Effects of rainbow trout (*Oncorhynchus mykiss*) cage culture on Western Cape irrigation reservoirs

Monika Maleri

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Promoter: Dr. A. Leslie (Stellenbosch University, South Africa)

Co-Promoter: Prof. Dr. U. Saint-Paul (ZMT, Bremen, Germany)

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Declaration

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Abstract

Water storage in reservoirs forms an integral part of the agricultural landscape in the Western Cape Province, South Africa. A few large reservoirs serve primarily as a drinking and industrial water supply, while on private farms, small reservoirs provide irrigation water for the dry summer period. Protection of water quality to secure irrigation and drinking water quality, and the extension of water use efficiency are priority issues in the Western Cape. In the current study, the suitability of rainbow trout (*Oncorhynchus mykiss*) cage farming as a non-abstractive water use was investigated. The current study concentrated on the identification, and where possible quantification of aquaculture impacts, the identification of successful sites and a description of requirements in which net-cage aquaculture has none or a very low negative impact on water quality (e.g. <15 % change from previous water quality conditions for phosphorus concentrations).

In order to study the effects of 5 t trout cage production units in reservoirs <15 ha in area, the general ecology of the irrigation reservoirs was assessed. Sixteen reservoirs without aquaculture production (reference reservoirs) were compared to 26 reservoirs with aquaculture production (production sites with varying production histories). Catchment characteristics were also monitored. Water from different depths (0 m, 2 m, 6 m and near bottom) was tested for physical and chemical qualities as well as nutrient concentrations. Phytoplankton and zooplankton biomass and species composition was monitored. In addition to the general phytoplankton findings, cyanophytes were evaluated for their contribution to algal taint problems that emerged at a number of production sites. Sediments were tested for total phosphorus content and phosphorus release capacity. Indicators and minimum conditions to avoid the most common production problems were formulated. In order to determine long-term production success, which prevents trophic level changes of reservoirs, a mass balance approach (nutrient budget) was employed to indicate the limits for nutrients that can be added. The phosphorus balance indicated long term trends for reservoirs with and without aquaculture. The detailed mass balance approach was compared to a "ready to use" carrying capacity model that estimated the maximum fish load each reservoir could support.

The anthropological input of phosphorus into the reservoirs causes a decreasing water quality in the studied reservoirs and this development was also reported for lowland rivers. Twenty percent of the studied reservoirs are in a condition that could be an immediate threat to fish or water bird health (e.g. free ammonia concentrations and pH). Harmful algal blooms were not observed.

Aquaculture production evoked changes in water chemistry and ecology in most of the studied reservoirs. Adverse effects of aquaculture sites versus non-aquaculture sites were: increased phytoplankton biomass and species shifts towards sizes >80 µm. The increased phytoplankton abundance influenced pH maxima to values >9 at mid-day. The high pH fluctuations were greatly influenced by the naturally low alkalinity and hence low calcium buffering capacity of Western Cape waters. The deoxygenation of the hypolimnion during stagnation (summer) occurred faster in reservoirs of certain character, greatly dependent on elevation and surface area, with consequent acidification of the hypolimnion, as well as ammonia and total phosphorus (TP) accumulation. In this context, a diversity of each reservoir with and without aquaculture production, with a similar ratio of undisturbed reservoirs to reservoirs with influence of e.g. agriculture, were compared to each other. When grouping the respective differences from the average reference reservoir (i.e. no trout production), a low impact on water quality was manifested at four sites (15 %) with <15 % increase of bottom TP and ammonia, while eight sites (31 %) showed medium effects (59 % mean increase), and a high impact was found at 54 % of the sites (312 % mean increase). In reservoirs without aquaculture, the extent of incoming phosphorus (which could represent an influence by runoff from agricultural land) was very high. However, in small reservoirs (<5 ha), these values were exceeded by the incoming phosphorus from aquaculture practices. In the case of small reservoirs where the carrying capacity was clearly surpassed, effects caused by aquaculture were severe and the assimilation of waste by the system was not possible (in extreme cases aquaculture waste delivered 60 to 90 % of all incoming phosphorus – two to nine times the phosphorus brought in by rivers and runoff).

With regards to sediment, only indirect conclusions could be drawn. Aquaculture production increased hypolimnetic anoxia and the latter was shown to increase potential phosphorus release from sediments. This implies that not only will aquaculture increase the phosphorus concentration of surface waters directly, but it will also increase internal loading. The sedimentation rate was increased with cage aquaculture which affects a hypothesized area of approximately 0.2 to 1.0 ha depending on reservoir hydrology. The composition of the sediment increased organic components which can impact on sediment processes. It can be postulated that increased sedimentation of aquaculture waste and extended anoxic conditions impact on macrozoobenthos.

Hydrological and nutrient mass exchange of the reservoirs indicated that no annual increase of phosphorus was achieved with low nutrient input (good inflowing water quality) or good water exchange (>5 times per year), and sometimes with extraction of hypolimnetic water during the stagnation period (summer). A model developed by Beveridge (1984) showed similar results to the mass balance approach and can therefore serve as a more ready model to determine suitable stocking rates.

The small (man-made) reservoirs in the Western Cape are in a eutrophication process which far exceeds the speed of natural eutrophication (trophic states indicating highly eutrophic or hypertrophic conditions after approximately 10 to 20 years following construction of the reservoirs) and this process is triggered by agricultural practice (indicated by literature – not a subject of this study). However, it is concluded from the data of this study that trout-cage aquaculture duplicated the total phosphorus already present (independent of continuation of the projects, the phosphorus introduced was trapped in the closed systems the reservoirs represent) in only 1 to 2 years of production - which means a significant acceleration of the eutrophication process already in place. There are positive exceptions where trout-cage production is possible without negative effects.

Careful site selection is the most important step in successful and sustainable trout production. No impact of aquaculture was recorded at four reservoirs (15 % of the investigated reservoirs) which shared the characteristics of good water exchange (>3 times per year) and a minimum surface area of 5 ha. Additionally, criteria that reduced the risk of algal taint included a minimum water depth of 6 to 7 m in a reservoir at its lowest water point (to avoid intermediate mixing during the stagnation period) as well as cold hypolimnetic conditions (<17 °C) to minimize cyanophyte cyst remobilization.

Further improvement of food conversion ratio (feeding management) and feed quality are the next (after site selection) two most important components that determine if a reservoir can be utilised for cage production without any long-term changes. There is potential in advances in feed quality, feed management and waste collection systems. These measures (e.g. the cage size could be decreased to efficate feeding management) can increase the number of sustainable sites and achieve multiplication of water use without water quality deterioration.

Opsomming

Die stoor van water in reservoirs vorm ‘n integrale deel van die landboulandskap in die Westelike Kaap gedeelte van Suid-Afrika. ‘n Aantal groter reservoirs voorsien primêr water primer vir drinkdoeleindes en vir aanwending deur industrieë, terwyl kleiner reservoirs op private plase gebruik word vir besproeiing gedurende die droë somerperiode. Die beskerming van die waterkwaliteit en die uitbreiding van watergebruik se doeltreffendheid, word bestempel as ‘n prioriteit vir die Wes-Kaap. In die huidige studie is die geskiktheid van forel hokkultuur as ‘n bykomende watergebruik, maar sonder waterverbruik, ondersoek. Die huidige studie fokus op die identifisering en waar moontlik die kwantifisering van die impak van akwakultuur op besproeiingsreservoirs; die identifisering van potensiële reservoirs vir akwakultuur; en die beskrywing van toestande waarin akwakultuur geen of ‘n baie lae negatiewe uitwerking op die waterkwaliteit (bv. < 15 % verandering vanaf oorspronklike waterkwaliteitstoestande vir fosforkonsentrasies) sal hê.

Om die impak van 5 t forel hokproduksie eenhede in <15 ha reservoirs te bestudeer, is die algemene ekologie van die besproeiingsreservoirs bepaal. Watermonsters, geneem op verskillende dieptes (0, 2, 6 en naby die bodem), is getoets vir fisiese- en chemiese kwaliteit sowel as vir die nutriëntkonsentrasies. Die biomassa en spesiesamestelling van fitoplankton sowel as zooplankton is gemonitor. Sestien verwysingsreservoirs is vergelyk met 26 produksiereservoirs (met wisselende vlakke van produksiegeskiedenis). Die karakeristieke eienskappe van die opvangsgebied is ook gemonitor. Tesame met die algemene fitoplankton bevindinge, is blougroen alge ook geëvalueer vir hul bydrae tot algae besmettingsprobleme wat voorgekom het by ‘n aantal produksie persele. Sedimente is getoets vir die totale fosforinhoud en fosforvrystellingskapasiteit. Perseelseleksie is geidentifiseer as een van die mees belangrikste faktore vir suksesvolle hokproduksie. Indikatore en minimum toetstande wat vereis word om die mees algemene produksieprobleme te verhoed, is geformuleer. Om die langtermyn produksie sukses te bepaal, wat trophiese vlakke se verandering in die reservoirs verhoed, is ‘n massa balans benadering (nutriënt balans) gebruik. Die fosfor balans het langtermyn tendense aangetoon en in die geval van die produksie reservoirs, is die akwakultuurimpak gekwantifiseer. Die gedetaileerde massa balans benadering is vergelyk met ‘n “gereed om te gebruik” drakapasiteitsmodel wat die maksimum vis wat die reservoir kan onderhou, geskat het.

Die antropogeniese toevoer van fosfor na die reservoirs veroorsaak ‘n afname in die waterkwaliteit van die reservoirs wat bestudeer is. Die verskynsel van ‘n afname in waterkwaliteit is ook vir verskeie laagliggende riviere geraporteer. Sowat 20 % van die

besproeiingsreservoirs wat bestudeer is, is tans in ‘n toestand wat die onmiddelike gesondheid van vis en watervoëls kan bedreig. Skadelike algebbloei is nie waargeneem nie.

Akwakultuurproduksie het negatiewe effekte getoon in ‘n aantal reservoirs. Die nadelige uitwerking van akwakultuur waar digthede te hoog was, is: toename in fitoplankton biomassa en spesiesverskuiwinge. Die toename in fitoplankton teenwoordigheid het die pH maxima beïnvloed tot waardes >9 teen 12 uur middag. Die hoë pH fluktuasies is grootliks beïnvloed deur die natuurlike lae alkaliniteit en gevolglike kalsium bufferkapasitiet van die Wes-Kaap se waterbronne. Die deoksiginasie van die hipolimnion gedurende stagnasie (somer) het vinniger plaasgevind in oorbelaaide reservoirs, met gevolglik ‘n versuring van die hipolymnion, sowel as die akkumelering van ammoniak en totale fosfor. In hierdie konteks word reservoirs met en sonder landbouproduksie, met ‘n gelyke verhouding van onversteurde reservoirs tot reservoirs wat deur landboubedrywighede beinvloed word, met mekaar vergelyk. By vier persele (15 %) is ‘n lae impak vasgestel (<15 % toename in bodem TF en ammoniak), terwyl agt persele (31 %) ‘n medium impak getoon het (59 % gemiddelde toename in bodem TF en ammoniak) en ‘n hoë impak is opgemerk by 54 % van die persele (31 % gemiddelde toename in bodem TF en ammoniak).

In reservoirs sonder akwakultuur, was die omvang van fosforinvloeiing baie hoog en is moontlik veroorsaak deur die invloei van afloopwater uit omliggende landbougebiede. Alhoewel in klein reservoirs (<5 ha) was hierdie waardes oorskadu deur die invloeiing van fosfor deur akwakultuur praktyke.In die geval van klein reservoirs, waar die drakrag duidelik oorskry is, was die gevolge, soos veroorsaak deur akwakultuur ernstig, en die assimilasie van afval deur die sisteem nie moontlik nie. In die uiterste gevalle het akwakultuurafval 60 % - 90 % van alle inkomende fosfor gelewer - twee tot nege maal die fosfor wat deur riviere en afloopwater ingevloei het.

Wat die sediment aanbetref, kon slegs indirekte gevolgtrekkings gevorm word. Akwakultuurproduksie het hipolimnetiese anoksia laat toeneem en die laasgenoemde verskynsel, het die potensiaal aangedui vir die toename in die vrystelling van fosfor vanaf die sediment. Dit dui daarop dat akwakultuur nie alleen die fosforkonsentrasie in die oppervlaktwater laat toeneem nie, maar sou ook die interne belading laat kon toeneem. Die sedimentasie tempo het toegeneem met die teenwoordigkeid hokkultuur en het ‘n hipotetiese area van ongeveer 0.05 tot 1.00 ha, afhangende van reservoir hidrologie, beïnvloed. Die samestelling van die sediment het toegeneem in teme van die organiese komponente wat die sedimentasie prosesse kon beïnvloed. Dit kan gepostuleer word dat die toename in sedimentasie van akwakultuurafvalprodukte tesame met verlengde anoksiese toestande, ‘n invloed op die makrosoöbentiese organismes het.

Hidrologiese en nutriënt massa uitruiling van die reservoirs het aangetoon dat geen jaarlikse toename in fosfor verkry kan word met lae nutrient toelae (kwaliteit van invloeiende water) of met goeie waterverplasing nie, en soms met die ekstraksie van hipoliminetiese water gedurende die stagnasie periode (somer). Die Beveridge model het soortgelyke resultate getoon tot die massabalans benadering en kan daarom dien as ‘n meer aanvaarbare model om gepaste beladingstempo vas te stel.

Kleiner mensgemaakte reservoirs in die Wes-Kaap is onderhewig aan ‘n eutrofikasie proses wat die spoed van natuurlike eutrofikasie (trofies verwys na ‘n hoogs eutrofiese of hipertrofiese toestand ongeveer 10 tot 20 jaar na reservoir konstruksie) oorskry. Literatuur (nie ondersoek in die huidige studie) dui aan dat hierdie versnelde eutrofikasie proses meegebring word deur landbouaktiwiteite in die opvangsarea van die reservoirs. Resultate van die huidige studie het getoon dat forelproduksie in hokstelsels, die konsentrasie van total fosfor wat reeds beskikbaar was, verdubbel het. Die toename in fosforkonsentrasie het binne 1 tot 2 jaar na die aanvang van forelproduksie in die betrokke reservoirs, plaasgevind. Daar is egter uitsonderings waar forelproduksie in hokstelsels moontlik was, sonder die gepaardgaande afname in die waterkwaliteit.

Die belangrikste stap vir suksesvolle en volhoubare forelproduksie is deeglike perseelseleksie. Daar is geen impak van akwakultuur waargeneem by vier persele (15 %) wat die eienskappe van goeie waterverplasing (>3 keer per jaar) en ‘n minimum oppervlakarea van 5 ha gehad het nie. Bykomend, sluit kriteria wat die risiko van algbesmetting laat afneem, ‘n minimum waterdiepte van 6 tot 7 m in ‘n reservoir by die laagste punt in (om te verhoed dat intermediêre vermenging plaasvind gedurende die stagnasieperiode) sowel as koue hipolimnetiese toestande (<17 °C) om sianobakterieë sist remobilisasie te minimaliseer.

Verdere verbetering van die VOH (voeromsettingsverhouding onder voedingsbestuur) en voerkwaliteit is na perseeleleksie, die volgende komponente wat kan aandui of ‘n perseel gebruik kan word vir hokkultuur sonder enige impak. Vordering met voerkwaliteit en voedingsbestuur kan die aantal volhoubare persele laat toeneem en daardeur meer effektiewe watergebruik teweeg bring, sonder die verwante waterkwaliteit verswakking.

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LIST OF ACRONYMS

AAP	Algal available phosphorus
"a.m.s.l."	Above mean sea level
DEAT	South African Department of Environmental Affairs and Tourism
DWAF	South African Department of Water Affairs and Forestry
dw	Dry weight
FAO	Food and Agriculture Organization of the United Nations
FCR	Food Conversion Ratio
Geosmin	1,2,7,7-tetramethyl-2-norborneol
MIB	2-methylisoborneol
RDP	Readily desorbable phosphorus
P	Phosphorus
TDS	Total dissolved solids
TN	Total nitrogen
TP	Total phosphorus
TSI	Trophic State Index
TSS	Total suspended solids
WMA	Water Management Area
WRC	Water Research Commission
WSP	Water soluble phosphorus
ww	wet weight

GLOSSARY

Aerosols

An aerosol is a suspension of fine solid particles or liquid droplets in a gas.

Alkalinity

Alkalinity is a measure of the ability of water to neutralise acids to the equivalence point of carbonate and bicarbonate. The alkalinity is equal to the stoichiometric sum of the bases in solution. In the natural environment carbonate alkalinity tends to make up most of the total alkalinity due to the common occurrence and dissolution of carbonate rocks and presence of carbon dioxide in the atmosphere.

Ammonia

Ammonia is a compound with the formula NH_3. In water, free ammonia (NH_3-N) and ionised-ammonia (NH_4^+-N) represent two forms of reduced inorganic ammonia-nitrogen which exist in equilibrium depending upon the pH and temperature of the waters in which they are found. Of the two, the free ammonia form is considerably more toxic to organisms such as fish. Free ammonia is a gaseous chemical, whereas the NH_4^+ form of reduced nitrogen is an ionized form which remains soluble in water.

Anoxia

Literally, anoxic means the absence of oxygen. Within this report, the term is primarily used in context with anoxic hypolimnia (depletion due to decomposition rates and lack of mixing). The threshold for an anoxic hypolimnion was set at an oxygen content of less than 1 mg/L.

Catchment area

An area from which surface runoff is carried away by a single drainage system (river, basin or reservoir).

Epilimnion

The epilimnion is the topmost layer in a thermally stratified water body, occurring above the deeper hypolimnion. It is warmer and typically has a higher pH and dissolved oxygen concentration than the hypolimnion. Being exposed at the surface, it typically becomes turbulently mixed as a result of surface wind-mixing. It is also free to exchange dissolved gases (O2 and CO2) with the atmosphere.

Eutrophication

This is the process of accumulating chemicals, especially the nutrients nitrogen and phosphorus. Eutrophication occurs as a natural phenomenon that is accelerated by human activities in the environment (sewage effluent, fertiliser runoff, etc.). Most commonly, the total phosphorus content defines the trophic level of the water body by phosphorus often being the limiting nutrient for algal growth. In South Africa, eutrophication levels are defined by the total inorganic phosphorus content (usually 50 to 80 % of the total phosphorus content).

