during the winter peak. *Keratella cochlearis* and *Keratella quadrata* were the main rotifera species recorded.

5.4 Discussion

The dominant zooplankters in all production sites were the cladocerans, most notably *Daphnia* spp., while reference sites were dominated by copepods. The copepod population of production sites was dominated by cyclopoid copepods, which are indicative of the eutrophic status of the dams (Gannon & Stemberger, 1978). Calanoid copepods were slightly more abundant in reference sites and are indicative of a less eutrophic system (Gannon & Stemberger, 1978). Although both cladocerans and copepods were present throughout the study period, their abundance and dominance fluctuated seasonally. Jarvis (1986) observed that zooplankton of a hypertrophic lake (Hartbeespoort Dam) in South Africa reached zooplankton biomass peaks in spring. Following the spring peak he observed a mid-summer decline in the *Daphnia* population (Jarvis, 1986). In the dams of the present study, the zooplankton biomass followed a similar seasonal pattern as stated by Jarvis (1986). The biomass peaks in reference sites occurred towards the end of winter and in spring, followed by a sudden decline in summer. In production sites, biomass peaks occurred earlier, in mid winter, and also declined in summer. The stenotherm cladoceran *Daphnia* was found to dominate the population during winter and spring months but decreased in summer. The zooplankton groups, rotifera and protozoa, were poorly represented in all the study sites. The exclusion of this group can be due to competition for a limited food source. A study by Gilbert (1988) documented that rotifers are often scarce in the presence of *Daphnia* spp. The annual peak in winter corresponded to the annual peak in phytoplankton biomass, indicating the importance of available food resources. The availability of sufficient food resources was the result of hydrological events such the continuous circulation in winter, which supported a high phytoplankton biomass (Chapter 4). The incidence of high zooplankton biomass in non-stratifying systems during winter months has been well documented in various reservoirs of South Africa (Hart, 1985).

Physical factors strongly influenced seasonality, but biological interactions (influence of food availability and quality) were undoubtedly observed. In production site 2, the peak in copepod biomass during March 2006 coincided with a sudden bloom of cyanophytes and an increase in the edible green alga, *Cosmarium* spp. The winter peak in zooplankton biomass also correlated with the high abundance of *Cryptomonas erosa* which were consumed by *Daphnia* spp. (Infante & Litt, 1985). In production site 1, the interactions between phytoplankton and zooplankton communities was also apparent. The peak in zooplankton biomass during May 2006, corresponded to a breakdown in a peak in the algal biomass. The phytoplankton biomass was dominated by cyanophytes and it has been documented that *Daphnia* spp. and *Bosmina longirostris* are able to feed indirectly on algal debris (Schoenberg & Carlson, 1984; Fulton & Paerl, 1987). However, during the winter months, production site 1 was dominated by *Ceratium hirundinella* which were too big to be consumed by cladocerans. Jarvis (1986) also found that lowest grazing rates of *Daphnia* only occurred when the reigning phytoplankton population was dominated by colonial and filamentous cyanophytes (> 60 µm). Thus, indicating that in highly eutrophic conditions it is unlikely that large filter feeders such as *Daphnia* will be able to hinder or limit the growth of cyanobacteria (Jarvis, 1986). A small number of *Cryptomonas*

erosa cells were present, but hardly enough to sustain the high zooplankton biomass observed. In reference site 2, the highest abundance of phytoplankton was found during winter that consisted largely of edible chlorophytes (*Scenedesmus, Crucigenia,* and *Kirchneriella*). During the same period, the zooplankton biomass reached its annual peak. Zooplankton peaks in reference site 1 correlated well with phytoplankton biomass during winter months. The most notable peak in zooplankton biomass was seen in July, with a simultaneous increase in phytoplankton biomass. The phytoplankton biomass was dominated by *Peridinium* spp. and a small amount of *Cryptomonas erosa* which are grazed upon by *Daphnia* spp. (Bergquist *et al*., 1985; Edgar & Green, 1994). Zooplankton biomass values correlated closely with the total phytoplankton biomass in the respective dams, thus indicating that zooplankton biomass is influenced by the availability and quality of the food resource (Moss, 1998). The high biomass measured during winter months also coincided with high turbidity levels derived from increased runoff (Chapter 3). Elevated turbidity levels can be beneficial or detrimental to zooplankters. The decrease in transparency reduces the vision of zooplanktivores fish, thereby minimising predation rates on zooplankton populations, however, some species are impacted by high suspended solids (Hart & Hart, 2006).

Although the number of species between the production and reference sites was almost similar, the most noticeable difference was detected between the total zooplankton biomass of the different dams. The two production sites had a higher total biomass as opposed to the biomass values of the reference sites. The observed trend is supported by the findings of Guo & Li (2003). The higher phytoplankton biomass in production sites corresponded well with the higher zooplankton biomass at the same time. Both reference and production sites were subjected to the same hydrological events, such as the monomictic mixing regime during the fish farming season, thus one would expect that the higher zooplankton biomass would have derived from a food resource that was impacted by cage aquaculture. The increased phytoplankton biomass in the production sites due to the addition of nutrients from cage aquaculture and hydrological events, therefore resulted in an increased zooplankton biomass.

5.5 Conclusions

The results of the study indicated that, at present production levels, cage aquaculture had short-term impacts on the zooplankton populations of the studied dams. This included the marked increase in zooplankton biomass in dams hosting cage aquaculture, as opposed to reference sites. The nutrient loading from the current level of production resulted in a distinct increase in the quantity of food resources. The long-term effects of increased trout production are unknown and are difficult to predict because of the complexity of food web interactions. However, long-term impacts of cage aquaculture can result in a shift from large- to small-bodied zooplankton species, as the increase in trophic status will favour phytoplankton species that are unedible to large-bodied copepods and cladocerans. Additionally, larger-bodied zooplankton species would not be able to delay or limit the development of cyanophyte species, that in turn can be detrimental to any future of the fish farming enterprise.