< 5 µg/L	oligotrophic
5 to 25 µg/L	mesotrophic
25 to 250 µg/L	eutrophic
> 250 µg/L	hypereutrophic

Food conversion ratio (FCR)

The true feed conversion ratio is the ratio of the gain in wet body weight of the fish to the amount of dry feed fed. Wasted feed and mortalities are included in that ratio. With commercial dry pellets, a feed conversion ratio of 0.8 is theoretically possible for trout. However, most performances range between 1:1 and 1:1.5 (e.g. 1 and 1.5 kg, respectively, of dry feed has to be applied to achieve 1 kg of wet weight gain in the fish).

Hypolimnion

The hypolimnion is the bottom layer of water in a thermally stratified lake, situated below the thermocline. In deeper water bodies, the hypolimnion temperatures are often similar to the average air temperatures in winter (in a Mediterranean or subtropical climate). Being at depth, the hypolimnion is isolated from surface wind-mixing, and usually receives insufficient irradiance (light) to enable photosynthesis. Oxygen exchange with the surface layer or air is also inhibited.

Monomictic

This term refers to the number of mixing events per year in a water body. In Mediterranean and subtropical regions, the temperatures of epilimnion and hypolimnion are isothermal (of the same temperature) in winter, so that there is only one mixing phase per year, lasting from two to several months. In temperate regions, most water bodies have two stagnation and two turnover phases and are referred to as dimictic (they mix twice in the course of one year). The periods of mixing here occur in spring and autumn and usually last a few weeks only.

Phosphorus species

By convention, the phosphorus content of a water body is determined primarily by determining total phosphorus (to describe the trophic state) and dissolved inorganic phosphorus (also referred to as soluble reactive phosphorus), the latter as the most readily biologically available form of phosphorus (e.g. to phytoplankton).

The total phosphorus content of a water body can be divided into its dissolved or particulate (sometimes also referred to as soluble and insoluble) and inorganic or organic components (the latter often also referred to as reactive and unreactive P, inorganic P). Theoretically, there are eight subspecies of total phosphorus: Total inorganic P divided into dissolved inorganic P and particulate inorganic P, total organic P divided into dissolved organic P and particulate organic P, as well as the sums of total particulate and total dissolved P. There are no analytical methods that would allow the exact measurement of the content of each of these fractions.

Analytically and by scientific convention, the division into particulate and dissolved P, is therefore made by filtration through a 0.45 µm membrane. Filtered material is defined as dissolved and the residual P as particulate (this can only be determined by subtracting the dissolved component from the total P content). Inorganic (reactive) and organic (unreactive) P are separated analytically by the building of a vanado-molybdo-phosphate complex (of blue colour) of only the inorganic partition. The organic (unreactive) component can only be determined by subtracting the inorganic content from the total phosphorus content.

Phytoplankton

Phytoplankton consists of microscopic free-floating, primarily autotrophic (photosynthetic) organisms in aquatic systems, mainly unicellular algae, sometimes organised in colonies. Own movement is sometimes possible, but phytoplankters, in contrast to larger hydrological forces, are drifters (plankton). In limnological research, for practical reasons, it is common to distinguish phyto- and zooplankton by size, as the two groups are divided by sieving through 150 to 200 µm mesh sizes. In some cases, genera were originally defined by autotrophic species, however, mixotrophic and heterotrophic species were later found to belong to the same taxonomical genus.

Reservoir (Dam)

A dam is either a man made barrier constructed across a waterway to control flow or raise the level of water or the body of water that is contained by such a barrier. Another term for the latter meaning of dam is reservoir. In South Africa, small farm dams primarily serve the purpose of water storage for irrigation or drinking supply.

Runoff

Surface runoff is the water flow that occurs when excess water from rain or other sources flows over the land. When runoff flows along the ground, it can pick up soil contaminants such as pesticides (in particular herbicides and insecticides), or fertilizers that become discharged or nonpoint source pollution.

Stagnation phase/Stratification

In Mediterranean and subtropical climates, a thermocline develops during the summer months and divides the upper water layer (epilimnion) from the lower water layer (hypolimnion). Due to reduced water exchange by prevented mixture of water, this phase is called the stagnation phase.

TSS (Total suspended solids)

Total suspended solids (TSS) include both suspended sediment and organic material collected with the water sample. Suspended solids in water reduce light penetration in the water column, can clog the gills of fish and invertebrates, and are often associated with toxic contaminants because organics and metals tend to bind to particles (e.g. phosphorus, bacteria). They also cause the build-up of sediments in water bodies.

Trophic State Index (TSI)

The trophic state index in this thesis was calculated according to Carlson (1977).

TSI < 30	Oligotrophy: Clear water, oxygen throughout the year in the hypolimnion
TSI 31-40	A lake will still exhibit oligotrophy, but some shallower lakes will become anoxic during the summer
TSI 41-50	Mesotrophic: Water moderately clear, but increasing probability of anoxia during the summer
TSI 51-60	Eutrophic: Lower boundary of classical eutrophy: Decreased transparency, warm water fisheries only
TSI 61-70	Highly eutrophic: Dominance of blue green algae, algal scum probable, extensive macrophyte problems
TSI 71-80	Hypertrophic: Heavy algal blooms possible throughout the summer, often hypereutrophic
TSI > 81	Algal scum, summer fish kills, few macrophytes

Turnover phase / Destratification phase

During the winter months, the temperature in Mediterranean and subtropical water bodies tends to be similar throughout the whole water body. During these months, the whole water body (depending on overall depth) can undergo mixing.

Zooplankton

Zooplankton consists of microscopic, free-floating, heterotrophic organisms in aquatic systems which can not produce their own food, but rely on debris and phytoplankton as a food source.

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CHAPTER 1 BACKGROUND & OBJECTIVES

1.1 Introduction

Aquaculture is a growing industry that strives to optimise and intensify traditional fish rearing techniques to supply much needed protein for humans, but also meet the needs of luxury demand. Aquaculture follows the long precedent developments in the agricultural sector, with the opportunity to incorporate comparatively recent ecological considerations at a fairly early point in its development, earlier than agriculture could. In many developing countries though, socio-economic pressure and the importance of economic growth relativise the importance of environmental concerns. With cage aquaculture, ecology and economy are closely linked. Primarily, cage aquaculture uses large water structures with sufficient water exchange. However, should ecological deterioration occur in the used reservoirs (due to smaller size, low water exchange rate etc.), it will usually entail long recovery times. When the environmental limits of the water system are exceeded and ecosystem services such as the provision of good water quality no longer function, reduced yields will no longer be an option with termination of production the only solution. Hence, the immediate gain will be diminished by future losses, which suggests overall sustainability as the solution of reason (Folke & Kautsky 1989). The knowledge of an ecological system and its consequent monitoring to avoid the loss of its functioning will enable and not inhibit the necessity of economic wellbeing.

Farming of rainbow trout (*Oncorhynchus mykiss*) is an industry which has experienced an exponential growth rate since the early 1950's (FAO 2007). The market is largely supplied by intensively producing nations such as the top producers Chile and Norway, both concentrating on seawater and brackish-water production. Chile produced approximately 150,000 t of rainbow trout in 2006 (Eurofish 2008) which accounted for 27 % of the world's total production (550,000 t) in the same year (FAO 2008), marine and freshwater. In freshwater only, the top producers of trout in 2007 were Iran (43,000 t), France (33,000 t), Italy, Denmark, the United States and Spain (McNevin 2008). Most operators use raceway or channelled pond systems which are supplied by cold water streams and are therefore systems with a large water supply demand. In contrast, freshwater cage farming can be combined with a low water consumption rate (Beveridge 2004) and therefore lower initial investment costs. It is consequently attractive for areas with many open water bodies and a population that depends on economic opportunities, given that the water quality conditions are suitable for the chosen species and could be maintained at the original ecological status or with a minimum status change.

South Africa is a minor role player in African (and world) aquaculture production, but has a long tradition in trout rearing. Rainbow trout were introduced into Western Cape rivers in 1892 (Kingfisher 1922) and have historically been used for sport fishing purposes. Growing demands by the domestic market evoked a trend in the well established, but modest industry. At present, about 1500 t of trout are produced within South Africa, with a 350 t production in raceways and cages of the Western Cape (Stubbs 2007). Irrigation reservoirs are currently being tested for trout production in cages. The Hands-On Fish Farmer's Co-operative Limited pairs the pressing need of socio-economic development with the opportunity of trout production in a variety of small reservoirs. The cooperative, an ideal framework for comparative studies, started with one site in 1993 and had increased to 23 cage sites by 2008. The average farm reservoir in the Western Cape has a 5 to 15 ha surface area and the perception was that they can support 5 to 10 t of trout. However, production problems such as tainted fish and increased mortality rates caused by low surface oxygen levels increased with quantity produced.

The ecology and nutrient cycles of these water bodies were little understood, but for further investigations, their general understanding is a precondition to assess the impact of cage aquaculture on these systems specifically. Reference sites (defined as sites without trout production) and production sites (defined as sites with aquacultural production) with common water quality parameters were assessed (Lind 1979, Wetzel & Likens 2000, Höll et al. 1986) as well as indicators known to be important when studying the possible impact of aquaculture (Beveridge 2004, Pillay 2004, Podemski & Blanchfield 2006). Analysis of water and sediment data gave insights into factors causing algal taint as well as short and long term predictions on the impact of aquaculture practises.

1.2 Aims of the thesis

In summarising the available literature on cage aquaculture and its effects on water quality (Chapter 2), it can be concluded that uncertainties still exist in predicting ecological changes. In the South African context of small farm reservoirs (<15 ha), the present status quo and the resilience of these systems contribute to our information gaps.

The broader aims of this thesis were: (i) to describe the ecology of Western Cape farm dams, (ii) to observe general effects of trout cage aquaculture on Western Cape farm dams in contrast to the conditions without aquaculture and (iii) to describe the threshold conditions in which a certain operation size (5 t) can be operated without inducing or enhancing algal taint in the fish flesh and surface oxygen deficiencies (Figure 1.1).

The more specific aims of the thesis were to:

1) Establish the ecology of 16 reservoirs without cage aquaculture in the Western Cape (water quality, phytoplankton, zooplankton, sediment, trophic status); These data serve as reference data due to the lack of literature on the ecology of small inland Western Cape farm reservoirs (the majority of the reservoirs); The aim was to quantify various water quality parameters and to show synergies and dependencies of these parameters with factors known to influence small water bodies (Chapter 3); these data are baseline water quality values;
2) Determine the differences between reservoirs with (26 reservoirs) and reservoirs without cage aquaculture (16 reservoirs) regarding various physico-chemical and nutrient water quality parameters and determine factors that can explain these differences (Chapter 4);
3) Describe sediment characteristics, phosphorus binding capacities and phosphorus release rates in Western Cape reservoirs by comparing reservoirs with and without production (Chapter 5);
4) Establish the nutrient budget (phosphorus) of the commonly used production unit within these reservoirs and compare reservoirs with and without production (Chapter 6);
5) Find indicators to predict 1,2,7,7-tetramethyl-2-norborneol (geosmin) and 2-methylisoborneol (MIB) presence, responsible for algal taint, in Western Cape reservoirs (Chapter 7);
6) Compare site specifics with the measurement of surface oxygen deficiencies and algal taint reportings to determine the threshold (band) at which conditions for production are reliably safe (Chapter 8);
7) Test the compatibility of the current effects (information from previous chapters) of trout cage aquaculture with official guidelines for water quality protection and sustainable production and suggest managerial options (Chapter 9).

1.3 Summary of methods

To achieve Aim 1, seven reservoirs were studied intensively over 26 months and nine other reservoirs were sampled occasionally at three different times per season over two years. For these 16 reservoirs, single important water quality parameters were described and the trophic status determined. Factors that influence the nutrient concentrations were determined and the phytoplankton communities described (Chapter 3).

For Aim 2, 12 production sites were sampled monthly throughout the year over 26 months as well as 14 production sites which were sampled three times per season. The production sites were compared to the water quality conditions and phytoplankton communities within the reservoirs without fish production. Two reservoirs were sampled without production for one year and with production the next year which makes for a good comparison (Chapter 4).

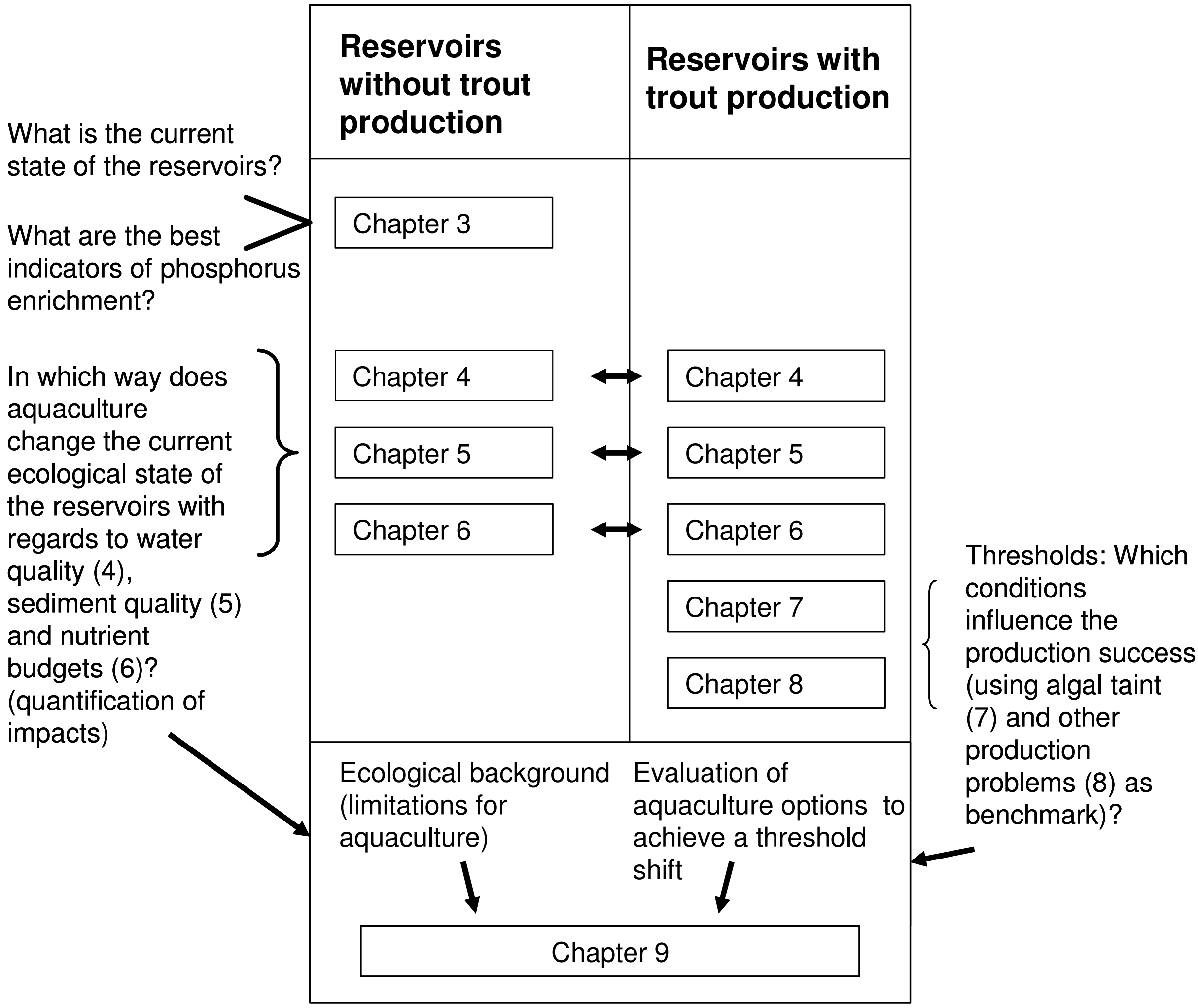

Figure 1.1: Overview of the main questions driving the single chapters or several chapters collectively. The chapters are divided according to the investigation of reservoirs without aquaculture (reference reservoirs) only, chapters comparing reference reservoirs with reservoirs that operated cage aquaculture (production reservoirs) and chapters dealing with the investigation of reservoirs with aquaculture production only (left and right column).

To address Aim 3, sediment samples were fractionated methodologically to identify release rates of phosphorus under different hypolimnetic conditions (Chapter 5).

To achieve Aim 4, preceding data (Aim 1 to 3) were combined to determine the phosphorus budget of 10 intensively studied sites (6 reservoirs with and 4 reservoirs without fish production). The Beveridge model (Beveridge 1984) was compared to the budget approach and the environmental carrying capacity of these reservoirs calculated (Chapter 6).

Aim 5 was addressed via direct comparison of cyanophyte composition and abundance with literature reports on species favouring algal taint, as well as the reports of tainted flesh from Western Cape sites (Chapter 7). Conditions generally favouring higher cyanophyte abundance were identified.

Aim 6 used the statistical comparison of specific conditions that caused production problems (oxygen deficiency and algal taint) with the production data, morphometric data, water quality data and descriptors of production performance to evaluate the quality of a site that would avoid these problems (Chapter 8).

Aim 7 addressed final conclusions and recommendations summarising all research findings of the thesis (Chapter 9).

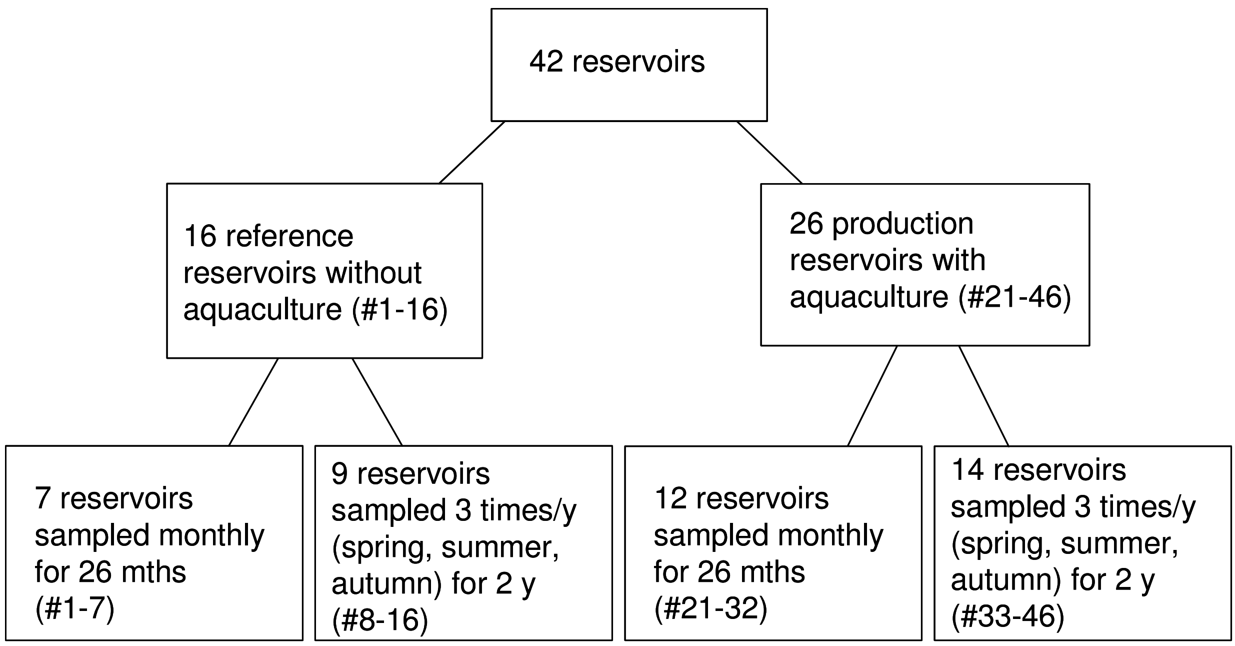

Figure 1.2: Number of reservoirs sampled, divided into the two main groups (with and without aquaculture) and with respective sampling regime.

1.4 Outline of the thesis

The thesis is presented in an article-based format with a brief thesis background (Chapter 1), a literature review (Chapter 2), six separate research articles (Chapters 3 to 8) and general conclusions with recommendations (Chapter 9). Chapter 8 has already been published in a peer-reviewed journal. To avoid unnecessary repetition to the reader of the thesis, publications from chapters may undergo some further adaptation.

To understand the impact of trout cage farming on Western Cape farm reservoirs, the following stepwise approach was used in this thesis:

Chapter 2 focussed on the embedding of all aims into the existing literature as well as to describe and discuss water quality parameters.

In Chapter 3, the current ecological status of 16 reservoirs without production (reference reservoirs) was discussed. Key predictors of eutrophication were established and relations among parameters identified.

Chapter 4 is dedicated to the water quality within farm reservoirs undertaking trout production (production sites) in comparison to the reservoirs without trout production. Effects of cage aquaculture on reservoir ecology were identified and quantified. The effects of aquaculture were the main focus, comparing average reservoir conditions without fish production to average reservoir conditions with trout cage aquaculture.

In Chapter 5, the outcome of sediment analyses is presented, concentrating on phosphorus as the main factor driving eutrophication. The phosphorus concentrations in different sediment fractions was assessed and nutrient release rates extrapolated.

Chapter 6 aims to describe the nutrient input of aquaculture production in the reservoirs and to formulate conditions necessary for sustainable trout cage production by using a nutrient budget approach and a model introduced by Beveridge (1984). The findings support the prediction of long-term sustainability of the currently used production unit in certain reservoir types by estimating the extent of additional nutrient input as well as its major sources and sinks.

Chapter 7 provides an understanding of the conditions favouring algal taint in rainbow trout. The presence and seasonal occurrence of cyanophytes was related to recordings of tainted flesh and threshold conditions to avoid reservoirs that cause algal taint can be set.

Chapter 8 concentrates on site selection criteria to improve the currently used ad hoc basis of site selection for trout cage production. When combining surface oxygen deficiencies and reportings of tainted fish flesh with fish production data, the minimum criteria for site selection can then be set (immediate ecological relief).

Chapter 9 summarizes the previous chapters and evaluates a mitigating method (bacterial kit by Amitek Solutions (Pty) Ltd.) to enhance nutrient release. The extent of europhication of the farm reservoirs and arising imperatives are discussed and the question raised as to whether cage aquaculture can be integrated into a sustainable water resource management of Western Cape irrigation reservoirs.

1.5 General remarks on the research approach

Originally, the aim of the thesis was to support trout cage production in Western Cape reservoirs and improve its success. However, to improve production, the focus transformed into the study of the reservoir system to better understand the conditions of the system that lead to specific patterns (which was basically done by extended correlation analyses of available numerical data and transformation of additional observations and statements from interviews into simple mathematical matrices).

The next important focus of the study was to evaluate possible effects of aquaculture which can be described as continuity or deviation of system rules (which was done by classical statistical comparison of two sets of data). Lastly, one can define the basic conditions that allow to the system to remain within optimum conditions with special reference to the addition of trout cage farming. Therefore, the first aim of this thesis was to understand small Western Cape reservoirs as a system, with secondary research questions that supported the identification of differences among different sets of reservoirs and the origin of these differences (or their co-dependencies).

Systems biology (in particular of rather unknown systems) can not be in the classic sense hypothesis driven from the start, but embraces the conservative hypothesis-approach and approaches it later. At the beginning of this thesis, the similarities among reservoirs and driving patterns were primarily unknown. A sampling collection scheme was selected that secured the generation of consistent data sets. The diversity of Western Cape Reservoirs was identified and reservoirs chosen that entailed evenly distributed qualities among reservoirs with aquaculture production and reservoirs without production.

As a consequence, the driving questions became:

- Are there any seasonal patterns in these reservoirs regarding water quality parameters and phytoplankton and zooplankton structure?
- With the diversity of reservoirs (geology, morphometry, water regime), will some significant patterns become apparent with the observation of a limited number of reservoirs (16 and 26 when referring to the two major groups) - limited by budget and man-power?
- Why was fish flesh flawed by algal taint harvested in some reservoirs?
- Under which conditions are surface oxygen deficiencies encountered?
- In what general (departing from the focus of the study one could also use the term baseline) condition/ecological situation are the Western Cape Reservoirs?

- And finally: will there be differences among reservoirs with and without aquaculture production on the level of single parameters, the sediment underneath the cages and the nutrient budget?

With the following in-depth research questions:

- Are there correlations between and among single parameters/groups of parameters?
- Can related parameters be explained by each other (are there co-dependencies)?
- Are there statistically sound differences between aquaculture and non-aquaculture reservoirs (compartmentalised into water body, sediment and a phosphorus budget)?
- How can the information on the system and factors triggering changes in the system be used to support the production of rainbow trout? What are thresholds for successful production (low mortalities, average growth of fish, no algal taint etc.)?

Since hypotheses could be defined as research questions that include a predicted outcome of the studied phenomena, the author abstains from the conversion of questions into hypothetical statements. When comparing two sets of data by declaring one as the baseline dataset (reservoirs without trout production also called reference reservoirs), the driving question will remain whether there is a difference deriving from adding aquaculture to the system or not (and if yes: where these differences are manifested). All research questions, aims, methods and the resulting structure of the thesis are described in Chapters 1.2 to 1.4 and are further supported by Figure 1.1.

The fact that agricultural nutrient input has a big influence on Western Cape reservoir systems was identified very early in the study, however, the quantification of such a predominantly indirect influence (surface water runoff) is very complex. Data with regards to agricultural influences on water bodies in South Africa are scarce. The study of agricultural influence on water bodies involves the collection of data on fertiliser utilisation (fertilisers types, their application forms, quantities applied, seasonal regimes), on the variety of crops with different binding capacities and soil-root mechanisms, on different soils with different sorption capacities (geologically a very heterogenic region), on rainfall versus fertilisation timing that influence quantity and quality of runoff and on many other factors. While the in depth study of the influence of agriculture is highly recommended, it was simply not available in sufficient detail for this study. For this reason, the given situation, including the agricultural influence was set as a baseline, while other reservoirs with a similar influence by agriculture and additional aquaculture were compared to these baseline reservoirs.

Methodologically, an efficient sampling strategy and analysis of the data was necessary due to budgetary and man-power constraints. Some assumptions were made after they were sufficiently tested. For one, surface water samples (0 and 2 m depth) were collected at different locations in ten reservoirs to investigate the difference between the surface water column above the deepest area/above the cage and for instance near the outlet of the reservoir. The variances among surface samples from central locations and locations near the shoreline did not exceed the variances between replicates which is why surface water samples from the deepest water area were defined as sufficient to reflect the water quality of the epilimnion of these small reservoirs.

The second assumption refers to the sediment situation underneath the production cages. The maximum area of the hypolimnetic zone is expected to vary from 1.3 to 6.5 ha (with approx. 0.5 ha being affected by aquaculture cages of 10 by 20 m), which implies decreasing influence of cage aquaculture on the sediment situation of most reservoirs (3 to 10 ha surface area). Hence, sediment samples were collected from beneath the cages and correlations among the sediment conditions/hypolimnetic waterer (representing the hypolimnion) and surface water quality (representing the epilimnion) were valid for the size of the reservoirs studied.

1.6 References

Beveridge, M. C. M. (1984). Cage and pen fish farming. Carrying capacity models and environmental impact. FAO Fisheries Technical Paper (255). Rome, FAO.

Beveridge, M. C. M. (2004). Cage aquaculture. Oxford: Fishing News Books.

Eurofish (2008). FishInfo Network Market Reports on Salmon & Trout. http://www.eurofish.dk/. Visited 2008.

FAO (2008). Cultured Aquatic Species Information Programme - Oncorhynchus mykiss. http://www.fao.org/fishery/culturedspecies/Oncorhynchus_mykiss.

FAO (2007). State of World Fisheries and Aquaculture 2006. Rome, FAO.

Folke, C. and Kautsky, N. (1989). The role of ecosystems for a sustainable development of aquaculture. Ambio 18: 234-243.

Höll, K. (1986). Wasser. Berlin: Walter de Gruyter.

Kingfisher (1922). A Trout Fisher in South Africa. Cape Town: F.W. Flowers & Co.

Lind, O. T. (1979). Handbook of common methods in Limnology. London: C.V. Mosby Company.

McNevin, A. (2008). WWF Trout Dialogue. Washington D.C., World Wildlife Fund. http://www.worldwildlife.org/what/globalmarkets/aquaculture/troutdialogue.html.

Pillay, T. V. R. (2004). Aquaculture and the environment. Oxford: Blackwell Publishing Ltd.

Podemski, C. L. and Blanchfield, P. J. (2006). Overview of the environmental impacts of Canadian freshwater aquaculture. A Scientific Review of the Potential Environmental Effects of Aquaculture in Aquatic Ecosystems - Volume 5. Canadian Technical Report of Fisheries and Aquatic Sciences. Ontario, Department of Fisheries and Oceans Canada.

Stubbs, G. (2007). Production figures of rainbow trout in South Africa. Western Cape Trout Association Meeting, May 2007.

Wetzel, R. G. and Likens, G. E. (2000). Limnological Analyses. New York: Springer.

CHAPTER 2 REVIEW OF FRESHWATER TROUT CAGE FARMING (*ONCORHYNCHUS MYKISS*) WITH RESPECT TO WESTERN CAPE IRRIGATION RESERVOIRS

Abstract

Western Cape reservoir irrigation water can be used more efficiently when introducing trout production as an additional source of protein and income. However, cage aquaculture is known to add nutrients to water systems which need to be assimilated or removed to avoid water quality deterioration. Depending on lake ecology and hydrology, the fate of phosphorus, the key element to eutrophication in freshwater will have different effects.

Aquaculture literature indicates impact on phosphorus content, hypolimnetic oxygen and phytoplankton compositions, but the effects depend on the production scale, reservoir size and hydrology. The aim of the literature review was to determine and describe available literature on current state, possible indicators and benchmarking between sustainable and non-sustainable aquaculture.

As concluded, there are various research gaps: (i) the general status of the ecology of small Western Cape reservoirs (<15 ha); (ii) quantification of changes in the ecology of Wesern Cape irrigation reservoirs caused by aquaculture, (iii) the criteria to identify reservoirs that produce trout successfully in Western Cape irrigation reservoirs (e.g. low mortalities, good fish growth and no algal taint) are unknown.

2.1 Trout farming and the environment

The operation of trout cage farming can influence the integrity of a water body at different levels. On a greater scale, the initial status of the water body and its general value to biodiversity and ecosystem functioning could be changed (Pillay 2004). Within the water body, the system should be able to continuously deliver ecosystem services such as, for example, the assimilation or removal of aquaculture waste and the supply of sufficient oxygen (Beveridge & Stewart 1997). Artificially constructed water systems are often not dedicated to aquaculture only, but to other uses such as the supply of irrigation and drinking water, power supply or recreation (Beveridge & Stewart 1997). Aquaculture is as a result of its recent development, usually the youngest and fastest growing stakeholder of water use and therefore in the public eye is primarily responsible for maintaining water integrity (Tisdell 1999). All water bodies, including artificial ones, are inevitably linked to the general water cycle via inflow, direct outflow or groundwater in- and outflow. To avoid user conflicts, lake users could employ the lake's ecological potential by allowing the system to assimilate introduced substances and to maintain a near-natural state (Folke & Kautsky 1989). However, in order to use a water body efficiently, an understanding of the system is required.

The ecological impact of cage farming in open water systems consists of the production of waste and increased sedimentation rates, the introduction of pathogens or disruption of disease and parasite cycles and a changed aquatic flora and fauna (Podemski & Blanchfield 2006, Beveridge 2004, Pillay 2004). The release of uneaten food and faeces into the environment depends strongly on the intensity of production and also on the trophic state of the farmed species. With carnivorous fish such as trout, intensive care and feeding are necessary where feeding efficiency determines greatly which amounts of organic waste will finally enter the water body carrying the cage (Beveridge 2004, Gavine et al. 1985). With nutrient increases primary production can be stimulated and can adversely affect water quality, through for example changed and increased phytoplankton composition and biomass which might affect water transparency (Nicholls & Dillon 1978). Sedimentation accumulation below the cages can lower profundal oxygen and have an impact on the benthic community (Molot et al. 1992). The primary concerns associated with cage aquaculture is that the fish are vulnerable to external water quality conditions such as algal blooms and low oxygen concentrations (Beveridge 2004) and these conditions can be self-induced via waste introduction (Podemski & Blanchfield 2006).

All these effects have been demonstrated, yet fish farming intensity and morphometric and hydrological conditions play the decisive role with regards to the presence and degree of degradation (Beveridge 1984). In larger lakes, the mixing rate within the larger water body of an enclosed area or basin used for production plays a major role. According to Boyd et al. (2001), sites are differentiated into (i) small water bodies (<100 ha) or enclosed basins, (ii) enclosed sites of all sizes with epilimnetic, but no hypolimnetic connection to the main basin (>100 ha) and (iii) exposed sites (>100 ha) that show good mixing of the hypolimnion with the larger water body. Whilst reviewing the literature no effects of cage farming were detected only in the latter type of reservoir (Boyd et al. 2001). At smaller sites or sites of low exchange, the degrading effects of increasing phosphorus content, decreasing profundal oxygen concentration as well as alterations in phytoplankton and zooplankton assemblages were detected (Gale 1999, Axler et al. 1996, Marsden et al. 1995, Stirling & Dey 1990).

2.1.1 Fish feed and aquaculture wastes

Uneaten fish feed and fish faeces are the primary sources of nutrient input into waters during intensive aquaculture production (Podemski & Blanchfield 2006). The quality of feeding methods as well as feed composition have a great influence on the pollution potential and can be quantified by the food conversion ratio (FCR), which is the ratio between dry food added to the water and that retained as wet biomass in the fish flesh (Beveridge 2004). With a FCR approaching the optimum, which would be 1.0 with carnivorous fish completely dependent on additional feeding, the phosphorus release into the environment comprises of the phosphorus contained in the food minus the phosphorus retained in the fish. Trout flesh contains approximately 0.48 % of phosphorus (Beveridge 2004) while the majority of trout grower and grow-out feed applied in South Africa contains 1.2 % phosphorus (AquaNutro 2010). With a realistically achievable FCR of 1.2 with rainbow trout production in cages (Beveridge 2004), the amount of phosphorus released into the environment would therefore be 10.8 kg per ton of trout produced. In comparison, with a FCR of 2.0 the amount of phosphorus released into the environment would be approximately 19.2 kg per t of trout. This is approximately twice the load discarded as an optimum energy and nutrient loss. The minimum phosphorus released, even under ideal feeding conditions (FCR 1.0) would be 8.8 kg per ton of trout, about two thirds of the phosphorus in the feed (Temporetti et al. 2000). Improved trout feed formulations e.g. from the United States, were able to half this figure by reducing the phosphorus content in trout feed to concentrations of 0.7 to 0.9 % (Hinshaw 1999).

There are different pathways for the release of phosphorus not incorporated into fish biomass. A proportion of the food offered to trout in cages will always remain and settle towards the lake bottom. The reason is that fish first have to recognize that food is present, and then secondly accept it as uncontaminated and of good tactile feel to the mouth. The location of the cage should be such that feed is not carried away, e.g. by currents (Beveridge 2004). To alleviate feed loss, one can improve food acceptance via olfactorial and visual attractants to meet tactile preferences. Float and sink characteristics of the food can be improved in order to optimally meet fish behaviour. Finally, feeding strategies and quantities can be adapted to certain weather conditions. All these factors will positively influence the FCR. The proportion of uneaten feed also depends on stocking density and behavioural aspects, but can with conventional feeding be assumed to vary between 10 and 20 % at food conversion ratios reaching the optimum, and at a 1 to 5 % loss of uneaten feed when using self feeders (Geurden et al. 2005).

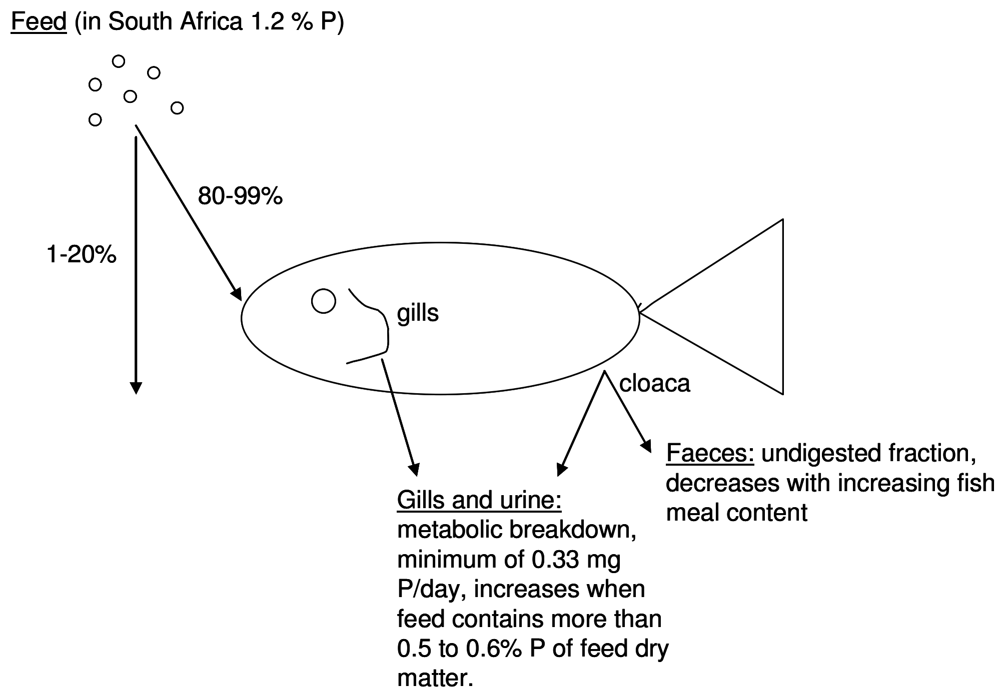

Figure 2.1: Pathways of fish feed remaining in water and wandering through fish. (The most commonly applied fish feed in South Africa was used for deduction of the average P content).