CHAPTER 6

EVALUATION OF THE SUITABILITY OF WATER FROM AQUACULTURE ACTIVITIES FOR IRRIGATION EQUIPMENT

6.1 Introduction

South Africa is a dry country with an annual precipitation (< 500 mm) well below the world average (Davies & Day, 1998; Prinsloo *et al*., 2000). The seasonal distribution of rainfall forces farmers in the Western Cape to rely heavily on irrigation during dry summer months. Freshwater resources are becoming scarce while demands from industrial, domestic and agriculture water users are increasing. Currently, the annual water usage in South Africa is estimated at around 22 400 million m^3, of which the agricultural sector consumes 60% for irrigation applications (Van der Merwe, 2001). As irrigation practices are a major consumer, the need to adapt and develop ways to use water more sparingly has been recognised. This lead to the development of low-volume water irrigation systems (micro-irrigation) in the 1970's, and today approximately 12 % (140 000 ha) of the agricultural land in the country is under micro-irrigation. The remaining irrigation water users are still making use of conventional sprinkler and flood irrigation methods (DWAF, 1996a).

6.1.1 Micro-irrigation

Micro-irrigation is the regular application of water at small flow rates directly to the root zone of the crop under irrigation (Koegelenberg *et al*., 2002). The advantage of the technology lies in the small volume of water that is required, as well as the direct application to the root zone. Direct application to the root zone minimises moisture loss to evaporation, making it a viable alternative in dry and windy areas or where rainfall and water supply are limited. Despite the advantages, a major drawback of micro-irrigation systems is vulnerability to problems arising from poor water quality. The small size of the emitters makes the system more susceptible to clogging than other conventional systems. As a preventative measure, various treatments have been developed to ensure optimal water quality. Treatments include settling tanks, centrifugal separators, disc and sand media filters, screens and chemical treatments with chloride or acid (Phillips, 1993).

6.1.2 Clogging, scaling and corrosion

The most important problems arising from poor water quality include not only the clogging of emitters, but also scaling and corrosion (Nakayama & Bucks, 1991; Koegelenberg *et al*., 2002). Clogging is caused by contaminants that are categorised as physical components (suspended solids), biological components (aquatic plants, bacterial growth) and end products of chemical precipitation (ferric oxide). All these contaminants occur naturally in aquatic ecosystems and the contribution to clogging will depend on their relative abundance. Higher loads of suspended solids and algal will increase filtration capacities that will consequently require more frequent back flushing and cleaning of filters. This will

increase maintenance and operational costs. Abrasive action by high concentrations of suspended solids can also cause rapid deterioration and early replacement of irrigation equipment (Capra & Scicolone, 1998). Additional small particles (2 – 50 µm) that are not removed by the filtration process can disable irrigation systems by clogging emitters (Koegelenberg *et al*., 2002). Target water quality ranges propose a suspended solid content of less than 50 mg/L (DWAF, 1996a).

Biological components contributing to emitter clogging include aquatic organisms such as zooplankton, insects, macrophytes, algae and bacteria. Algal growth is found naturally in water bodies and can appear as single cells, as groups of cells (colonies) and as branched or unbranched filaments. Algal growth is strongly influenced by the availability of nutrients (nitrogen & phosphorous) in the water.

Chemical clogging is caused by chemical interactions between dissolved ions to form precipitates in water. The most important dissolved ions to form precipitates in irrigation water are calcium and magnesium carbonates, as well as iron and magnesium. These elements occur naturally in freshwater and concentrations of these elements are derived from geological processes and soil characteristics of the surrounding catchment. Encrustation or scaling is the conversion of calcium bicarbonate to the precipitate calcium carbonate (calcite), carbon dioxide and water. The drivers behind this reaction are heat energy (via sun), pH of the water and a sudden drop in pressure when water is emitted at the drippers. Apart from calcium and magnesium concentrations, pH is the most important catalyst of precipitation reactions and pH values exceeding 7.5 are likely to cause precipitation of calcium and magnesium carbonates (Koegelenberg *et al*., 2002). Iron is another element that is known to form precipitates. Ferrous iron is the reduced form and is soluble in water. Upon oxygenation, ferrous iron is converted to insoluble ferric oxide and can cause emitter clogging. Iron concentrations of less than 2 mg/L are recommended but will still have minor clogging effects during irrigation (Bucks *et al*., 1979; DWAF, 1996a; Koegelenberg *et al*., 2002). Water containing low concentrations of calcium and magnesium are known to be corrosive or aggressive. Corrosive water attacks metal components and will result in premature replacement of irrigation equipment. The corrosiveness of a water resource is heavily dependent on the reigning pH values at the time. A low pH and low hardness increase the tendency of water to act corrosively towards metal and concrete structures (DWAF, 1996a).

6.1.3 Implications of cage aquaculture

The introduction of cage culture into irrigation dams is a way to optimise the use of existing water bodies. However, the nature of cage culture can affect water quality in a way that may be harmful to irrigation equipment. Cage culture introduces large amounts of nutrient rich particulate matter to the water, thereby adding to the suspended solid and nutrient concentrations of the water. Excessive nutrients will in turn stimulate the growth of phytoplankton. Increased suspended solid concentrations and algal biomass can lead to severe clogging problems in irrigation components. Phytoplankton biomass is one of the major components affecting the pH of a water body (Talling, 1976, Wurtz &

Durborow, 1992). As mentioned earlier, pH is one of the key players in determining the water's tendency to be corrosive or prone to scale formation. During photosynthesis, phytoplankton consumes carbon dioxide causing the pH equilibrium to shift towards more alkaline conditions and scale formation can become a possibility. As the population crashes, bacterial decomposition will take place and oxygen will become depleted (Boyd *et al*., 1975; Boyd *et al*., 1978). This will shift the pH equilibrium towards a more acidic condition that will raise the water's potential to be corrosive. Organic matter derived from cage culture, following a phytoplankton population decline will sink down and accumulate in the sediment. Decomposition will deplete hypolimnetic oxygen reserves and anoxic conditions will develop (Boyd, 1995). It is well known that anoxic conditions in bottom waters favour the release of iron and magnesium that are bound in the sediment (Mortimer, 1941). As water is exposed to oxygen during irrigation, iron and manganese can precipitate and increase the risk of emitter clogging.

This chapter addresses the suitability of water from cage culture on the performance of irrigation equipment. The focus is on water quality parameters that are responsible for clogging, scale formation and corrosion, which may have been influenced by fish farming. Water quality characteristics from production and reference sites were compared to target water quality values proposed in the literature (Bucks *et al*., 1979; DWAF, 1996a; Koegelenberg *et al*., 2002).

6.2 Materials and methods

6.2.1 Irrigation regime at study sites

In the Western Cape, the irrigation season start towards the end of October for two or three hours per day, around noon. As the heat becomes more intense, irrigation duration is slowly increased and can reach up to 24 hours per day that continues until March/April (various farmers, pers. comm.). Irrigation terminates after harvesting or when winter rains set in, usually during April. Irrigation on Western Cape farms consists of water extraction from the resource, filtration and distribution to the fields. Water is usually extracted at a fixed level of one or two meters below the water surface. This extraction level stays consistent irrelevant of changes in the water level of the dam, although in some dams bottom water is used. After extraction, water is pumped to a pump house where a two step filtration process takes place to remove suspended material. Primary filtration includes water passing through either a sand filter or a ring filter. Secondary filtration consists primarily of ring filters and is often distributed at various points between the vineyard blocks. After the secondary filtration process, water is then directed to the lateral pipes for delivery. During the filtration process, suspended material gets caught in the filter material (sand grains, filter discs) and is removed by back flushing of the system. Back flushing involves the forced movement of clean water through the filter material, in the opposite direction as when filtration takes place. Back flushing frequency differs for each system, depending on the quality of the resource water. Back flushing of sand filters is carried out four times a day, whereas ring filters are cleaned after each irrigation interval (various farmers, pers. comm.).

6.2.2 Sample collection and analysis

Water samples were collected to determine factors that may have an effect on the performance of irrigation equipment. Sampling was undertaken from a deep-water station and samples were also taken at surface and near bottom (> 6 m) depths (See Chapter 2). Duplicate samples were preserved with mercury chloride ($HgCl_2$) and sent to the Department of Water Affairs and Forestry for the determination of inorganic constituents. Whole water samples, for phytoplankton analyses, were routinely taken between November 2005 and March 2006. Samples were obtained by opening the ring filters prior to cleaning and scraping the surface of the individual rings within the ring filters. Samples were immediately fixated with Lugol's solution (Lind, 1979; Cole, 1994). On two occasions, additional phytoplankton samples were collected from the inside of the ring filters to determine if species responsible for clogging were related to their abundance at the time of sampling.

Water samples were analysed in order to determine the most important factors affecting clogging, encrustation and corrosion of irrigation systems. Water analyses focussed on variables that could have been influenced by the presence of fish farming, for example: pH, suspended solids (mg/L), alkalinity (mg/L), dissolved oxygen (mg/L), nutrients and dissolved solids (mg/L). Water temperatures (°C) and dissolved oxygen (mg/L) were measured using an Oxyguard MKIII oxygen meter (OxyGuard

International). Dissolved solids were measured by means of a Hach CO 150 conductivity meter and pH values were determined with a Hanna pH 211 microprocessor. Nutrients, including inorganic nitrogen (ammonia, nitrate, nitrite) and total phosphorous, were all determined using a Hach colorimeter (See Chapter 2 for complete methodology of analyses). Inorganic constituents were determined at the Department of Water Affairs and Forestry, Pretoria. The suitability of the water resource was evaluated according to the water classification system compiled by Bucks *et al*. (1979) (Appendix I, Table 1) and the South African Water Quality Guidelines for irrigation water (DWAF, 1996a). The Langelier Saturation Index (LSI), Ryznar Index (RI) and Aggressiveness Index (AI) were used to calculate the likelihood of the resource water to calcium carbonate scaling and corrosion at the time of analysis. The Langelier Saturation Index is based on the calcium concentration (mg/L), alkalinity (mg/L as $CaCO_3$), pH and temperature (°C). The Ryznar Index was developed to distinguish between different water sources that have the same values as the Langelier Saturation Index, but more emphasis is placed on pH values. The Aggressiveness Index is used to identify the corrosion potential of water in an asbestos pipe. All three indices are usually determined as one value but are insufficient to draw any conclusions. Interpretation of indices results were performed following the classification proposed in the South African Water Quality guidelines for irrigation water (DWAF, 1996a; Koegelenberg *et al*., 2002). See Appendix I for formula and further information on the indices.

Phytoplankton analysis of the water resource and from the ring filters, was carried out using the inverted microscope technique (Lund *et al*., 1958). The high pressure at which the water flows through the filter system distorted the majority of phytoplankton species. Species that were intact were identified to genus level. Qualitative results from ring filter samples were then compared to abundance results to establish if species responsible for clogging were in fact related to abundance, and if that coincided with the irrigation season.

6.3 Results

Sample analyses focussed on elements that can cause clogging, scaling and corrosion and that could have been influenced by aquaculture activities. Table 6.1 indicates the mean values of water quality parameters that are likely to cause clogging determined during the irrigation season of 2005/2006.

Table 6.1: Mean values of surface water quality of both production and reference sites during the irrigation season from November 2005 and March 2006. Ranges indicated in brackets. PS 1 = production site 1; PS 2 = production site 2; RS 1 = reference site 1; RS 2 = reference site 2

Site		pH	TSS mg/L	TDS mg/L	DIN mg/L
PS 1	*x* *n* =11	8.3 (7.7-8.9)	6 (3-9)	206 (195–235)	0.13 (0.06 – 0.3)
RS 1	*x* *n* =11	7.7 (7.4-8.7)	2 (1-4)	61 (58-74)	0.23 (0.1 – 0.3)
PS 2	*x* *n* =11	7.7 (6.7-9.7)	12 (7-19)	61 (57–74)	0.26 (0.1 – 0.3)
RS 2	*x* *n* =11	8.0 (7.4–8.6)	4 (2-6)	72 (66–77)	0.13 (0.07 – 0.2)

Compared to water quality classifications by Bucks *et al*. (1979) (Appendix I, Table 1) the total suspended solid content of all the dams falls below the proposed 50 mg/L and therefore posed only a minor potential for clogging. Average summer and winter TSS values showed that winter months experience higher overall TSS values which can partly be caused by increased runoff during rainfall events and aquaculture in the case of production sites (Chapter 3). From Table 6.1 it is clear that the TDS concentrations (< 500 mg/L) measured in both the production and reference sites, could only have caused a minor clogging effect on drippers and filters. The mean inorganic nitrogen content measured in all the dams was below the target water quality range of less than 0.5 mg/L inorganic nitrogen (DWAF, 1996a; DWAF, 1996b). The average pH of surface water in all the dams, except for production site 1, was within the target water quality range (Table 6.1). According to the classification from Bucks *et al*. (1979) moderate problems with clogging might occur with pH values between 7 and 8.