After ingestion, the fate of the food will be determined by the available gut enzymes and the food’s general digestibility. Products of digestion are absorbed into the bloodstream while the undigested fraction is rejected as faeces. Metabolic breakdown products are released via the gills and as urine. One factor influencing digestibility is the proportion of fish meal added. Diets with high concentrations of fish meal lipids and proteins are referred to as nutrient-dense, and hence they are low in carbohydrates and moderate in protein content. The declared disadvantage of fish meal based feeds is their contribution (depending on the actual source of the main component fish meal) to wild fish stock depletion instead of the relief to wild stocks promised by aquaculture production. On the other hand, the advantage of fish meal based high-energy, is higher efficiency and digestibility. Therefore, they offer a considerable reduction in feed waste, both in uneaten feed and faeces, which holds true especially for the phosphorus fraction (Adelizi et al. 1998, Kim et al. 1998).

According to Rodehutscord et al. (2000), who investigated the relationship between urinal and faecal deposition, the metabolic response to phosphorus supply in food is regulated at the intestinal level. Approximately 0.33 mg phosphorus per fish per day is inevitably lost by non-faecal excretion (gills and urinal), even if the diet contains no phosphorus. Up to the maximum retention capacity level of phosphorus (close to the phosphorus requirements of trout between 0.5 % and 0.6 % of feed dry matter), the non-faecal release of phosphorus stays stable and increases dramatically with phosphorus oversupply. From their results the authors extrapolated that the contribution of non-faecal phosphorus surplus excretion becomes a major pathway at levels of phosphorus intake higher than the maximum physiological retention capacity. This was also confirmed by the results of Bureau and Cho (1999) where 46 % of the phosphorus intake was excreted via urine when the dietary phosphorus amounted to 2.2 % phosphorus.

2.1.2 Phosphorus in surface waters

Phosphorus is applied as the principal controlling variable to describe temperate freshwater lakes (Vollenweider & Kerekes 1980) as well as subtropical and tropical lakes (Osborne 2005). The primary reason being that phosphorus is a major trigger of lake productivity in freshwater and is the most powerful and practical indicator to explain eutrophication, the process of nutrient enrichment (Schindler 1977, Dillon & Rigler 1974). In relation to the abundance of other major nutritional and structural components of the biota, such as carbon, hydrogen, nitrogen, oxygen and sulphur, phosphorus is least available in the hydrosphere, and thus commonly the limiting component of biological productivity, primarily algal growth. The natural rate of phosphorus accumulation is accelerated by human activities such as wastewater inflow (point source) or agricultural runoff (non-point source) (Bennett et al. 2001).

Salmonid cages are another significant point source for phosphorus (Beveridge 1984) which was specifically elevated in reservoirs of limited size or restricted profundal water exchange (Yan 2005). In an observational study in Scotland, Stirling and Dey (1990) reported significantly increased phosphorus levels and phytoplankton composition of a highly eutrophic lake within a 71 ha loch supporting a 200 to 300 t trout production, after aquaculture introduction. Similar observations were made in two Minnesota (USA) mine pit lakes where about 15 t of trout were produced in basins with a size of 28 ha and 56 ha, and 30 m and 65 m depths, respectively. Within five years, total phosphorus levels increased by factors of ten and twenty respectively (Axler et al. 1996). In Lake Wolsey in Canada, a 2315 ha large enclosed section of Lake Huron, approximately 2500 t of trout were produced, with a doubling of total phosphorus levels over 10 years (Hamblin & Gale 2002).

An example of low discernable phosphorus increase was in the 250 ha Lake Menteith in Scotland, where trout production was limited to 20 t per year (Marsden et al. 1995). A low impact of aquaculture was also reported from the Georgian Bay main section within Lake Huron, Canada, where a basin size of 1.5 million ha can sustainably support a production of several hundred tons of trout per year (Gale 1999). In south-western Australia, trout are successfully reared in saline farm reservoirs of less than 1 ha in size, with the reservoirs being completely flushed once a year (Gooley & Gavine 2003). Hence, to maintain the water quality required for trout production, the upper production limit is recommended to remain between 200 and 500 kg per ha of a farm reservoir (Boyd 2006, Gooley & Gavine 2003).

2.1.3 Sediments

With custom-made feed formulations and feeding efficiency, waste introduction by trout aquaculture can be minimized to an optimum of 5 kg phosphorus per t of trout in contrast to up to 40 kg phosphorus input per t with unspecific feed and feeding methods (calculation according to Beveridge (2004), using literature values presented in Chapter 2.1.1). Sediments below fish cages generally show enrichment in phosphorus, nitrogen, organic carbon, and zinc (Alpaslan & Pulatsü 2008, MacIsaac & Stockner 1995, Cornel & Whoriskey 1993, Kelly 1993, Phillips et al. 1985a, Penczak et al. 1982). Decomposition of wastes may result in hypoxia in sediments and the water column (Axler et al. 1998). Beveridge (1984) also reported that intensive fish farming lowered oxygen concentrations in certain lakes. Deoxygenation is primarily a localized effect, usually found confined to bottom waters beneath cages (Veenstra et al. 2003). In extreme conditions and with small water bodies (<60 ha), the whole water column may experience hypoxia (Axler et al. 1996). At Lake Wolsey, Canada, the hypolimnetic oxygen concentration was reduced from 6 to 9 mg/L to 0 mg/L in the profundal region between 13 and 40 m depth after aquaculture operations commenced (Hamblin & Gale 2002, Gale 1999).

The majority of phosphorus in farm wastes is lost to sediments as solids (Phillips et al. 1993) consisting of uneaten food and faeces, while the rest of the phosphorus that is not retained in the flesh will be released by excretion. Once residual feed accumulates at the bottom, its decomposition results in the release of labile phosphorus in the water column (Kelly 1995, Kelly 1992). This fraction of phosphorus at the sediment-water interface depends on water quality of the overlying water as well as the retaining capacities of the sediment, the morphometry of the basin, water renewal rates and the history and magnitude of external phosphorus loading (Boström et al. 1988).

Generally, phosphorus fractions can be permanently bound to sediments or be potentially mobile depending on the sediment and water conditions (Boström et al. 1988). Phosphorus may be discharged from the sediment as organic phosphorus or orthophosphate (PO_4^{3-}) . In actively metabolising sediment the phosphorus will diffuse into the interstitial pore water where it accumulates (Fedotov & Spivakov 2008). Further movement of phosphorus is regulated by the laws of diffusion (Reynolds & Davies 2001).

Inorganic phosphorus is found in sediments as part of a mineral or as precipitated phosphate salt (Reynolds & Davies 2001), depending on the geochemical background and mineral deposition and these fractions are primarily immobile. When phosphates are fixed as adsorbed anions on the surface or interior of metal oxides and hydroxides (primarily iron, aluminium and manganese), this fraction will be the strongest influence as a phosphorus sink or source (Wang et al. 2005, Danen-Louwerse 1993), a process controlled by pH or oxygen availability (redox potential) at the sediment-water interface. Oxygen content and pH value also control the phosphorus that will be associated with calcium, primarily as calcium carbonate (An & Li 2009). Further, the iron to calcium ratio affects pH associated phosphorus re-suspension (Huang et al. 2005). In sediments containing more iron relative to calcium, a pH value >8 will increase the phosphorus release rate and when sediments contain more calcium relative to iron and the pH value approaches 6, the phosphorus release rate will also increase (Christophoridis & Fytianos 2006). To estimate the general phosphorus binding capacity of sediments, Jensen et al. (1992) introduced the total phosphorus to iron ratio (TP:Fe). Anoxic conditions generally increase the release rate of phosphorus from the conditionally available iron oxides and hydroxides and calcium (Gonsiorczyk et al. 1997, Nürnberg 1994, Redshaw et al. 1990). The phosphorus that is only loosely adsorbed to the surface of sediment particles will additionally be controlled by sediment particle grain size (the smaller the greater the surface area) and mechanical disturbance. Very high silicate levels as well as high humic acid levels in lake sediments can also interfere negatively with the binding capacities of metals with phosphorus (de Vicente et al. 2008). However, humic material usually acts as a phosphorus sink by chelating with iron (Boström et al. 1988). Another known controlling sediment component is sulfate (Gächter & Müller 2003, Caraco et al. 1993). Under sulfate reduction, hydrogen sulfide will dissolve ferric oxyhydroxides and release iron-bound phosphorus and also deplete pore water iron through iron monosulfide (FeS) precipitation.

In its organic form, phosphorus is associated with carbon through organic phosphate esters. In decomposing algal deposits, roughly one third of cell carbon is typically labile and subject to oxic breakdown within one to two days (Jewell & McCarty 1971). With progressing carbon oxidation, a surplus of phosphorus will be released. The second third will be broken down within months while another third, primarily cellulose derivatives, could be considered inert. The decomposed fractions and time frames are similar for anaerobic decomposition of plant material (Foreel & McCarty 1970).

The majority of phosphorus in uneaten feed and faeces will be either organically bound or calcium associated and these two fractions therefore primarily regulate the phosphorus availability beneath cages (Temporetti & Pedrozo 2000). To date little is known about the bioavailability of phosphorus that leaches from aquaculture waste, but several authors have estimated the release rate of phosphorus from sediments beneath cages. Temporetti and Pedrozo (2000) reported values from 9 mg to between 1800 and 2280 mg P per m^2 per day, primarily depending on sediment age. Samples were derived from the 400 ha Argentinean Lago Moreno and the 6700 ha Alicura Reservoir. Kelly (1993) examined oxic sediments from beneath two, five and seven year old fish farms and found release rates of 0.9, 5.3 and 57.2 mg P per m^2 per day, respectively, in Scottish lochs. Boström and Pettersson (1982) found a basic release rate of 50 mg P per day per m^2 for a shallow eutrophic lake in Sweden, without aquaculture input. Without an additional aquaculture load, Rose and Robertson (2007) recorded values of between 0.3 and 6 mg P per m^2 per day for Wisconsin lakes.

Phosphorus fractions of sediments can be separated by physico-chemical properties (Rydin 2000) or by ecological importance and bioavailability (Zhou et al. 2001). To estimate environmental impact (related to enhanced algal growth by nutrient release), several authors use the categories of water soluble, readily desorbable RDP) and algal available phosphorus (AAP) (Branom & Sarkar 2004, Zhou et al. 2001). The water soluble fraction can immediately become bioavailable to primary production. RDP and AAP, however, are only potentially released fractions. RDP is released by mechanical turbulence and resuspension, whereas AAP can be accessed by change of chemical conditions and binding capacities, such as pH or redox potential (Ting & Appan 1996). The RDP would primarily correspond to loosely attached phosphorus adsorbed at the sediment particle surface. AAP would primarily coincide with the physico-chemical fractions of metal-bound, calcium associated and organically bound phosphorus (Zhou et al. 2001).

2.1.4 Bioindicators

Bioindicators are organisms that help to verify the ecological status of a water body (Government of Canada 2008). The advantage of bioindicators lies with the integration of all environmental factors, even those not necessarily known (Ellenberg et al. 1986). Additionally, cumulative effects are represented without detailed knowledge of cause and effect and thereby organisms integrate over time, balancing temporal and spatial cycles. Successfully applied indicator organisms are phytoplankton and zooplankton assemblages for the pelagic phase, as well as benthic organisms for sediment health (Conti 2008, Cottingham & Carpenter 1998, Canfield & Jones 1996, Bays & Crisman 1983).

Aquaculture can increase soluble phosphorus in the water column (dependent on stocking density and reservoirs size), and a phosphorus increase can enhance algal blooms (Diaz-Pardo et al. 2007, Baldwin et al. 2003, Diaz et al. 2001). Phosphorus is often the first descriptor of eutrophication in freshwater. However to assist interpretation of the actual phosphorus content, many other parameters can be used, such as surface and bottom oxygen levels, water temperature and climate and importantly the chlorophyll a levels or biomass of phytoplankton, or when it comes to interpretation of species composition the ratio of nitrogen to phosphorus (Wetzel 2001). The relationship between phosphorus and chlorophyll concentration has been well described for temperate systems (Kalff 2002, Straskraba et al. 1993, Vollenweider 1968), but less established for subtropical and tropical systems (Huszar et al. 2006, Kalff 2002, Salas & Martino 1991). Under warmer conditions, there seems to be a more variable relationship between the total phosphorus and chlorophyll a content, and also a lower chlorophyll a yield per unit total phosphorus (Huszar et al. 2006).

Longgen and Zhongije (2003) showed that cage aquaculture impacts on phytoplankton density. They found a decrease in chlorophyll a content from within the cage to 130 m away in a 35 ha sized bay off Niushanghu Lake in China, with an average water depth of 2 m. In Lake Alicura, Argentina, with a mean depth of 46 m in a 6700 ha sized reservoir, a study before and after introduction of a 150 t trout production unit, showed an increase in nutrient concentrations, algal density and phytobiomass as well as changes in species composition. However, these effects were very localized around the cage farm (Diaz et al. 2001). Typical dominant species found represented enriched waters, including *Anabaena spiroides*, *Staurastrum sp.*, *Elakatothrix sp.* and *Ceratium hirundinella* (Reynolds & Irish 1997, Reynolds 1992). In a shallow 71 ha loch in Scotland supporting 200 to 300 t of trout production, Stirling and Dey (1990) found that *Microcystis aeruginosa* dominated phytoplankton characterizing highly eutrophic conditions. Many studies show the link between the shift in phytoplankton species and eutrophication status (Diaz-Pardo et al. 2007, Baldwin et al. 2003, Bettinetti et al. 2000, Dasi et al. 1998, Huszar et al. 1998, Reynolds 1998, Jensen et al. 1994) and why they can be more informative than classic water chemistry values (Huszar & Caraco 1998). A study by du Plessis (2007) in South African farm reservoirs showed significantly higher phytoplankton biomass in production reservoirs versus reference reservoirs during the production period. Dominant species consisted of *C.*

hirundinella, *Anabaena sp*., *Microcystis sp*. as well as cryptophytes with *C. hirundinella* and *Peridinium sp*. dominating in sites without fish production.

Zooplankton has been shown to be a valid indicator for trophic status (Wang et al. 2007, Canfield & Jones 1996, Bays & Crisman 1983). The advantage of zooplankton assemblages can be its delay in responding to environmental changes in contrast to phytoplankton (conserve temporal variations longer) and its power to explain phytoplankton composition in more detail (Hunt & Matveev 2005). Apart from food availability, water temperature and characteristics, zooplankton is regulated by predators such as invertebrates and planktivorous fish (Sommer 1993). Hence, whether aquatic ecosystems are perturbed by changes in the top predator fish abundance or by nutrients as stressors, zooplankton is a sensitive integrator of these changes (McNaught and Buzzard 1973).

Protozoans, rotifers, copepods and cladocerans are varying groups of zooplankton with different food preferences and top down regulators. Rotifers were the best suited group to describe trophic state changes (Eeirans 2007). Their abundance was higher in waters of higher trophic state, certain species were missing completely and others dominated such as *Keratella cochlearis*, *Brachionus sp*., *Filinia longiseta* and *Pompholyx sulcata* (Berzinš & Pejler 1989, Maemets 1983, Hakkari 1972). In a shallow lake in China practising fish aquaculture, the number of rotifers near the cage increased while the number of cladocerans followed the opposite trend (Longgen & Zhongije 2003).

A few freshwater studies have shown that benthic communities are impacted by aquaculture (Dobrowolski 1988). Alpaslan and Pulatsü (2008) and Karaca and Pulatsü (2003) show a decrease in species diversity and an increase in species abundance beneath a 30 t production cage within the 650 ha Kesikköprü Reservoir in Turkey. The predominantly impacted species were resistant to sedimentation and low oxygen availability (Johnson et al. 1993). At freshwater sites, which are usually less hydrologically disturbed than marine sites (no tidal fluctuations), impacts on benthic fauna were rarely observed >25 m from the cage sites (Weston et al. 1996, Phillips et al. 1985b). However, in smaller lochs in Scotland (<70 ha), significant alterations of benthic communities were still apparent beneath cage sites that were vacated three years previously (Doughty & McPhail 1995).

2.2 Trout cage farming in the Western Cape, South Africa

Cage aquaculture, especially of species such as trout, relies on available freshwater resources, primarily rivers and open water bodies (Beveridge 2004). 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 with a highly seasonal and unpredictable distribution (Davies & Day, 1998). The eastern and southern regions of the country have the highest rainfall. In the east, the climate is subtropical with annual or summer rainfall and mild winters whereas the south has a Mediterranean climate with winter rainfall and hot, dry summers (Goldblatt 1997). Both areas share the distribution pattern of alternating rainfall and a dry season which makes the country dependent on water storage. More than 60 % of the mean annual runoff of about 49,000 million m^3 per year is caught and stored in 320 major dams (DEAT 2006; Allanson 2004). About 20 % of runoff is released into river ecosystems to allow their continued functioning (DEAT 2006). In addition to these large reservoirs, many farms have small reservoirs providing water for irrigation, livestock watering and human consumption. Small reservoirs (<100 ha) are predominantly in KwaZulu-Natal and the Western Cape Province.

In the Western Cape Province, these small storage reservoirs are primarily used for irrigation during the dry summer months. In 1994, over 4000 farm reservoirs with a total storage volume greater than 100 million m^3 and a dam wall higher than 5 m were registered (Berg et al. 1994). Surface areas vary between 1 and 15 ha. Due to its non-consumptive water use, cage aquaculture can add to the water productivity of these reservoirs (which means the volume of production per unit of water). The region in the north and north-east rarely shows water temperatures beneath 20 °C and is primarily suitable for warm water fish with the exception of highland areas such as Lesotho and the Drakensberg region (Pike 2009). The situation is different with regards to water temperatures in the south-west of the country where average winter water temperatures are around 10 °C. In the Western Cape, warm water species can not tolerate the winter conditions, but temperate fish species can be supported throughout the winter (summers again too warm) or with spring water access even throughout the year (Maleri et al. 2008).