Algal identification of samples from inside ring filters is presented in Table 6.2. Primary species were from the genera *Ceratium, Anabaena, Dinobryon, Tribonema* and *Fragilaria*. The contribution of phytoplankton to the clogging of filters depended largely on the abundance of potential clogging-causing species in the water resource. Species abundance was compared to their seasonal abundance in order to determine which species contributed significantly to clogging during periods of irrigation. *Ceratium hirundinella* was most commonly found during the winter months (June, July,

August). The most prominent heterokontophytes, *Dinobryon divergens* and *Tribonema* spp. were also present during the winter months but only occurred in very low numbers. The cyanophytes, especially *Anabaena circinalis*, were highly abundant from November, with peak densities from December 2005 to March 2006. The occurrence of this group concurs with months during which irrigation took place.

Table 6.2: Phytoplankton species responsible for physical clogging of filters at production and reference sites

Division	Genus	Size (µm)
Dinophyta	*Ceratium hirundinella*	180 (single cells)
Cyanophyta	*Anabaena circinalis*	> 300 (filaments)
	Anabaena spp.	> 300 (filaments)
Heterokontophyta	*Dinobryon divergens*	> 200 (colony)
	Tribonema spp.	> 1700 (filaments)
	Fragilaria spp.	80 – 400 (single cells)

Table 6.3 summarises the inorganic constituents that can contribute to clogging of irrigation components.

Table 6.3: Inorganic constituents of production site 1 (PS 1), reference site 1 (RS 1), production site 2 (PS 2) and reference site 2 (RS 2) and their clogging potential

Site	Date	Fe mg/L	Mg mg/L	Ca mg/L	Mn mg/L
PS 1	Dec 2005	0.04	11	20	0.002
	Apr 2006	*	12	21	*
	May 2006	*	13	22	*
	Jul 2006	0.2	11	21	0.001
RS 1	Dec 2005	0.05	3	4	0.002
	Apr 2006	*	4	4	*
	May 2006	*	4	4	*
	Jul 2006	0.06	3	5	0.022
PS 2	Dec 2005	0.021	3	5	0.001
	Apr 2006	*	4	5	*
	May 2006	*	5	11	*
	Jul 2006	0.107	3	7	0.002
RS 2	Dec 2005	0.01	4	6	0.001
	Apr 2006	*	5	6	*
	May 2006	*	3	4	*
	Jul 2006	0.02	3	5	0.006

The current available water classification standard does not include information regarding target water quality ranges for calcium (Ca) and magnesium (Mg) precipitation. Current literature however, stated that Ca and Mg precipitation occurs if concentrations exceed 50 mg/L (Koegelenberg *et al*., 2002). The calcium and magnesium content of production site 1 was generally higher (20 – 22 mg/L) than all the other sites, which can be ascribed to the nature of one of its inflows. Calcium and magnesium concentrations measured in the remaining dams were low and ranged between 3 – 11 mg/L. Calcium and magnesium levels in all the dams were below the proposed 50 mg/L and therefore had minor clogging effects.Manganese (Mn) concentrations remained low (< 0.1 mg/L) in all the dams and only posed a minor clogging hazard. Both reference sites presented iron (Fe) concentrations well below the 0.2 mg/L, which may cause minor problems with clogging. Production site 1, however, had iron levels of 0.04 mg/L during December 2005, but a sample taken in July 2006 measured iron levels of 0.2 mg/L. The latter iron concentrations can lead to precipitation of ferric oxide that can cause blockages. The iron levels of production site 2 showed the same pattern as production site 1. Initial iron concentrations during December 2005 were measured at 0.021 mg/L and during July 2006 a concentration of 0.107 mg/L was measured.

The likelihood of irrigation water causing corrosion or scale formation is largely dependent on the pH value and hardness of a water sample. Table 6.4 indicates the potential of the surface water from the respective production sites to scaling and corrosion. Calculation of the Langelier Saturation Index showed that the surface water of production site 1 was neither prone to scaling or corrossion (balanced), with a possibility of pitting corrosion.

Despite the high pH values measured in December 2005 and May 2006, production site 1 remained balanced. The naturally low hardness combined with the low Ca and Mg content of the water, inhibited scale formation. In July 2006 the water of production dam 1 was only slightly corrosive. The Ryznar Index supported the low corrosive nature of production site 1 with index values calculated between 7 and 9. High pH values in December 2005 and May 2006 promoted a non-aggressive nature (AI > 12.0) to the water of production site 1. The water of reference site 1 also had a low corrosive nature. The low pH (6.4) measured during the December sampling event encouraged a highly corrosive outcome (LSI > -2.0) with a very high aggressiveness (AI > 12.0) towards irrigation structures. This was strongly supported by the Ryznar Index that indicated the corrosion level to be intolerable (RI > 9.0). From April 2006 the situation improved, but the water remained slightly corrosive and moderately aggressive. The improvement was due to the slight increase in actual pH measured. The surface water of production site 2 was of a corrosive character and the condition seemed to worsen during sampling events that followed (Table 6.4). The Langelier Index in December described the water of production site 2 as balanced. During the following sampling event, in April and May, it was slightly more corrosive.

Table 6.4: Potential of surface water from production site 1 (PS 1), reference site 1 (RS 1), production site 2 (PS 2) and reference site 2 (RS 2) to corrosion and scale formation

Site	Date	pH	Hardness mg/L	Langelier Index (LSI)	Ryznar Index (RI)	Aggressiveness Index (AI)
PS 1	Dec 2005	8.6	95	0.42	7.7	12.4
	Apr 2006	7.8	102	-0.20	8.2	11.8
	May 2006	8.2	108	-0.06	8.2	12.1
	Jul 2006	7.6	98	-0.76	9.1	11.4
RS 1	Dec 2005	6.4	22	-2.94	12.2	9.0
	Apr 2006	8.7	26	-0.64	9.9	11.5
	May 2006	8.2	26	-1.14	10.4	11.0
	Jul 2006	7.7	25	-1.71	11.1	10.4
PS 2	Dec 2005	9.1	25	-0.20	9.5	11.7
	Apr 2006	8.0	29	-1.26	10.5	10.8
	May 2006	7.7	48	-0.99	9.6	11.0
	Jul 2006	7.1	30	-2.12	11.3	9.9
RS 2	Dec 2005	7.8	31	-1.23	10.2	10.7
	Apr 2006	8.2	36	-0.76	9.7	11.3
	May 2006	7.2	22	-2.30	11.8	9.8
	Jul 2006	7.6	25	-1.86	11.3	10.2

The Aggressiveness Index ranged between 10 and 11, indicating a moderate aggressiveness to irrigation components. During July 2006 the combination of very low Ca and Mg levels, low hardness and a neutral pH caused the water of production site 2 to be highly corrosive (LSI < -2.0). The Ryznar Index supported these findings (RI > 9.0). Table 6.4 shows that the water from reference site 2 was also of a corrosive nature. A combination of the indicator indices showed that the surface water of reference site 2 was slightly corrosive during December 2005, April and July 2006 (-0.5 < LSI < -2.0 & RI > 9.0) and was moderately aggressive towards irrigation structures (10.0 < AI < 11.0). Results from the sampling event in May indicated a tendency for the water of reference site 2 to be highly corrosive (LSI < -2.0 & RI > 9.0). The Aggressiveness Index calculated a value of 9.84, which is associated with a high aggressiveness.

Table 6.5 summarises the water quality parameters of near bottom water in all the dams. When comparing parameters to those of surface waters, bottom water differed primarily in terms of pH and inorganic nitrogen measurements. Total suspended solids content of all the dams was less than 50 mg/L and could have had a minor clogging risk. Additional data indicated that the total suspended solids in the dams increased with an increase in depth. The concentration of total dissolved solids differed from surface samples and posed only a minor risk to clogging of irrigation systems (< 500

mg/L). There was a decrease in pH values with an increase in depth. This observation can be ascribed to reduced photosynthesis, stratification during summer and depletion of oxygen by bacterial activity. The average pH values for each dam ranged between 6.0 and 7.0 and would have had a minor effect on clogging of filters and emitters. Inorganic content showed an increase in all dams with an increase in depth. Production site 2 had inorganic nitrogen levels (1.05 mg/L) exceeding the target range proposed by DWAF (< 0.5 mg/L). The likelihood of clogging problems due to nuisance aquatic plants and algal growth may arise.

Table 6.5: Mean values of near bottom water quality in production site 1 (PS 1), production site 2 (PS 2), reference site 1 (RS 1) and reference site 2 (RS 2) during the irrigation season (November 2005 - March 2006). Ranges indicated in brackets

Site		pH	TSS mg/L	TDS mg/L	DIN mg/L
PS 1	*x*	7.1	8	210	0.47
	n =11	(6.6 – 8.0)	(3 - 12)	(195 – 235)	(0.2 – 0.8)
RS 1	*x*	7.0	5	63	0.28
	n =11	(6.3 – 7.6)	(3 - 10)	(58 - 74)	(0.1 – 0.9)
PS 2	*x*	6.2	13	62	0.64
	n =11	(5.5 – 7.1)	(7 - 19)	(57 – 74)	(0.09 – 1.05)
RS 2	*x*	7.7	5	70	0.24
	n =11	(6.7 – 7.9)	(3 - 8)	(66 – 77)	(0.1 – 0.5)

The hardness, Ca and Mg concentrations of near bottom water varied little to that of surface water samples (Table 6.3). Water quality data indicated that pH values decreased with depth (Chapter 3). During the summer stagnation period, a significant difference (p = 0.05) was observed between surface and bottom water samples. The lower pH could be ascribed to oxygen depletion due to bacterial activity and stratification. Water from deeper layers in production site 1 also showed a slight corrosive character (Table 6.6). During December 2005, the slight corrosive tendency was supported by the Ryznar Index values with a moderate aggressiveness. The situation improved during April and May 2006 with the water being balanced. The Ryznar Index values were also slightly lower supporting the Langelier Index calculations. During July, the situation worsened and a higher degree of corrosion was expected.

Table 6.6 indicates that near bottom water of reference site 1 was slightly corrosive during December 2005, April and May 2006. The Ryznar Index supported these findings with values exceeding 9. At the same time the Aggressiveness Index determined the water to be moderately aggressive (10.0 > AI < 11.9). During July 2006, the combination of a low hardness and a neutral pH resulted in the near bottom water of reference site 1 to be highly corrosive (LSI < -2.0) with a highly aggressive action towards irrigation equipment.

Table 6.6: Potential of near bottom water from production site 1 (PS 1), reference site 1 (RS 1), production site 2 (PS 2) and reference site 2 (RS 2) to corrosion and scale formation

Site	Date	pH	Hardness mg/L	Langelier Index (LSI)	Ryznar Index (RI)	Aggressiveness Index (AI)
PS 1	Dec 2005	7.4	95	-0.86	9.11	11.28
	Apr 2006	7.9	108	-0.10	8.10	11.93
	May 2006	7.8	102	-0.41	8.63	11.78
	Jul 2006	7.4	98	-0.96	9.31	11.27
RS 1	Dec 2005	7.5	22	-1.99	11.43	10.11
	Apr 2006	7.5	34	-1.17	10.15	10.86
	May 2006	7.8	26	-1.59	10.98	10.63
	Jul 2006	7.8	25	-2.34	11.79	9.82
PS 2	Dec 2005	8.9	25	-0.56	10.03	11.53
	Apr 2006	6.6	31	-2.58	11.76	9.49
	May 2006	7.0	29	-2.18	11.50	10.01
	Jul 2006	6.9	30	-2.34	11.57	9.76
RS 2	Dec 2005	7.9	31	-1.17	10.24	10.85
	Apr 2006	8.2	36	-0.76	9.72	11.38
	May 2006	7.1	25	-2.37	1.85	9.75
	Jul 2006	7.1	25	-2.37	11.87	9.80