Trout was introduced into Western Cape river systems more than 100 years ago and the debate as to whether the species should be supported any further continues between governmental and research ecologists (Woodford & Impson 2004, Cambray 2003, Skelton 2002, Davies 2002). The current modus vivendi by Cape Nature, the institution representing the Western Cape Nature Conservation Board, is to allow trout rearing activities and the species' release into standing waters within water management areas and river systems that have already been disturbed by trout introduction. The application procedure for 5 to 10 t production units has also been simplified. Trout hatcheries have a long history and years of experience in the area. Most producers of trout buy juvenile fish from the hatcheries in late autumn (April to May) and grow them in cages or raceway systems from juvenile sizes varying between 100 and 250 g, to market size of 900 to 1300 g, during the winter season when water is abundant and temperatures are suitable for trout production. However, in most reservoirs, the temperature profile and range in winter diverges from the optimal trout rearing temperature of 18 °C most of the time (Maleri et al. 2008). Other water quality factors critical for trout rearing are oxygen levels above 7 mg/L at all times, low ammonia levels and absence of cyanophytes releasing off-flavour substances which are easily absorbed by trout flesh.

2.3 The ecology of small reservoirs in the Western Cape

South Africa is highly dependent upon reservoir water and as a country is struggling to balance supply and demand even in the water-richer regions (DEAT 2006). Data on water quality of surface water in South Africa are primarily restricted to large reservoirs, the main supply of drinking water, and river systems (Hart & Hart 2006). Limnological research in privately owned farm reservoirs, mainly used for irrigation of private land, has been left unexploited (Hart & Hart 2006).

The threat of eutrophying surface waters was recognized fairly early and research during the 70's and 80's addressed the extent of eutrophication within major reservoirs in South Africa (Grobler & Silberbauer 1985, Steyn et al. 1976, Toerien et al. 1975). Toerien et al. (1975) found that most impoundments in the Western Cape Province, especially in the mountainous areas, are of low trophic state, whereas some enrichment was detectable in the Klein and Groot Karoo areas. An examination of 64 man-made reservoirs in Southern Africa, only representing the north-east of South Africa, found that 75 % of the reservoirs in the region could be regarded as enriched and 10 % as hypereutrophic (Thornton 1987).

Most South African water bodies are warm monomictic which means that the overturn of the water occurs once a year (Hart & Hart 2006). With strong wind action and horizontal inflow establishing in late summer and autumn, stratification is often disturbed. With the cooling of surface water temperatures in autumn, the temperatures of the epilimnion and hypolimnion assimilate and density differences can no longer prevent the mixing of the water body. The extent to which bottom water becomes mixed into the surface water mass depends on the strength of wind activity, the shape of the water basin and the overall basin depth (Sommer 1993). Throughout autumn and winter, the water circulation can be maintained until the surface water is again warmed by spring and summer conditions. The mixing pattern triggers the abundance of algae which is highest in the autumn and winter period, when most nutrients are abundant in the water body being recirculated from the sediment (Pieterse & van Zyl 1988). Due to water temperature requirements of trout and their grow-out period during winter, the addition of extra nutrients by aquaculture feeds and wastes concurs with the natural cycle of internal nutrient availability and highest phytoplankton biomass.

Depending on the presence of organic material or dead algal biomass at the lake bottom, oxygen depletion by decomposition is more or less prominent. Toerien et al. (1975) use oxygen depletion in the hypolimnion during the summer stagnation as an indicator of trophic status and see it as an imbalance between photosynthetic and respiration activities. They did not measure chemical or nutrient concentrations as a reference for trophic state, but compared algal growth potential of epilimnetic water with oxygen profiles measured by Schutte and Bosman (1973). They found absence/presence and extension of anaerobic zones to relate to the extension of algal growth in sampled and filtered lake water, hence a method to estimate algal available nutrients. Due to the relatively high mean water temperature in subtropical waters, bacterial growth and activity and deoxygenation are rapid processes, especially in nutrient and algal rich systems (Allanson 1995). Deoxygenation is more likely in these systems than in temperate climates where most literature on lake ecology was produced. Hart & Hart (2006) suggested that many South African reservoirs of low nutrient status show considerable oxygen depletion in the hypolimnion which was also found to be the case in Australia (Townsend 1995). However, the hypolimnetic water temperatures were between 26 and 30 °C, with temperatures not as elevated in the Western Cape where bottom water temperatures, even of shallow impoundments (5 to 6 m depth) remained below 23 °C in summer (Du Plessis 2007). Townsend (1999) also suggests that a trophic classification of reservoirs based on hypolimnetic deoxygenation would not be globally applicable, however it is a useful lake descriptor on a regional scale when hypolimnetic water temperatures are similar and only morphometric influences have to be considered.

The water level of most irrigation reservoirs in South Africa is not only controlled by in- and outflows, rainfall and evaporation, but also by intensive water extraction during the summer period. Reservoir volumes may be reduced by 80 % or more during this time period, but are sometimes kept at maximum volume in other years. Only during the rainy season, from April to September, will rivers carry water again, since most rivers in South Africa which supply the reservoirs are non-perennial or of low water volume during summer. The water exchange rate will depend on the actual volume of water that is diverted to the reservoir as runoff water, direct river through-flow, channelled river water or via pipe from other reservoirs. The development of a thermocline depends on the climatic conditions as well as the overall depth of the reservoir and its morphometric conditions (Sommer 1993).

The nutrient balance of a reservoir can be very different if water has left the reservoir via an overflow during winter or if it is simply filled to capacity and additional water channelled elsewhere (Hart & Hart 2006). Nutrients are trapped in the sediment and hypolimnion during summer, but get re-suspended in winter. If mixed water leaves the reservoir during winter via an overflow, excessive nutrients can leave the system. Reservoirs with no overflow will therefore enrich faster. The effect of water exchange also supposedly depends on the hydrology of the basin and if inflowing and outflowing water are located close to each other or at opposite ends of the reservoir and if there is a difference between top or bottom water extraction during the summer months for subsequent nutrient or pollutant removal. Many farmers start with bottom water extraction and when filters become repeatedly blocked due to deteriorating water quality, they usually continue with surface water extraction, diminishing nutrient and pollutant removal. Most South African reservoirs have a water exchange rate (inflowing water volume divided by full supply volume) of one year or more.

2.4 Descriptors for ecological status and its change

To define ecological change, indicators and baseline information of original ecological status are necessary (Government of Canada 2008). Change will then be the difference in these predictors (indicators) which can be deterioration of water quality, eutrophication with limnological research or improvement, which can be referred to as recovery. Spatial comparisons are possible between reservoirs, however no two sites will share exactly the same history, conditions and circumstances which can become a problem. Still, the understanding and comparison of two or several reservoirs enables the cautious transfer of results to similar systems. When aiming to avoid differences among reservoirs, temporal comparisons can be made between the conditions prior to an event and after an event within the same water body. However, differences in weather conditions and other unforeseen changes can disrupt the coherency of the results.

Official decrees on how to describe the ecological status of water bodies are legal documents such as the South African Water Quality Guidelines (DWAF 1996a, DWAF 1996b) as part of the National Water Act No. 36 of 1998 and the National Environmental Management Act: Biodiversity Act No. 10 of 2004 and its amendments. Australia and New Zealand have developed a National Water Quality Management Strategy (e.g. ANZECC and ARMCANZ 2000, Davies 2000) which is helpful when comparing this to the South African guidelines, as well as US American and Canadian documents (e.g. Podemski & Blanchfield 2006, Boyd et al. 2001), and additionally there are recommendations by the scientific community which are communicated via publications (e.g. Ellenberg et al. 1986, Vollenweider 1980).

2.4.1 Water quality parameters as indicators of water quality

Phosphorus is the key freshwater quality parameter to estimate trophic status and it has therefore found its way into many legally binding regulations as a quantitative indicator. For South Africa, the Water Quality Guideline for Aquatic Ecosystems (DWAF 1996b) proposes that no changes of trophic state should occur, and within the course of a year the soluble reactive phosphorus content should not be changed by more than 15 % unimpacted conditions at any time. Oligotrophic conditions refer to soluble reactive phosphorus (as P) concentrations between 0 and 0.005 mg/L, between 0.005 and 0.025 mg/L for mesotrophic waters, between 0.025 and 0.25 mg/L for eutrophic conditions and for hypereutrophic conditions phosphorus levels are >0.25 mg/L. Within the aquaculture guidelines, waters with 0.1 mg/L soluble reactive phosphorus (as P) are proposed as target rearing conditions where no change in trophic status is likely to occur due to aquaculture and most fish species would be comfortable. The Australian Water Quality Guideline (Chapter 3, ANZECC & ARMCANZ 2000), works with key and trigger values which are defined as 0.01 mg/L total phosphorus as P for south-western Australia, as well as 0.005 mg/L soluble reactive phosphorus as P for that area. If these indicator values are exceeded, management actions are recommended from a precautionary principle to ensure that ecological damage is avoided. As a threshold value for algal bloom development, James and Havens (1996) found 0.1 mg/L total phosphorus as P which when exceeded in certain sections of a large subtropical lake, resulted in a 95 % chance of an algal bloom.

Other parameters used in water quality estimations are, for example ammonia, as an indicator of the presence of nitrogen in a toxic form, pH and oxygen (Wetzel 2001, Beveridge 2004). High ammonia concentrations only occur with a large volume of ammonia inflow or when anoxic conditions favour the conversion of inorganically present nitrogen into ammonia. If ammonia is not directly entering the lake, it will accumulate in the hypolimnion under anoxic conditions, depending on the availability of nitrates. When present, total ammonia occurs as a balance between ammonium ions (unionized form) and free ammonia of which only the latter may be harmful to organisms. The ratio between the two in water depends primarily on temperature and pH. The South African Water Quality Guidelines for Aquatic Ecosystems (DWAF 1996b) recommend levels of <0.007 mg/L free ammonia as N as a target range. The chronic effect value lies at 0.015 mg/L. The aquaculture guidelines suggest a target water quality range of <0.020 mg/L free ammonia as N (they suggest 0.025 mg/L as NH_3). The Australian Guidelines (ANZECC & ARMCANZ 2000) suggest levels <0.01 mg/L ionized ammonia as N, which would in most circumstances (pH<8 and water temperature ≤25 °C) mean <0.001 mg/L free ammonia as N.

Another important parameter to describe water quality and system health, is pH (Wetzel 2001). The pH is determined by geological and atmospheric influences. Depending on the buffer capacity of the water body, e.g. by silicate rich sediments and the surrounding vegetation, the pH can drop as low as 3.9 in some areas of the Western Cape dominated by fynbos with the influence of humic and fulvic acids (DWAF 1996b). The pH is diurnally and seasonally influenced by primary production, releasing or incorporating hydrogen ions in the process of photosynthetic carbon dioxide fixation or release (Whitney 1942). This pH fluctuation is therefore enhanced in poorly buffered waters (of low alkalinity). With excessive algal production showing with extreme eutrophication, the pH can rise to levels >10 and can serve as an indicator for excessive blooms (DWAF 1996b). The pH can also vary seasonally depending on the drainage system and its vegetation (such as in fynbos areas where the presence of acids is reduced during the rainy season). The South African Guideline (DWAF 1996b) suggests that the daily fluctuation of pH should be less than a unit of 0.5 or 5 % of the pH value. According to the Aquaculture Guideline, the ideal water quality range lies between a pH of 6.5 and 9.0 (DWAF 1996a).

Surface dissolved oxygen in waters used for trout production, is recommended to remain between 80 to 120 % saturation, which is dependent on water temperature and would therefore vary between 11 mg/L at 10 °C and 8 mg/L at 30 °C in Western Cape reservoirs (Maleri et al. 2008). In a seven day period, the mean minimum should not drop below 6 mg/L nor the absolute minimum be less than 4 mg/L during the course of one day (DWAF 1996a). The oxygen dissolved in water is primarily regulated by phytoplankton production and respiration with the lowest oxygen level around 0600 hours. Aquaculture guidelines recommend an oxygen content necessary for fish health, which for trout would be oxygen concentrations of a minimum of 6 mg/L at all times within the confinement of the net to a depth of 4 m.

Carlson (1977) developed a combined indicator index, the trophic state index (TSI), based on Secchi depth, chlorophyll a levels and the total phosphorus concentration. Carlson uses the July and August levels of chlorophyll a and the Secchi measurements, as well as the highest total phosphorus level measured during spring based on the phases in a dimictic, temperate lake in the Northern Hemisphsere. Despite this fact, the Carlson index has been widely applied and accepted under different climatic conditions (Felsing & Glencross 2004). In Western Cape reservoirs the typical season for phytoplankton growth would be winter and despite the difference in climate, the typical phytoplankton composition would establish between June and August (Du Plessis 2007). In lakes in which the Secchi depth is not determined primarily by algal biomass but rather by inorganic turbidity, one must be aware that the three TSI values show dissimilarities (Jayasinghe et al. 2005, Felsing & Glencross 2004). The measurement of suspended solids can be used as another parameter in such systems. There is some agreement that the Carlson Index can be reliably applied in all climates, while more refined versions of the Carlson Index that build averages among the three TSI values such as e.g. Osgood (1982), would be applicable in temperate regions only (Havens 2000). The TSI value as calculated by Carlson (1977) is designed to range between 0 and 100, with lower numbers representing lower overall productivity.

2.4.2 Bioindicators to identify water quality

Algae (phytoplankton, periphyton, macroalgae), macrophytes, zoobenthos or macroinvertebrates, zooplankton and fishes are primarily the applied bioindicators in lake ecology. When several water bodies need to be compared, the chosen indicator taxa should fulfil most criteria of a good indicator organism, such as being sensitive to the stressor, relevant to the ecosystem, timely and cost-effective and at best anticipatory. Moreover, the chosen taxa should occur at all sites (Nixdorf et al. 2000).

Phytoplankton monitoring is often routinely chosen and many factors effecting its seasonal and daily fluctuations are known and fairly easy to determine. Several studies have used phytoplankton to determine if aquaculture has an effect on overall lake ecology (Longgen & Zhongije 2003, Diaz et al. 2001, Stirling & Dey 1990). In these studies, overall biomass measured as chlorophyll a content was used, as well as a more detailed study of species present and biomass was estimated via biometrical determination. The latter analysis allows further insight by knowledge of community structures. Dominant species and species abundance and diversity provide even more substantial information on eutrophication status than biomass alone. Dominant species typical for certain trophic states have been described by authors such as Reynolds (1998, 1997, 1992), Jensen et al. (1994) and Bettinetti et al. (2000). The Shannon-Weaver index, a formula based on the sum of single species occurrence as a ratio of overall phytoplankton abundance, helps to understand the diversity of a biological population at any one time (Shannon & Weaver 1949).

A factor influencing phytoplankton composition is the availability of nutrients, such as phosphorus, nitrogen and silicate (Bohle 1995). The ratio between Chlorophyll a content and total phosphorus (Chl a:TP) is a useful tool to explain the seasonal biomass change of phytoplankton and how strongly it is influenced by phosphorus (White 1989). Deviations from the classical dominant role of phytoplankton were explained by the total nitrogen to total phosphorus ratio (TN:TP) since in some cases nitrogen becomes limited and the limitation controls phytoplankton presence rather than phosphorus levels. Also, in some cases, suspended solids control the phytoplankton abundance or a high water exchange rate of the basin (Kamenir et al. 2007, Kaweekityota et al. 2007). Silicate as an important structural component of diatoms helps to verify if their absence would be caused by silicate scarcity or a species shift by organisms with advantages under high nutrient conditions (Bohle 1995).

Knowledge of zooplankton composition and abundance will serve as an additional indicator to identify the trophic status (Bays and Crisman 1983), especially in regions where a phytoplankton indicator system has not been established or adapted to local conditions.

2.4.3 Sediment and hypolimnetic water quality

As discussed, hypolimnetic conditions strongly influence the amount of phosphorus that can be released into the bottom water (Reynolds & Davies 2001, Boström et al. 1988). Sediment nutrient accumulation is an indication of enrichment history and helps to estimate the internal loading capacities, and therefore recovery times of a water body. Total phosphorus levels as well as conditionally released phosphorus fractions are good indicators of present and future trends and seasonal changes, especially if surrounding sediment conditions such as temperature, pH and oxygen levels are known (Zhou et al. 2001).

Generally, hypolimnetic conditions are most helpful in determining water quality particularly related to cage production where remnants of the production process end as a sedimentary layer (Podemski & Blanchfield 2006). Hypolimnetic water quality has regularly been studied in scientific publications, but has rarely found its way into official regulations. Depending on the water exchange rate of the basin, oxygen consumption caused by decomposition will be the prime effect by cage production with resulting hypoxia (Axler et al. 1996, Cornel & Whoriskey 1993, Dobrowolski 1988). In lakes with an already established hypoxia, this process will be enhanced and accelerated with the consequences of ammonia accumulation in the hypolimnion and additional phosphorus release. The presence of free ammonia in contrast to nitrate and oxygen content of bottom waters is therefore a good indicator of water quality changes. While the ammonia concentration can easily be quantified, the expansion period and duration of hypolimnetic oxygen depletion changes is more problematical as a global indicator, however, applicable within a system of similar water bodies (Townsend 1999).

Many publications have concentrated on macrozoobenthos communities as a bioindicator of sediment quality and hypolimnetic conditions, in particular chironomids which can be applied as dependable indicators of medium and long term trophic status. In particular because oxygen concentrations at the lake bottom influences benthic life (Ruse 2002, Little & Smol 2001).

2.5 Descriptors of sustainable aquaculture

Sustainable aquaculture requires a sustainable water quality. To apply the precautionary principle, it should be estimated how many fish a system can support without risking water quality deterioration (Beveridge 1984). Many factors play a role here, such as basin morphometry and hydrology and more specifically the water exchange rate, the presence of a thermocline and the lake ecology and nutrient history. To estimate the environmental carrying capacity of a water body, Beveridge (1984) developed a carrying capacity model based on assumptions and conclusions introduced by Vollenweider (1968) and Dillon and Rigler (1975, 1974). However, Beveridge (2004) propagates cautiousness with the model and refers to it as a guide rather than a strict prediction. A sound knowledge of lake ecology and processes are recommended to be able to interpret, verify and refine the outcome. The model uses phosphorus content as well as morphometric and production specific input data. Davies (2000) investigated the application of the model for Tasmanian lakes and concluded that the principles behind the mass-balance approach introduced by Beveridge (1984) and Kelly (1995) could be readily adapted to the conditions of Australia’s southern island and possibly any other area if the underlying principles are thoroughly scrutinised.