The results of all three indication systems point out that the bottom water of production site 2 had a slight corrosive tendency during December 2005, with no risk of scaling. The Aggressiveness Index value indicated the water to possibly act moderately aggressively towards irrigation equipment. From April onwards the pH of the bottom water in production site 2 declined to values < 7. The low pH values, in conjunction with the low hardness levels, lead to corrosive water (LSI < -2.0). The Ryznar Index for these sampling events exceeded 9, supporting the strong corrosive nature of the water. Additional calculations of the Aggressiveness Index (AI < 10.0) found the water to be highly aggressive. Calculation of the Langelier Index indicated that the water of reference site 1 and reference site 2 had a slight corrosive nature. The corrosive tendency of the water intensified during May and July 2006 and calculations of the Ryznar Index strongly support these findings (RI > 9.0). Low calcium and magnesium concentrations, low hardness and neutral pH values (Table 6.6 and 6.3) brought on the corrosive character during this period. The results of the Aggressiveness Index verified that the water of reference site 2 was highly corrosive during May and July 2006 (AI < 10).

6.4 Discussion

The integration of cage culture into existing farm dams offers an opportunity for multiple resource utilisation to produce alternative fish protein and to apply the nutrient enriched water for crop growth (Naegel, 1994; Fernando & Halwart, 2000; Ingram *et al*., 2000; Prinsloo *et al*., 2000). However, a number of problems can arise between the management of water quality for both irrigation and cage aquaculture in these dams (Bucks *et al*., 1979; Nguyen-Khoa & Smith, 2004). The following discussion will assess water quality parameters that could have been altered by the preceeding trout production season and whether this could have an impact on the performance of irrigation equipment.

Cage culture can potentially increase the suspended solid and nutrient content in the water column by the addition of high inputs of metabolic waste and uneaten fish feed (Sterling & Dey, 1990; Selong & Helfrich, 1998). Such organic matter will result in a change in trophic status of the dam and could lead to a eutrophic and even hypereutrophic system. Hypereutrophic conditions are characterised by great fluctuations in dissolved oxygen and pH levels and a higher phytoplankton and zooplankton biomass (Sterling & Dey, 1990).

Higher suspended solid concentrations will increase the risk of clogging by small particles that were not removed during the filtration process. The total suspended solid content in all the dams was well below the proposed standard (< 50 mg/L) (DWAF, 1996a). Capra & Scicolone (2005) evaluated the importance of suspended solids on the performance of irrigation components and found that suspended solids content greater than 50 mg/L did not allow optimal emission uniformity. Suspended solid concentrations were higher during winter months when compared to the summer months but never exceeded 50 mg/L. The higher suspended solids coincided with the trout production season as well as higher rainfall and runoff rates, whereas the summer concentrations indicated that suspended solids had settled to the bottom (Chapter 3). Therefore, suspended solids posed no threat to irrigation equipment, as irrigation water is only extracted during the summer months and only surface water is extracted in the dams.

Prinsloo *et al*. (2000) evaluated the impact of fish farming on water quality. Nitrogen-nitrate values from this study ranged between 4 - 12 mg/L, which exceeded the recommended target range (< 0.50 mg/l) set by DWAF (1996a). Orthophosphate values from Prinsloo *et al*. (2000) were also extremely high and the combination of these two nutrients favoured the growth of high phytoplankton biomass. Nitrogen values from the present study were very low when compared to the findings of Prinsloo *et al*. (2000). However, nitrogen did exceed the proposed target range and can therefore cause an increase in phytoplankton biomass.

The phytoplankton species identified were all small enough to pass through sandfilters but were larger than the filtration capacity of the secondary ring filters (80 µm). The highest abundance of dinophytes and heterokontophytes were found during the winter months and this does not coincide with the

irrigation season. During the irrigation season these groups were present in fairly low numbers and therefore posed no threat to physical clogging of filters. The group to be concerned about were the cyanophytes. The most dominant species, *Anabaena circinalis*, occurred in the midst of the irrigation season and could cause major clogging problems in irrigation structures and equipment (Bucka, 1989; Shapiro, 1990). Warmer water temperatures in summer favour the growth of cyanophytes in dams where nutrient conditions are favourable (Chapter 4). The frequency and magnitude of *Anabaena circinalis* were more prominent in production sites, whereas they were almost absent in reference sites (Chapter 4). This suggests that the additional nutrients from aquaculture practices could have created favourable nutrient conditions for cyanophytes to thrive in the summer months following the fish farming season. Species from the classes Cyanophyceae, Chlorophyceae, Bacillariophyceae and various Protozoa were identified in a study by Ahmed *et al*. (2007), as biological factors that cause rapid clogging of micro-irrigation equipment.

Decomposition of excess organic matter, following a phytoplankton collapse, would deplete oxygen from the hypolimnion and create anoxic conditions (Boyd *et al*., 1975; Boyd *et al*., 1978). Anoxic bottom water favours the rate of iron and manganese release which increases the soluble iron and manganese content (Mortimer, 1941). As irrigation water is emitted, soluble iron and manganese will come into contact with oxygen and insoluble oxides will precipitate. The iron content of the reference sites remained fairly low (< 0.2 mg/L) during all the sampling periods and posed no serious clogging problems to irrigation components. The two production sites showed a similar pattern, but experienced slightly higher iron concentrations during the turnover phase in winter. Iron concentrations in production sites reached up to 0.2 mg/L and posed a moderate clogging risk for irrigation equipment. The slightly higher iron content at production sites coincided with the trout production season. Irrigation activities only recommenced after the trout production season, by which time iron levels would have settled to the sediment during summer stratification. Manganese levels were very low (< 0.006 mg/L) in all the dams, therefore not contribute to clogging of equipment.