Phytoplankton data are not included in these models, but algal presence and biomass plays a crucial role in salmonid rearing. Algal blooms can cause oxygen depletion in surface waters and therefore cause physical stress to fish. Certain species of blue-green algae and actinobacteria, release substances that cause unfavoured muddy-musty flavours in the fish flesh, primarily MIB (2-methylisoborneol) and geosmin (Klausen et al. 2005, Izaguirre & Taylor 2004). The prediction of algal blooms and certain species compositions follows some global principles, such as the general dependence on nutrient presence, but also underlies unpredictable and regional factors. Beginning with certain phosphorus levels, the risk of an algal bloom or cyanophyte occurrence will increase. Additionally, certain weather conditions create conditions allowing otherwise more disadvantaged species to proliferate as well as sudden nutrient access by intermediate mixing events or other situations. Elaborate models for phytoplankton prediction have been designed for specific water bodies (Elliott & Thackerey 2004, Lewis et al. 2002, Elliott et al. 2000), however, with limitations in transfer and practicability. The more detailed phytoplankton communities are described within lakes of a certain region, together with the associated physical and chemical water properties, the more predictable phytoplankton related problems become. Combining phytoplankton and water quality data for an area via regression models was found useful to predict algal blooms and the presence of certain phytoplankton classes (Shuck 2005, Yan et al. 2004).

2.6 Conclusions

There is a general shortage of whole lake research studying the impacts of aquaculture that tries to incorporate water quality, phytoplankton, zooplankton and sediment data and that considers nutrient sinks within the system (Podemski & Blanchfield 2006). Specifically, there is little information on the water ecology of farm reservoirs in the Western Cape region of South Africa and their trophic status (with most reservoirs being surrounded by agricultural land), nor the effects of trout cage aquaculture, even though the industry has been operating in farm reservoirs for the last 20 years and is rapidly expanding (Hart & Hart 2006).

The most important sources of waste with cage operations is the composition of fish feed and the feeding methods. Locally used specialised trout feed in South Africa contains about 35 % more phosphorus than comparable fish meal based feed formulations in the United States and has scope for improvement. Feeding methods need to strive for FCR's approaching 1.0. When feed formulation and feeding achievement in the form of the FCR's are known, the introduction of phosphorus into the water body can be calculated relatively accurately (since environmental research can neglect the relative proportion of uneaten feed, excreted nutrients and faeces). The sink of these nutrients will vary from lake to lake and can be established with physico-chemical indicators as well as bioindicators. The nutrient retaining capacity within the sediment of Western Cape farm reservoirs is unknown as is the internal loading capacity of these reservoirs during the winter mixing period. The production of trout and therefore waste introduction concurs with that period which could enhance phytoplankton production. Farm reservoirs in South Africa are relatively small in contrast to other sites where the impacts of trout cage aquaculture have been studied. Consequently, production units of 5 t will almost inevitably lead to accelerated hypolimnetic deoxygenation as well as to increased phytoplankton biomass with the respective consequences in those reservoirs, dependent on specific hydrodynamic conditions.

2.7 References

Adelizi, P. D., Rosati, R. R., Warner, K., Wu, Y. V., Muench, T. R., White, M. R., and Brown, P. B. (1998). Evaluation of fish-meal free diets for rainbow trout, *Oncorhynchus mykiss*. Aquaculture Nutrition 4(4):255-262.

Allanson, B. R. (1995). An introduction to the management of inland water ecosystems in South Africa. Water Research Commission Report TT 72/95. Pretoria, Department of Water Affairs and Forestry (DWAF).

Alpaslan, A. and Pulatsü, S. (2008). The Effect of Rainbow Trout (*Oncorhynchus mykiss* Walbaum, 1792) Cage Culture on Sediment Quality in Kesikköprü Reservoir, Turkey. Turkish Journal of Fisheries and Aquatic Sciences 8: 65-70.

An, W. C. and Li, X. M. (2009). Phosphate adsorption characteristics at the sediment-water interface and phosphorus fractions in Nansi Lake, China, and its main inflow rivers. Environmental Monitoring and Assessment 148(1-4): 173-184.

ANZECC and ARMCANZ (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality. National Water Quality Management Strategy Paper No 4. Canberra, Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand.

AquaNutro (Pty) Ltd. (2010). Near-infrared spectroscopy results for phosphorus content for various trout formulations. Personal communication Lourens de Wet, Malmesbury, South Africa.

Axler, R. P., Larsen, C., Tikkanen, C., McDonald, M., Yokom, S., and Aas, P. (1996). Water quality issues associated with aquaculture: A case study in mine pit lakes. Water environment research 68(6): 995-1011.

Axler, R. P., Yokom, S., Tikkanen, C., McDonald, M. E., Runke, H., Wilcox, D., and Cady, B. (1998). Restoration of a Mine Pit Lake from Aquaculture Nutrient Enrichment. Restoration Ecology 6(1): 1-19.

Baldwin, D. S., Whittington, J., and Oliver, R. (2003). Temporal variability of dissolved P speciation in a eutrophic reservoir - implications for predicating algal growth. Water Research 37: 4595-4598.

Bays, J. S. and Crisman, T. L. (1983). Zooplankton and Trophic State Relationships in Florida Lakes. Canadian Journal of Fisheries and Aquatic Sciences 40(1813): 1819.

Bennett, E.M., Carpenter, S.R., and Caraco, N.F. (2001). Human Impact on Erodable Phosphorus and Eutrophication: A Global Perspective. BioScience 51(3): 227-234.

Berg, R., Thompson, R., Little, P. R., and Görgens, A. H. (1994). Evaluation of farm dam area-height-capacity relationships required for basin-scale hydrological catchment modelling. Water SA 20(4): 265-272.

Berzinš, B., and Pejler, B. (1989). Rotifer occurrence and trophic degree. Hydrobiologia 182: 171-180.

Bettinetti, R., Morabito, G., and Provini, A. (2000). Phytoplankton assemblage structure and dynamics as indicator of the recent trophic and biological evolution in the western basin of Lake Como (N. Italy). Hydrobiologia 435(1-3): 177-190.

Beveridge, M. C. M. (1984). Cage and pen fish farming. Carrying capacity models and environmental impact. FAO Fisheries Technical Paper (255). Rome, FAO.

Beveridge, M. C. M. (2004). Cage aquaculture. Oxford: Fishing News Books.

Beveridge, M. C. M. and Stewart, J. A (1997). Cage Culture: Limitations in Lakes and Reservoirs. Inland Fishery Enhancement. FAO Fisheries Technial Paper 374. Dhaka, Bangladesh, FAO.

Bohle, H. W. (1995). Limnische Systeme. Berlin: Springer-Verlag.

Boström, B., Andersen, S., Fleischer, S., and Jansson, M. (1988). Exchange of phosphorus across the sediment-water interface. Hydrobiologia 170: 229-244.

Boström, B. and Pettersson, K. (1982). Different patterns of phosphorus release from lake sediments in laboratory experiments. Hydrobiologia 91/92(1): 415-429.

Boyd, D., Wilson, M., and Howell, T. (2001). Recommendations for Operational Water Quality Monitoring at Cage Culture Aquaculture Operations Environmental Monitoring and Reporting

Branch, Ontario Ministry of Environment. Report to the Ontario Ministry of the Environment. Toronto, Canada, Ministry of the Environment.

Boyd, S. (2006). Fish in farm dams. Primefact 89(3): 1-6.

Branom, J. R. and Sarkar, D. (2004). Phosphorus bioavailability in sediments of a sludge-disposal lake. Geosciences 11(1): 42-52.

Bureau, D. P. and Cho, C. Y. (1999). Phosphorus utilization by rainbow trout (*Oncorhynchus mykiss*): estimation of dissolved phosphorus waste output. Aquaculture 179(1-4): 127-140.

Cambray, J.A. (2003). The global impact of alien trout species - a review; with reference to their impact in South Africa. African Journal of Aquatic Science 28(1):61-67.

Canfield, T. J. and Jones, J. R. (1996). Zooplankton Abundance, Biomass, and Size-Distribution in Selected Midwestern Waterbodies and Relation with Trophic State. Journal of Freshwater Ecology 11(2): 171-191.

Caraco, N. F., Cole, J. J., and Likens, G. E. (1993). Sulfate control of phosphorus availability in lakes. Hydrobiologia 253:275-280.

Carlson, R. E. (1977). A trophic state index for lakes. Limnology and Oceanography 22: 361-369.

Christophoridis, C. and Fytianos, K. (2006). Conditions Affecting the Release of Phosphorus from Surface Lake Sediments. Journal of Environmental Quality 35(4): 1181-1192.

Conti, M.E. (2008). Biological Monitoring: Theory and Applications - Bioindicators and Biomarkers for Environmental Quality and Human Exposure Assessment. WIT Press, Boston.

Cornel, G. E. and Whoriskey, F. G. (1993). The effects of rainbow trout (Oncorhynchus mykiss) cage culture on the water quality, zooplankton, benthos and sediments of Lac du Passage, Quebec. Aquaculture 109: 101-117.

Cottingham, K.C. and Carpenter, S.R. (1998). Population, community, and ecosystem variates as ecological indicators: Phytoplankton responses to whole-lake enrichment. Ecological Applications 8: 508-530.

Danen-Louwerse, H., Lijklema, L., and Coenraats, M. (1993). Iron content of sediment and phosphate adsorption properties. Hydrobiologia 253(1-3): 311-317.

Dasi, M. J., Miracle, M. R., Camacho, A., Soria, J. M., and Vicente, E. (1998). Summer phytoplankton assemblages across trophic gradients in hard-water reservoirs. Hydrobiologia 369/370: 27-43.

Davies, B. and Day, J. A. (1998). Vanishing waters. UCT Press, South Africa.

Davies, B. (2002). Trout: Conservationists and stakeholders speak out. www.scienceinafrica.co.za/2002/may/trout.htm.

Davies, P. E. (2000). Cage Culture of Salmonids in Lakes: Best practice and risk management for Tasmania. Report to Minister for Inland Fisheries and Inland Fisheries Service. Hobart and Lanceston, Tasmania, Department of Primary Industries and Water.

DEAT (2006). South Africa Environment Outlook. A report on the state of the environment. Pretoria, Department of Environmental Affairs and Tourism.

De Vicente, I., Jensen, H.S., and Andersen, F.O. (2008). Factors affecting phosphate adsorption to aluminum in lake water: Implications for lake restoration. Science of the Total Environment 389(1): 29-36.

Diaz-Pardo, E., Vazquezb, G., and Lopez-Lopeza, E. (2007). The phytoplankton community as a bioindicator of health conditions of Atezca Lake, Mexico. Aquatic Ecosystem Health and Management 1(3-4): 257-266.

Diaz, M. M., Temporetti, P. F., and Pedrozo, F. L. (2001). Response of phytoplankton to enrichment from cage fish farm waste in Alicura Reservoir (Patagonia, Argentina). Lakes & Reservoirs: Research and Management 6: 151-158.

Dillon, P. J. and Rigler, F. H. (1974). A Test of a Simple Nutrient Budget Model Predicting the Phosphorus Concentration in Lake Water. Journal of the Fisheries Research Board of Canada 31: 1771-1778.

Dillon, P. J. and Rigler, F. H. (1975). A Simple Method for Predicting the Capacity of a Lake for Development Based on Lake Trophic Status. Journal of the Fisheries Research Board of Canada 32(9): 1519-1531.

Dobrowolski, Z. (1988). The effect of cage aquaculture of rainbow trout on the distribution and stability of macrobenthos in eutrophic Lake Letowskie. Polish Journal of Ecology 35(3-4): 611-638.

Doughty, C. R. and McPhail, C. D. (1995). Monitoring the environmental impacts and consent compliance of freshwater fish farms. Aquaculture Research 26(8): 557-565.

Du Plessis, D. (2007). Investigations on the likely impacts of cage aquaculture on water quality and plankton communities of farm dams in the Western Cape. Stellenbosch, Department of Conservation Ecology and Entomology, Stellenbosch University, South Africa.

DWAF (1996a). South African Water Quality Guidelines. Volume 6: Agricultural Water Use: Aquaculture. Department of Water Affairs and Forestry, Pretoria, South Africa.

DWAF (1996b). South African Water Quality Guidelines. Volume 7: Aquatic Ecosystems. Department of Water Affairs and Forestry, Pretoria, South Africa.

Eeirans, A. (2007). Zooplankton indicators of trophy in Latvian lakes. Acta Universitatis Latviensis 723: 61-69.

Ellenberg, H., Mayer, R. and Schauermann, J. (1986). Ökosystemforschung – Ergebnisse des Solling-Projektes. Ulmer, Stuttgart.

Elliott, J. A., Irish, A. E., Reynolds, C. S., and Tett, P. (2000). Modelling freshwater phytoplankton communities: an exercise in validation. Ecological Modelling 128: 19-26.

Elliott, J. A. and Thackeray, S. J. (2004). The simulation of phytoplankton in shallow and deep lakes using PROTECH. Ecological Modelling 178: 357-369.

Fedotov, P. S. and Spivakov, B. Y. (2008). Fractionation of elements in soils, sludges and sediments: batch and dynamic methods. Russian Chemical Review 77(7): 649-660.

Felsing, M. and Glencross, B. (2004). Defining the impact of hydrological changes associated with lake-turnover events on barramundi cage aquaculture in Lake Argyle. North Beach, Western Australia, Fisheries Research Division.

Folke, C. and Kautsky, N. (1989). The role of ecosystems for a sustainable development of aquaculture. Ambio 18: 234-243.

Foreel, E. G. and McCarty, P. L. (1970). Anaerobic decomposition of algae. Environmental Science and Technology 4(10): 842-849.

Gale, P. (1999). Addressing Concerns for Water Quality Impacts from Large-Scale Great Lakes Aquaculture. Roundtable Discussion Habitat Advisory Board of the Great Lakes Fishery Commission and Great Lakes Water Quality Board of the International Joint Commission. Toronto, Ontario Ministry of Environment.

Gavine, F. M., Phillips, M. J., and Murray, A. (1985). Influence of improved feed quality and food conversion ratios on phosphorus loadings from cage culture of rainbow trout, Oncorhynchus mykiss (Walbaum), in freshwater lakes. Aquaculture Research 26(7): 483-495.

Gächter, R. and Müller, B. (2003). Why the phosphorus retention of lakes does not necessarily depend on the oxygen supply to their sediment surface. Limnology and Oceanography 48: 929-933.

Geurden, I., Cuvier, A., Goundouin, E., Olsen, R. E., Ruohonen, K., Kaushika, S., and Boujard, T. (2005). Rainbow trout can discriminate between feeds with different oil sources. Physiology and Behaviour 85(2): 107-114.

Gonsiorczyk, T., Casper, P., and Koschel, R. (1997). Variations of Phosphorus Release from Sediments in Stratified Lakes. Water, Air, and Soil Pollution 99(1-4): 427-434.

Gooley, G. J. and Gavine, F. M. (2003). Integrated Agri-Aquaculture Systems: A Resource Handbook for Australian Industry Development. RIRDC Publication No *03/012*; RIRDC Project No. MFR-2A, 1-189. Rural industries research and development corporation. Kingston, Rural Industries Research and Development Corporation.

Goldblatt, P. (1997). Floristic diversity in the Cape Flora of South Africa. Biodiversity and Conservation 6(3): 359-377.

Government of Canada (2008). Biobasics: Bioindicators. http://www.biobasics.gc.ca/english/View.asp?x=740. Last modified 2008-07-08.

Grobler, D. C. and Silberbauer, M. J. (1985). Eutrophication: A look into the future. Water SA 11(2): 69-78.

Hakkari, L. (1972). Zooplankton species as indicators of environment. Aqua Fennica 1972: 46-54.

Hamblin, P. F. and Gale, P. (2002). Water Quality Modeling of Caged Aquaculture Impacts in Lake Wolsey, North Channel of Lake Huron. Journal of Great Lakes Research 28(1): 32-43.

Hart, R. and Hart, R. C. (2006). Reservoirs and their management: a review of the literature since 1990. WRC Report KV 173/06. Pretoria, Department of Water Affairs and Forestry.

Havens, K. E. (2000). Using Trophic State Index (TSI) Values to Draw Inferences Regarding Phytoplankton Limiting Factors and Seston Composition from Routine Water Quality Monitoring Data. Korean Journal of Limnology 33(3): 187-196.

Hinshaw, J. (1999). Trout Production: Feeds and Feeding Methods. SRAC. Austin, Southern Regional Agricultural Center and the Texas Aquaculture Extension Service.

Huang, Q., Wang, Z., Wang, C., Wang, S., and Jin, X. (2005). Phosphorus release in response to pH variation in the lake sediments with different ratios of iron-bound P to calcium-bound P. Chemical Speciation and Bioavailability 17(2): 55-61.

Hunt, R. J. and Matveev, V. F. (2005). The effects of nutrients and zooplankton community structure on phytoplankton growth in a subtropical Australian reservoir: An enclosure study. Limnologica 35: 90-101.

Huszar, V. L. M., Silva, L. H. S., Domingos, P., Marinho, M., and Melo, S. (1998). Phytoplankton species composition is more sensitive than OECD criteria to the trophic status of three Brazilian tropical lakes. Hydrobiologia 369-370: 59-71.

Huszar, V. L. M., Caraco, N. F., Roland, F., and Cole, J. (2006). Nutrient-chlorophyll relationships in tropical-subtropical lakes: do temperate models fit? Biogeochemistry 79(1-2): 239-250.

Huszar, V. L. M. and Caraco, N. F. (1998). The relationship between phytoplankton composition and physical-chemical variables: a comparison of taxonomic and morphological-functional descriptors in six temperate lakes. Freshwater Biology 40: 679-696.

Izaguirre, G. and Taylor W.D. (2004). A guide to geosmin- and MIB-producing cyanobacteria in the United States. Water Science and Technology 49(9): 19-24.

James, R. T. and Havens, K. E. (1996). Algal bloom probability in a large subtropical lake. Water Resources Bulletin 32(5): 995-1006.

Jayasinghe, U. A. D., Amarasinghe, U. S., and de Silva, S. S. (2005). Trophic Classification of Non-Perennial Reservoirs Utilized for the Development of Culture-Based Fisheries, Sri Lanka. International Review of Hydrobiology 90(2): 209-222.

Jensen, J. P., Kristensen, P., Jeppesen, E., and Skytthe, A. (1992). Iron-phosphorus ratio in surface sediment as an indicator of phosphate release from aerobic sediments in shallow lakes. Hydrobiologia 235-236: 731-743.

Jensen, J. P., Jeppesen, E., Olrik, K., and Kristensen, P. (1994). Impact of Nutrients and Physical Factors on the Shift from Cyanobacterial to Chlorophyte Dominance in Shallow Danish Lakes. Canadian Journal of Fisheries and Aquatic Sciences 51: 1692-1699.

Jewell, W. J. and McCarty, P. L. (1971). Aerobic decomposition of algae. Environmental Science and Technology 5: 1023-1031.

Johnson, R. K., Wiederholm, T., and Rosenberg, D. M. (1993). Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. In D. M. Rosenberg and V. H. Resh (eds.), Freshwater biomonitoring and benthic macroinvertebrates.

Kalff, J. (2002). Limnology - Inland water Ecosystems. New Jersey: Prentice Hall.

Kamenir, Y., Dubinsky, Z., Alster, A., and Zohary, T. (2007). Stable patterns in size structure of a phytoplankton species of Lake Kinneret. Hydrobiologica 578(1): 79-86.

Karaca, I. and Pulatsü, S. (2003). The Effect of Rainbow Trout (*Oncorhynchus mykiss* Walbaum, 1792) Cage Culture on Benthic Macrofauna in Kesikköprü Dam Lake. Turkish Journal of Veterinary and Animal Sciences 27: 1141-1146.

Kaweekityota, T., Taparhudee, W., Limsuwan, C., and Chuchird, N. (2007). Comparison study on production and plankton between two water exchange rates of recirculating shrimp culture (Penaeus monodon) system using low salinity water. Kasetsart University Fishery Research Bulletin 31(3): 1-10.

Kelly, L. A. (1995). Predicting the effect of cages on nutrient status of freshwater lochs using mass-balance models. Aquaculture Research 26: 469-477.

Kelly, L. A. (1993). Release rates and biological availability of phosphorus released from sediments receiving aquaculture wastes. Hydrobiologia 253: 367-372.

Kelly, L. A. (1992). Dissolved reactive phosphorus release from sediments beneath a freshwater cage aquaculture development in West Scotland. Hydrobiologia 235-236: 569-572.

Kim, J. D., Kaushik, S. J., and Breque, J. (1998). Nitrogen and phosphorus utilisation in rainbow trout (Oncorhynchus mykiss) fed diets with or without fish meal. Aquatic Living Resources 11: 261-264.

Klausen, C., Nicolaisen, M. H., Strobel, B. W., Warnecke, F., Nielsen, J. L., Jørgensen, and N.O.G. (2005). Abundance of actinobacteria and production of geosmin and 2-methylisoborneol in Danish streams and fish ponds. FEMS Microbiology Ecology 52(2): 265-278.

Lewis, D. M., Elliott, J. A., Lambert, M. F., and Reynolds, C. S. (2002). The simulation of an Australian reservoir using a phytoplankton community model: PROTECH. Ecological Modelling 150: 107-116.

Little J. L. and Smol, J. P. (2001). A chironomid-based model for inferring late-summer hypolimnetic oxygen in southeastern Ontario lakes. Journal of Paleolimnology 26(3): 259-270.

Longgen, G. and Zhongjie, L. (2003). Effects of nitrogen and phosphorus from fish cage-culture on the communities of a shallow lake in middle Yangtze River basin of China. Aquaculture 226: 201-212.

MacIsaac, E. A. and Stockner, J. G. (1995). The environmental effects of lake pen reared Atlantic salmon smolts. Burnaby, Science Council of British Columbia.

Maemets, A. (1983). Rotifers as indicators of lake types in Estonia. Hydrobiologia 104: 357-361.

Maleri, M., Du Plessis, D., and Salie, K. (2008). Assessment of the interaction between cage aquaculture and water quality in irrigation storage dams and canal systems. WRC Report No. 1461/1/08. Pretoria, Department of Water Affairs and Forestry.

Marsden, M. W., Fozzard, I. R., Clark, D., McLean, N., and Smith, M. R. (1995). Control of phosphorus inputs to a freshwater lake: a case study. Aquaculture Research 26: 527-538.

McNaught, D.C. and Buzzard, M. (1973). Changes in zooplankton populations in Lake Ontario. (1939–1972). Proceedings to the 16 th Conference on Lakes Research 1973: 76–86.

Molot, L. A., Dillon, P. J., Clark, B. J., and Neary, B. P. (1992). Predicting end-of-summer oxygen profiles in stratified lakes. Canadian Journal of Fisheries and Aquatic Sciences 49: 2363-2372.

Nicholls, K. H. and Dillon, P. J. (1978). An evaluation of phosphorus-chlorophyll-phytoplankton relationships for lakes. Internationale Revue der Gesamten Hydrobiologie 63: 141-154.

Nixdorf, B., Knopf, K., Mischke, U., Hoehn, E., and Riedmüller, U. (2000). Literaturstudie über vorhandene Klassifizierungs- und Bewertungsverfahren sowie Ansätze für den Merkmalskomplex Phytoplankton bei Fließgewässern und Seen einschließlich kritischer Wertung bezüglich ihrer Anwendbarkeit entsprechend den Anforderungen der EU-Wasserrahmenrichtlinie. Literaturstudie BTU Cottbus und LBH Freiburg.

Nürnberg, G. K. (1994). Phosphorus release from anoxic sediments: what we know and how we can deal with it. Limnetica 10: 1-4.

Osborne, P.L. (2005). Eutrophication of Shallow Tropical Lakes. In: O'Sullivan, P.E. and Reynolds, C.S (eds.). The Lakes Handbook, Volume 2: Lake Restoration and Rehabilitation. Wiley-Blackwell.

Osgood, R. A. (1982). Using differences among Carlson's trophic state index values in regional water quality assessment. Water Resources Bulletin 18: 67-74.

Penczak, T., Galicka, W., Molinski, M., Kusto, E., and Zalewski, M. (1982). The enrichment of a mesotrophic lake by carbon, phosphorus and nitrogen from the cage aquaculture of rainbow trout, Salmo gairdneri. Journal of Applied Ecology 19: 371-393.

Phillips, M. J., Beveridge, M. C. M., and Muir, J. F. (1985a). Waste Output and Environmental Effects of Rainbow Trout Cage Culture. Proceedings ICES Conference and Meeting Documents (C.M.). Copenhagen, International Council for the Exploration of the Sea.

Phillips, M. J., Beveridge, M. C. M., and Ross, L. G. (1985b). The environmental impact of salmonid cage culture on inland fisheries: present status and future trends. Journal of Fish Biology 27: 123-137.

Phillips, M. J., Clark, R., and Mowat, A. (1993). Phosphorus leaching from Atlantic salmon diets. Aquaculture Engineering 12(1): 47-54.

Pieterse, A. J. H. and van Zyl, J. M. (1988). Observations on the relation between phytoplankton diversity and environmental factors in the Vaal River at Balkfontein, South Africa. Hydrobiologia 169(2): 199-207.

Pike, T. (2009). Warm Water Fish Production. http://www.kznwildlife.com/index.php?/Warm-Water-Fish-Production.html. Ezemvelo KwaZulu-Natal Wildlife (Governmental Department).

Pillay, T. V. R. (2004). Aquaculture and the environment. Oxford, UK: Blackwell Publishing Ltd.

Podemski, C. L. and Blanchfield, P. J. (2006). Overview of the environmental impacts of Canadian freshwater aquaculture. A Scientific Review of the Potential Environmental Effects of Aquaculture in Aquatic Ecosystems - Volume 5. Canadian Technical Report of Fisheries and Aquatic Sciences. Ontario, Department of Fisheries and Oceans Canada.

Redshaw, C. J., Mason, C. F., Hayes, C. R., and Roberts, R. D. (1990). Factors influencing phosphate exchange across the sediment-water interface of eutrophic reservoirs. Hydrobiologia 192(2-3): 233-245.

Reynolds, C. S. (1992). Eutrophication and management of planktonic algae: What Vollenweider couldn't tell us. In D. W. Sutcliffe and J. G. Jones (eds.). Eutrophication: Research and Application to Water Supply. Ambleside: Freshwater Biological Association.

Reynolds, C. S. and Irish, A. E. (1997). Modelling phytoplankton dynamics in lakes and reservoirs: the problem of in-situ growth rates. Hydrobiologia 349: 5-17.

Reynolds, C. S. (1998). What factors influence the species composition of phytoplankton in lakes of different trophic status? Hydrobiologia 369-370(0): 11-26.

Reynolds, C. S. and Davies, P. S. (2001). Sources and bioavailability of phosphorus fractions in freshwater: A British perspective. Biological Reviews of the Cambridge Philosophical Society 76(1): 27-64.

Rodehutscord, M., Gregus, Z., and Pfeffer, E. (2000). Effect of phosphorus intake on faecal and non-faecal phosphorus excretion in rainbow trout (*Oncorhynchus mykiss*) and the consequences for comparative phosphorus availability studies. Aquaculture 188(3-4): 383-398.

Rose, B. and Robertson, D. (2007). Sediment Phosphorus Release Rate Data for Butternut Lake, Price County, Wisconsin. Middleton, Wisconsin, US Geological Survey.

Ruse, L. (2002). Chironomid pupal exuviae as indicators of lake status. Archiv für Hydrobiologie / Advances in Limnology 153: 367-390.

Rydin, E. (2000). Potentially mobile phosphorus in lake Erken sediment. Water Research 34(7): 2037-2042.

Salas, H. J. and Martino, P. A (1991). Simplified phosphorus trophic state model for warm tropical lakes. Water Research 25: 341-350.

Schindler, D. W. (1977). Evolution of phosphorus limitation in lakes. Science 196: 260-262.

Schutte, J. M. and Bosman, H. H. (1973). Fisiese en chemiese eienskappe van damme in die Republiek van Suid-Afrika. Technical Report No. 56. Division of Hydrology, Pretoria, Department of Water Affairs and Forestry.

Shannon, C. and Weaver, W. (1949). The Mathematical Theory of Communication. Urbana: University & Illinois Press.

Shuck, J.P. (2004). Development and assessment of models for predicting the phytoplankton assemblage patterns in Lake Kemp. Texas Tech University. Broadway, Lubbock, Texas.

Skelton, P. (2002) Trout: Conservationists and stakeholders speak out. www.scienceinafrica.co.za/2002/may/trout.htm.

Sommer, U. (1993). Limnoökologie. Heidelberg: Springer.

Steyn, D. J., Toerien, D. F., and Visser, J. H. (1976). Eutrophication levels of some South African Impoundments IV. Vaal Dam. Water SA 2(2): 53-57.

Stirling, H. P. and Dey, T. (1990). Impact of intensive cage fish farming on phytoplankton and periphyton of a Scottish Freshwater loch. Hydrobiologia 190:193-214.

Straskraba, M., Blazka, P., Brandl, Z., Hejzlar, P., Komarkova, J., Kubecka, J., Nesmerak, I., Prochazkova, L., Straskrabova, V., and Vyhnalek, V. (1993). Framework for investigation and evaluation of reservoir water quality in Czechoslovakia. In M. Straskraba, J. G. Tundisi, and A. Duncan (eds.). Comparative Reservoir Limnology and Water Quality management. The Netherlands: Kluwer Academic Publishers.

Temporetti, P. F. and Pedrozo, F. L. (2000). Phosphorus release rates from freshwater sediments effected by fish farming. Aquaculture Research 31: 447-455.

Thornton, J. (1987). A review of some unique aspects of the limnology of shallow Southern African man-made lakes. Geojournal 14(3): 339-352.

Ting, D. S. and Appan, A. (1996). General characteristics and fractions of phosphorus in aquatic sediments of two tropical reservoirs. Water Science and Technology 34(7-8): 53-59.

Tisdell, C. (1999). Overview of environmental and sustainability issues in aquaculture. Aquaculture Economics and Management 3: 1-5.

Toerien, D. F., Hyman, K. L., and Bruwer, M. J. (1975). A Preliminary Trophic Status Classification of some South African Impoundments. Water SA 1(1): 15-23.

Townsend, S. A. (1995). Metalimnetic and hypolimnetic deoxygenation in an Australian tropical reservoir of low trophic status. In K. H. Timotius and F. Goltenboth (eds.). Tropical Limnology. Vol II. Tropical Lakes and Reservoirs. Salatiga, Indonesia.

Townsend, S. A. (1999). The seasonal pattern of dissolved oxygen, and hypolimnetic deoxygenation, in two tropical Australian reservoirs. Lakes & Reservoirs: Research & Management 4(1): 41-53.

Veenstra, J., Nolen, S., Carroll, J., and Ruiz, C. (2003). Impact of net pen aquaculture on lake water quality. Water Science and Technology 47(12): 293-300.

Vollenweider, R. A. (1968). Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. OECD Technical Report DA5/SU/68-27 , 1-61. Paris, Organisation for Economic Co-operation and Development.

Vollenweider, R. A. and Kerekes, J. (1980). The loading concept as basis for controlling eutrophication philosophy and preliminary results of the OECD programme on eutrophication. Progress in Water Technology 12(2): 5-38.

Wang, S., Xie, P., Wu, S., and Wu, A. (2007). Crustacean zooplankton distribution patterns and their biomass as related to trophic indicators of 29 shallow subtropical lakes. Limnologica 37(3): 242-247.

Wang, S. R., Jin, X. C., Pang, Y., Zhao, H. C., Zhou, X. N., and Wu, F. C. (2005). Phosphorus fractions and phosphate sorption characteristics in relation to the sediment compositons of shallow lakes in the middle and lower reaches of Yangtze River region, China. Journal of Colloid and Interface Science 289: 339-346.

Weston, D. P., Phillips, M. J., and Kelly, L. A. (1996). Environmental impacts of salmonid culture. In W. Pennell and B.A. Barton (eds.). Principals of Salmonid Culture - Developments in Aquaculture and Fisheries Science. Amsterdam: Elsevier.

Wetzel, R. G. (2001). Limnology: Lake and River Ecosystems. London: Academic Press.

White, E. (1989). Utility of relationships between lake phosphorus and chlorophyll a as predictive tools in eutrophication control studies. New Zealand Journal of Marine and Freshwater Research 23: 35-41.

Whitney, R.J. (1942). Diurnal Fluctuations of Oxygen and pH In Two Small Ponds and A Stream. Journal of Experimental Biology 19: 92-99.

Woodford, D.J. and Impson, N.D. (2004). A preliminary assessment of the impact of alien rainbow trout (Oncorhynchus mykiss) on indigenous fishes of the upper Berg River, Western Cape Province, South Africa. African Journal of Aquatic Science 29: 107-111.

Yan, Q.Y., Yu, Y.H., Feng, W.S., Deng, W.N., and Song, X.H. (2004). Genetic Diversity of Plankton Community as Depicted by PCR-DGGE Fingerprinting and its Relation to Morphological Composition and Environmental Factors in Lake Donghu. Microbial Ecology 54(2): 290-297.

Yan, D. (2005). Research needs for the management of water quality issues, particularly phosphorus and oxygen concentrations, related to salmonid cage aquaculture in Canadian freshwaters. Environmental Reviews 13: 1-19.

Zhou, Q., Gibson, C. E., and Zhu, Y. (2001). Evaluation of phosphorus bioavailability in sediments of three contrasting lakes in China and the UK. Chemosphere 42: 221-225.

CHAPTER 3 WATER QUALITY STATUS AND ECOLOGY OF WESTERN CAPE IRRIGATION RESERVOIRS (BASELINE STUDIES)

Abstract

The water quality of small irrigation reservoirs within the Western Cape is primarily unknown. However, these reservoirs store approximately 10 % of the region's water resources and simulate a natural lake network. The main purpose of these reservoirs is irrigation water storage. Overall, 16 reservoirs were monitored over a study period of 26 months. Dry summers (October to March) and winter rainfall (April to September) within a Mediterranean climate characterise the sites. Physical and chemical water properties as well as phytoplankton and zooplankton communities were recorded.

The combined results of water properties, nutrients, phytoplankton and zooplankton reflect mesotrophic to hypertrophic water conditions (one site mesotrophic, four eutrophic, six highly eutrophic and five hypertrophic). With higher trophic states, the total surface phosphorus concentrations (overall range 17 to 153 µg/L as phosphorus) and surface pH values increased. Intensive phytoplankton production combined with naturally low alkalinity and hardness, shifted pH values up to 9.8 at some sites. Total nitrogen and nitrate were not indicators of trophic state, however, surface ammonia levels reached acute chronic toxicity during hot summer days at the two most eutrophicated sites (free ammonia up to 0.5 mg/L) and ammonia accumulated in the hypolimnion during stagnation. Overall nitrogen levels were low and TN:TP (total nitrogen:total phosphorus) ratios (2.1 to 13.5) indicated a tendency towards nitrogen limitation.

When plotting total phosphorus concentrations against algal biomass, three groups emerged: sites at elevation >500 m a.m.s.l. of a low agricultural impact, sites <500 m a.m.s.l. with underlying shale and sites <500 m a.m.s.l. with a underlying granite. Phosphorus concentrations were lowest in the group with low agricultural impact (35-45 µg/L total phosphorus), with higher concentrations in the shale group (60-80 µg/L total phosphorus) and highest concentrations in the granite group (>100 µg/L total phosphorus).

The phytoplankton biomass increased with increasing trophic state (up to 1.5 mg/L in mesotrophic conditions, with 40 mg/L and more in highly eutrophic conditions), and the structure developed from several peaks and co-dominance of some species to single excessive biomass peaks with dominating large, motile chlorophytes. Within the hypertrophic sites, a dinophyte dominance emerged which suppressed the diversity to five species while it lasted. Cyanophyte abundance was low at hypertrophic sites. Overall, phytoplankton diversity decreased with increasing eutrophication. The size of the organisms increased and the phytoplankton resilience

decreased with increasing eutrophication with a strong tendency towards blooms of *Ceratium hirundinella*.

The biomass of zooplankton ranged from 40 to 160 mg/L from mesotrophic to highly eutrophic reservoirs. Within the zooplankton structure, copepods had the greatest biomass overall, however rotifers were the most species abundant group with up to 14 species identified (independent of trophic status). The number of zooplankton peaks decreased with increasing trophic state. Zooplankton and dominating (primarily heterotrophic) phytoplankton belong to a similar functional group within lake ecology by taking advantage of periods of high nutrient availability (turnover events).

When extrapolating the findings on total phosphorus in the surface water and comparing differences among highland to lowland sites, a net increase of 2 to 10 µg/L TP can be attributed to phosphorus accumulation in lowlands and external sources (mostly agriculture).