In general, pH values in all the dams fluctuated around neutral, however, the two production sites reached higher pH levels (> 8) during the summer stratification period. Both of these dams sustained a high phytoplankton biomass. It is well documented that phytoplankton biomass can have considerable effects on the pH of aquaculture ponds (Talling, 1976). During phytoplankton photosynthesis carbon dioxide (CO_2) and more oxygen (O_2) is produced that cause the pH to increase (Talling, 1976). Irrigation water containing high concentrations of Ca, Mg and high hardness, an increase in pH act as catalyst for precipitation reactions and cause scale formation (Prinsloo *et al*., 2000; Koegelenberg *et al*., 2002; Ahmed *et al.,* 2007). On the other hand, oxygen levels are reduced when photosynthesis terminates and respiration continues. A lowering in oxygen levels will increase acidity and give rise to problems with corrosion (Koegelenberg *et al*., 2002). Cumulative results from three indices, indicate that the water from all the sites had a tendency to be slightly corrosive, even during events of high pH (pH > 8) when scale formation was expected. The absence of scale formation could be ascribed to the low hardness and low calcium and magnesium content of waters in the Western Cape (Day & King,

1995). One of the sites (production site 1) experienced a lower degree of corrosiveness and more balanced water. This effect could be related to the somewhat higher hardness levels and higher calcium and magnesium content of the resource. In three of the sites, the corrosive character of the water increased to extremely corrosive conditions during May and July.

Unlike a case study by Milstein & Zoran (2001) where the production season of the culture species and the irrigation season overlap, the trout in the study dams were harvested before full-scale irrigation commenced. Since the trout production season did not overlap with the irrigation season, it would not have any direct impacts on the performance of irrigation equipment. However, the impacts of cage aquaculture on the irrigation resource water were evident from elevated pH values and increased phytoplankton biomass.

6.5 Conclusions

A number of vital problems exist concerning the conservative use of our freshwater resources and this is especially the case where multiple users are involved. For example, the integrated usage of farm dams for both irrigation and cage aquaculture pose a number of conflicts. Aquaculture can affect the performance of irrigation systems by altering water quality parameters that can lead to severe clogging or corrosion of irrigation equipment. Clogging effects of aquaculture could derive from changes in the phytoplankton biomass and species composition, suspended solid concentrations and the pH of the source water.

The annual maxima in phytoplankton biomass in the studied dams did not coincide with the irrigation season, however, the shifted species composition may pose problems for clogging (Chapter 4). The irrigation season was characterised by higher temperatures and nutrient conditions, which encouraged the growth of a clogging causing phytoplankton species. Increased suspended solids were restricted to winter months and tend to settle out before the irrigation season commences. The pH of the dams did not pose a direct threat to clogging, but rather indirectly by accelerating precipitation reactions. In some cases, very high pH values were recorded and scaling was expected. The low mineral (Ca & Mg) content and low hardness, however, exclude scaling as a possibility. The mineral and heavy metal content was in general very low, and could be ascribed to the geological nature of the region. The present study indicated that water quality related problems might arise from the integration of aquaculture into irrigation resource water, primarily through changes in pH, phytoplankton abundance and species composition. Therefore, the practice of aquaculture in the study dams and its resultant effect was more likely to contribute to clogging of irrigation equipment as opposed to scaling and corrosion.

CHAPTER 7

GENERAL DISCUSSION AND CONCLUSIONS

7.1 General discussion and conclusions

Cage aquaculture of rainbow trout (*Oncorhynchus mykiss*) has been introduced in small farm dams (<20 ha) in the Western Cape Province. Of primary concern is the impact of cage aquaculture introduction on water quality of the water resource and the subsequent feedback on future aquaculture projects (production performance) and for irrigation water use.

Rainbow trout, as an aquaculture production species, are very sensitive in terms of their water quality requirements. Conditions outside their optimal range can lead to physiological stress resulting in poor production performance (low feed conversion rates, suboptimal growth, poor flesh quality etc.). The primary water quality parameters that cause physiological stress to rainbow trout are extremely alkaline conditions (pH exceeding 9.0), low dissolved oxygen levels (levels below 5 mg/L), high concentrations of free ammonia (> 0.05 mg/L) and toxins secreted by living cyanobacterial cells. During the fish farming season the cultured fish species are at times subjected to extreme fluctuations in these parameters.

The distribution of nutrients in a farm dam ecosystem is driven by the presence of a thermocline and the mixing characteristics of the water column. The prevailing Mediterranean climatic conditions of the Western Cape Province caused the dams to stratify (stagnation period) during the summer months. The stagnation period was associated with an oxygen depleted hypolimnion and with the accumulation of high ammonia concentrations (anoxic conditions hinder nitrification). The presence of a thermocline during the stagnation period restricted the even distribution of ammonia and phosphorus (released from sediment during anoxic conditions) throughout the water column and nutrients remained concentrated in the hypolimnion. During winter (when fish farming takes place) the dams experienced the annual turnover of the water column, causing the nutrients in the hypolimnion to mix with the epilimnion. The mixing of nutrients into the epilimnion leads to extreme phytoplankton fluctuations which again cause dissolved oxygen concentrations and pH to fluctuate. Hydrological aspects such as water level, size and depth of the dams also affect the mixing regime of farm dams. The study dams were subjected to the abstraction of water for irrigation during summer months. This resulted in a shallow monomictic system. Two dams were affected during the second year of study. The shallow depth and the absence of a thermocline caused nutrients to mix continuously into the epilimnion. This gave rise to continuing increases in phytoplankton biomass that was followed by a sudden die-off of the population, causing extreme fluctuations in dissolved oxygen and pH levels.

Data analysis showed that differences in nutrient concentrations did not differ significantly between reference and production sites, indicating that the introduction of cage aquaculture did not alter the

nutrient conditions of the production dams. However, higher ammonia levels together with oxygen depletion in the hypolimnion of production dams indicated a higher degree of organic enrichment compared to reference sites. Organic enrichment from aquaculture and agricultural input (fertiliser and pesticide application, burning of adjacent vegetation, etc.) lead to the higher phytoplankton biomass in production sites when compared to references sites. The phytoplankton composition also portrayed the more eutrophic character at the production sites that were frequently dominated by algal (e.g. *Ceratium hirundinella*) and cyanobacterial blooms (e.g. *Anabaena* spp. and *Microcystis* spp.).

The extent to which aquaculture adds to the nutrient condition of farm dams is still unknown. The amount of nutrients that accumulate in the system from surrounding land use management and agricultural input (fertiliser and pesticide application, burning of vegetation) has not yet been estimated. Additionally, the coincidence of the winter mixing period and the production season (when fish are in cages) masks the actual nutrient loading by aquaculture. Overall the effects of cage aquaculture can currently be seen in the severity of oxygen depletion in the hypolimnion, elevated ammonia concentrations, increased phytoplankton biomass as well as changes in the species composition of the phytoplankton.

The heterogeneity of farm dams makes it difficult to compare reference sites with production sites, as all sites have different additional environmental variables that may influence water quality conditions. The degree to which aquaculture adds to the current nutrient conditions of farm dams in the Western Cape Province is also still unknown, mainly due to limited historical water quality data of the mostly privately owned dams, the lack of data on initial water quality conditions prior to the introduction of aquaculture and the lack of a national monitoring program for farm dams. It is recommended that better information and insight on overall effects could be derived from comparisons within the same dam over a few years, as well as studies on water quality before and after the introduction of aquaculture. The data and conclusions from the present study already provide a starting point for future comparisons and management considerations.

With or without the presence of aquaculture, parameters that influence the performance of irrigation systems include anoxic water (release Fe and Mn from sediment), pH (water's ability to scale formation or corrosion), total suspended solids and filamentous algal species. Aquaculture can add to the deterioration of irrigation water quality by increasing the amount of organic waste (faeces, unconsumed fish feed) to the bottom of the dam. The breakdown of these wastes increases the suspended solids content and nutrients, which accelerate phytoplankton growth resulting in fluctuations in dissolved oxygen concentrations and pH.

High phytoplankton biomass during the winter months did not overlap with the irrigation season during summer months. More nutrients introduced by aquaculture will not only increase the phytoplankton biomass, but could lead to changes in species composition. The more frequent appearance of filamentous cyanobacteria (e.g. *Oscillatoria* spp. and *Anabaena* spp.) as well as larger species (e.g.

Ceratium hirundinella) experienced in production dams could therefore increase problems with clogging of filters and emitters. The phytoplankton species responsible for clogging problems appeared during the summer months in the study dams. Additional nutrients added by aquaculture during the winter production season can cause an increase in their biomass in the consecutive summer irrigation season. The rise in pH values associated with the eutrophied systems (due to photosynthesis by algae) could cause the occurrence of scaling and subsequent clogging of irrigation systems. However, the naturally low concentrations of calcium and the low alkalinity of the study dams contribute more to corrosion of irrigation structures than scale formation.

The effect of the water on micro dripper irrigation structures could not bring significant results on changes, but some effects could provide future research opportunities. The mayor influence of cage aquaculture on irrigation water quality were phytoplankton composition shifts, elevated biomass and fluctuations in pH due to photosynthetic activity of phytoplankton.

APPENDIX I

Table 1: Irrigation clogging factors and their clogging potential according to Bucks *et al*. (1979)

		Clogging potential		
Clogging factor		**Minor**	**Moderate**	**Severe**
Physical				
suspended solids	mg/L	< 50	50 - 100	> 100
Chemical				
pH		< 7	7 - 8	> 8
dissolved solids	mg/L	< 500	500 - 2 000	> 2 000
manganese	mg/L	<0.2	0.2 - 1.5	> 1.5
iron	mg/L	<0.2	0.2 - 1.5	> 1.5
hardness, as $CaCO_3$	mg/L	<150	150 - 300	> 300
Biological				
bacteria (plate count/mL)		< 10 000	10 000 - 50 000	

Langelier Saturation Index

$$LI = pH_a - pH_s$$

where: LI = Langelier Saturation Index
pH_a = actual measured pH
pH_s = saturation pH

For TDS, 200 mg/L

$pH_s = -0.014732 \times t + 2.30149 + 0.00065 \times TDS + 9.70167 - \log(2.4972 \times Ca) - \log(alkalinity)$

For 200 < TDS < 300 mg/L

$pH_s = -0.014732 \times t + 2.30149 + 9.84 - \log(2.4972 \times Ca) - \log(alkalinity)$

For TDS > 300 mg/L

$pH_s = -0.014732 \times t + 2.30149 + 0.00006786 \times TDS + 9.8336 - \log(2.4972 \times Ca) - \log(alkalinity)$

where: t = temperature (°C)
TDS = total dissolved solids (mg/L)

Ca = calcium concentration (mg/L)

Alkalinity = total alkalinity (mg/L)

Table 2: Prediction of water characteristics following the Langelier Saturation Index

LI	Tendency of water
+ 2.0	Scale forming, non-corrosive
+ 0.5	Slightly scaling and non-corrosive
0.0	Balanced, but pitting corrosion possible
- 0.5	Slightly corrosive and non-scale forming
-2.0	Highly corrosive

(Following Koegelenberg *et al*., 2002; DWAF, 1996a)

Ryznar Stability Index

$$RSI = pH_a - 2 \times LI$$

where: RSI = Ryznar Stability Index

pH_a = actual measured pH

LI = Langelier Saturation Index

Table 3: Prediction of water characteristics by the Ryznar Index

RSI	Tendency of water
4.0 – 5.0	Heavy scale
5.0 – 6.0	Light scale
6.0 – 7.0	Little scale or corrosion
7.0 - 7.5	Corrosion significant
7.5 – 9.0	Heavy corrosion
> 9.0	Corrosion intolerable

(Following Koegelenberg *et al*., 2002; DWAF, 1996a)

Aggressiveness Index

$$AI = pH + \log(A \times H)$$

where: AI = Aggressiveness Index

A = total alkalinity (mg/L $CaCO_3$)

H = calcium hardness (mg/L $CaCO_3$)

Table 4: Interpretation of the Aggressiveness Index

AI	Water property
> 12.0	Non-Aggressive
10.0 – 11.9	Moderately aggressive
< 10.0	Highly aggressive

(Following Koegelenberg *et al.*, 2002; DWAF, 1996a)

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