3.1 Introduction

The near-coastal climate in the Western Cape, South Africa, can be referred to as Mediterranean with warm, dry summers and mild, humid winters. In coastal regions annual precipitation can exceed 1000 mm, which decreases to approximately 500 mm and less 50 to 100 km inland (SAWS 2007). Approximately 80 % of the rain falls between April and September, the winter period. As water availability in the Western Cape is seasonal and most water drains into fluvial systems, natural open freshwater water systems are scarce. However, a large proportion of the seasonal run-off is stored in reservoirs and approximately 4000 reservoirs were already registered by the early 90’s (Berg et al. 1994). There are less than 20 reservoirs larger than 100 ha in surface area and together they store around one billion cubic meters of water. The 4000 smaller reservoirs in four major river basins, total an estimated 120 million m^3 (Berg et al. 1994). These small, open water storage systems resemble a natural lake network. They are usually directly connected to natural or piped inflowing water, but often have no direct outflow into a river system. The primary purpose of these small reservoirs is the storage of irrigation and drinking water as well as water for cleaning of production facilities, the catchment of drainage water and others.

The ecological state of small farm reservoirs is largely unknown. The governmental monitoring scheme includes larger reservoirs in the south-western Western Cape region such as Theewaterskloof Reservoir, Steenbras Reservoir, Wemmershoek Reservoir, Kogelberg and Voelvlei Reservoir as well as most rivers, but excludes privately owned farm reservoirs (Hart & Hart 2006). Past studies investigated mountainous areas of the Western Cape and found the reservoirs in these areas to be of low trophic state (Toerien et al. 1975). More recent studies investigated farm reservoirs, but were locally restricted to the north-eastern area of South Africa which differs greatly from the Cape region. In the north-eastern areas, three quarters of all impoundments were categorized as enriched (Thornton 1987).

Small reservoirs in South Africa are usually warm monomictic which means they stagnate during summer and mix during the winter period (Thornton 1989, Steyn et al. 1976, Toerien et al. 1975). The duration of summer stagnation depends on many other factors, for example wind action or water inflow and outflow regimes (Hart & Hart 2006). Hypolimnetic oxygen depletion can develop during stagnation (and is enhanced with higher eutrophication levels) and has implications on chemical exchange processes in the sediment, the nutrient budget and other managerial factors such as options for the location of water extraction and subsequent water use. Large Australian and South African reservoirs of low trophic state have been reported to develop slow hypolimnetic deoxygenation (Hart 2001, 1999; Townsend 1995), but deoxygenation is temperature dependent and occurred between 25 to 30 °C in these cases.

Nutrients enter water bodies via different pathways, and these are divided into point and non-point sources (Mason 2002). Point sources comprise of any inflowing water when river water or channelled reservoir water accesses the lake. Sewage channels and stormwater are other point sources. Most pollution in farm reservoirs derives from non-point sources such as rainfall, runoff water, fertiliser and pesticide application, informal settlements, leaf litter, faeces by waterfowl and sediment re-suspension. Runoff water is the most difficult parameter to estimate and strongly depends on land uses within the watershed. As one of the purposes of farm reservoirs is to supply irrigation water, most farm reservoirs are surrounded by agricultural land. Therefore, all nutrients that are applied to the surrounding land and that are not bound to the soil or plant material will reach the water body. Agricultural runoff has been identified as the leading pollution type impacting water quality in lakes in the United States (EPA 2003). Any excess fertiliser or fertiliser applied just before rainfall, will enter the water and impair water quality by adding nutrients, primarily nitrogen, phosphorus and potassium. Point sources can be controlled to a certain degree which is why non-point sources have been given more attention recently. In Europe, input of phosphorus via non-point runoff has been shown to be consistently sufficient in feeding ongoing eutrophication (Hillbricht-Illkowska & Pieczynska 1993).

In the coastal Western Cape, the prevailing natural landscape consists of different types of fynbos, an indigenous scrubland. The evergreen hard-leaved shrubs contain many humic and fulvic acids and strongly influence the pH of waters receiving runoff from fynbos vegetation. Some reservoirs are surrounded by intensively cultivated land where runoff water is influenced by the management of for example: orchards, vines, various cereal crops, vegetables and olives. Agriculture in South Africa is still intensifying with increasing fertiliser and pesticide application in general, but in the Western Cape water saving irrigation systems and fertilisation adapted to soil character and demands has become the rule (DEAT 2006). With a change of land use, runoff water quality is usually modified. The pH reducing effect of fynbos material declines and instead runoff water carries nutrients and pesticides into the water body which increases pH of the water bodies by enhancing algal production. Primarily inorganic nitrogen and potassium are leached from soils via runoff water and to a lesser degree, phosphorus. The crucial point, however, is the great sensitivity of aquatic ecosystems towards phosphorus, when transferred from terrestrial systems. The quantity of phosphorus that substantially alters the trophic status of a water body is negligible and often not even measurable (Tiessen 1995). However, there are reported cases where soil erosion actually reduces soluble phosphorus concentrations in surface waters by binding the mineral (Sequi et al. 1989). With leached pesticides, the insecticide Endosulfan was found to be above the critical effect value (CEV) in 47 % of 60 reservoirs in a study near Elgin and Caledon, and to be above the acute effect value

(AEV) in 24 % of the reservoirs (Davies 1997). Other pesticides (parathion-methyl, penconazole, Azinphos-methyl, Dimethoate) were also found above recommended concentrations at many sites.

The primary goal of this study was to determine the present trophic status of selected Western Cape farm reservoirs in which no aquaculture has ever been practiced. General ecology of the selected reservoirs, in terms of water mixing processes, chemistry, phytoplankton biomass and abundance and zooplankton biomass was described. These data can be helpful in understanding the ecological functioning of artificial open water bodies in the area and their role in supporting natural flora and fauna. Additionally, irrigation planning can profit from understanding ecological processes that cause changes in water extraction points and effect irrigation equipment.

3.2 Sites and climatic conditions

All sites were situated within the south-western area of the Western Cape, South Africa, near the towns of Hermanus, Stellenbosch, Worcester, Malmesbury and Ceres (Figure 3.1).

There are four major water management areas (WMA) within the Western Cape. Eleven of 16 study sites were within the Breede management area, five of 16 sites were within the Berg management area and one site was within the Olifants/Doring management area (Table 3.1).

The Western Cape is a very diverse geological region, with all sites in this study falling within the Cape Supergroup or Malmesbury Group (DEADP 2007). The predominant underlying rock type in the Breede WMA consists of shale and sandstone, whereas in the Berg WMA granite and shale dominate. Source water in the area is of a low mineral content which is later enriched by aerosols and rainwater runoff. Three of the 16 reservoirs were located >500 m a.m.s.l. and were not influenced by any source other than groundwater inflow and runoff influenced by the natural fynbos vegetation. The other 13 reservoirs were additionally affected by agricultural activities such as vineyards, orchards and pine plantations.

Five reservoirs were supplied with river water via channels or sluices, with controllable quantities entering the reservoirs. All others were supplied with groundwater and runoff only. The effluent from only one reservoir re-entered a river course. The function of the studied reservoirs was irrigation water storage and additionally three of the 16 reservoirs served as a drinking water supply. The water exchange rate (inflowing water volume/maximum water volume of reservoir) varied from 0.3 to 5.5 times/y, with a median of 0.7 times/y.

Department of Water Affairs and Forestry (DWAF) unpublished data for six larger reservoirs from the area was made available (DWAF 2007). Misverstand and Voelvlei Reservoir are within the Berg Water Management Area whereas Steenbras, Kogelberg, Eikenhof and Kraaibos (Uylenkraal) Reservoirs lie within the Breede Water Management Area (Figure 3.1).

Table 3.1: The reservoirs sampled during the study with hydromorphological characteristics and surrounding land use (a.m.s.l. = above mean sea level).

Reservoir	Location	Surface area	Max. depth	Elevation (a.m.s.l.)	Water management area	Underlying rock type of the catchment	Surrounding land use
Unit		ha	m		- contributing river		
1	Elgin	1.7	11.0	165	Breede – Palmiet	sandstone, shale	fruit orchards
2	Elgin	6.9	16.0	132	Breede – Palmiet	sandstone, shale	fruit orchards, fynbos
3	Drakenstein	7.4	9.5	152	Berg - Kromme	granite, alluvial, shale	vineyards, olives
4	Franschhoek	6.9	10.0	210	Berg – Paarl Mountain runoff	granite, alluvial, shale	vineyards
5	Stellenbosch	1.8	9.5	248	Berg – Eerste	granite, shale	forest, pasture
6	Elgin	11.0	9.0	75	Breede – Palmiet	sandstone, shale	fruit orchards (apples)
7	Stellenbosch	1.4	6.0	207	Berg – Eerste	granite, shale	vineyards, fruit orchards
8	Elgin	240.0	18.0	312	Breede – Palmiet	sandstone, shale	fynbos, forestry, watersport
9	Near Hermanus	2.3	7.0	128	Breede – Bot River	sandstone, ferricrete, silcrete	fruit orchards, vegetables
10	Franschhoek	7.2	11.0	225	Berg – Paarl Mountain Runoff	granite, alluvial, shale	vineyards
11	Rawsonville	7.4	9.5	255	Breede - Romans	shale, sandstone	fynbos, vineyards, human settlement
12	Rawsonville	6.5	6.0	254	Breede - Romans	shale, sandstone	rocks, fynbos
13	Villiersdorp	7.1	16.5	558	Breede – Riviersonderend	sandstone, shale	fynbos
14	Villiersdorp	73	10.0	738	Breede - Riviersonderend	sandstone, shale	human settlement, fynbos
15	Worcester	2.2	7.0	224	Breede - Breede	shale, sandstone	cereals, vineyards
16	Ceres - Op die Berg	16.5	10.0	949	Olifants/Doring – Upper Olifants	sandstone, granite	rocks, fynbos

Climatic conditions (air temperature, total rainfall, wind direction and wind speed) were analysed using South African Weather Service information (SAWS 2007) derived from the monitoring stations at Elgin (SE of Stellenbosch), Paarl (NE of Stellenbosch), Strand (SW of Stellenbosch), Hermanus, Worcester and Ceres (Figure 3.1).

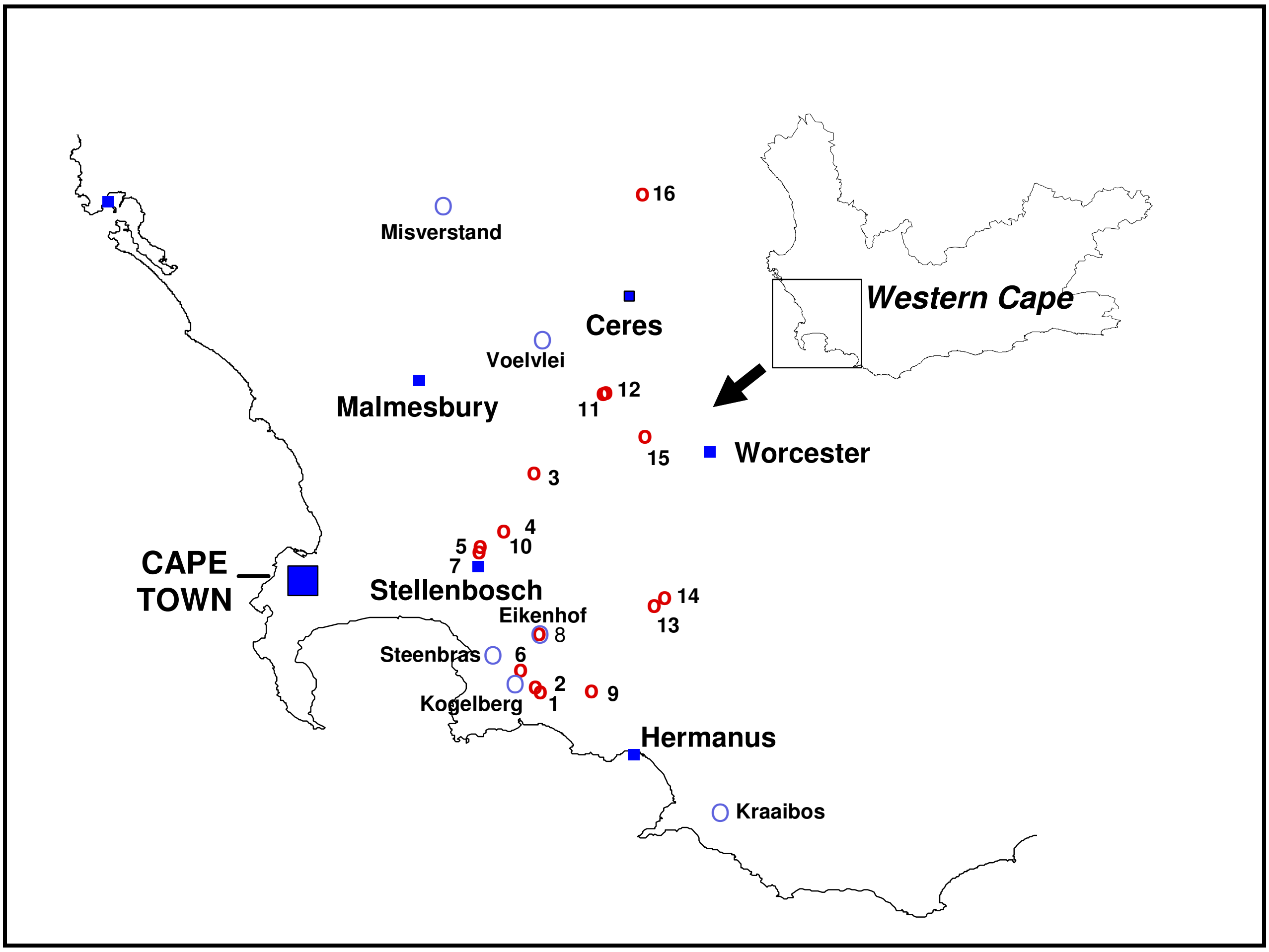

Figure 3.1: Sampling locations within the Western Cape Province, South Africa. Main cities and towns are marked with squares, six large reservoirs are represented by large circles and names (DWAF data), the 16 small reservoirs sampled during the study are represented by small circles and numbers. Reservoir 1 to 7 were intensively studied. Reservoir 4 and Reservoir 10 were so close the circles overlap, Eikenhof and Reservoir 8 are the same reservoir (sampled by DWAF and the author).

Summer maximum air temperatures ranged from a monthly average of 28 to 32 °C with slightly warmer temperatures in Paarl when compared to the Strand and Elgin, and highest temperatures in Ceres. Summer minimum temperatures varied between 15 and 19 °C, with Ceres showing the coolest night temperatures. During the winter period, average maximum temperatures ranged from 15 to 20 °C, with Ceres again showing cooler average winter temperatures, while monthly minimum temperatures averaged from 2 to 10 °C. Ceres experienced the coldest winters within the sampling area whereas the Strand area, influenced by close proximity to the ocean, experienced the warmest winters. Paarl and Elgin's minimum night temperatures ranged from 5 to 7 °C.

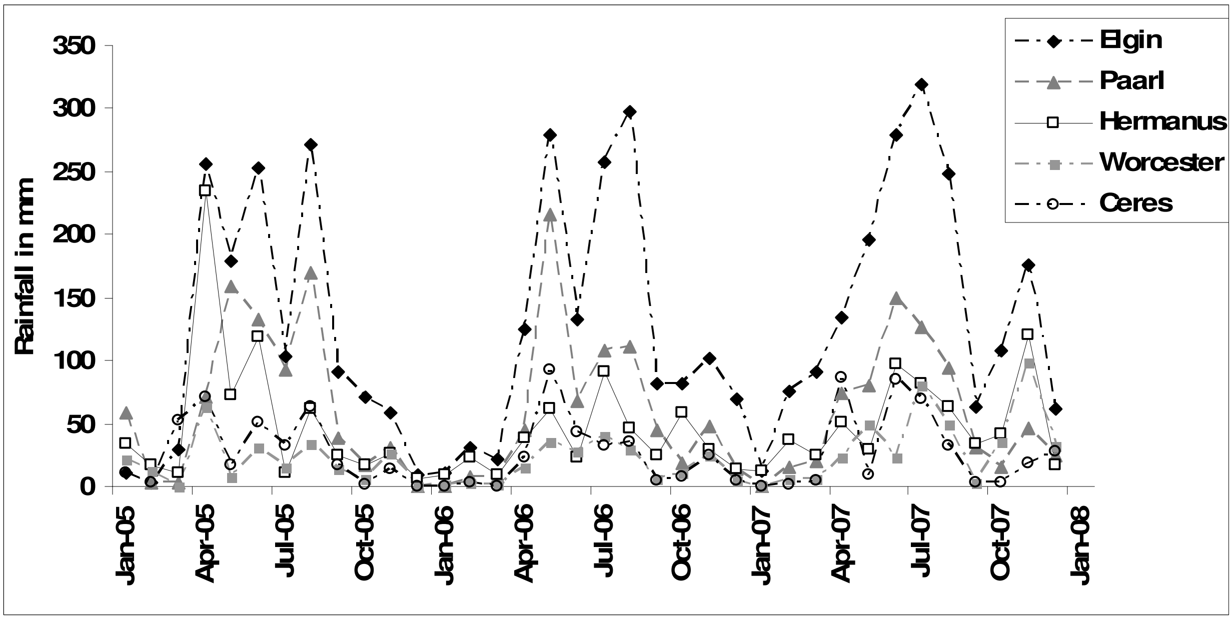

Figure 3.2: Monthly (summated) rainfall in mm from January 2005 to December 2007 at five weather stations near the sampling sites in the Western Cape (data supplied by SAWS).

All sites lie within the Mediterranean climatic belt with winter rainfall which is usually highest from April to October (Figure 3.2). Total rainfall was highest in Elgin (>1300 mm/y) followed by Paarl and Hermanus (400 to 700 mm/y) and Worcester and the Ceres Valley (200 to 400 mm/y).

The predominant wind direction at the Paarl station was south to south-east during the summer period with prevailing westerly winds during the winter months. The hours of wind action varied from a monthly average of 17 to 18 h/d during summer, to 10 to 12 h/d during the winter months, with monthly average wind speeds of 26 km/h during summer and 18 km/h during winter. The maximum wind speed in the area varied from 70 to 80 km/h. These fresh to strong gales developed for several hours only, with no seasonal preference, and induced wave action and intermediate mixing in the smaller reservoirs (SAWS 2007). In Ceres, average and maximum wind speeds were higher with 15 to 16 wind h/d throughout the year, average monthly wind speeds varied between 18 and 35 km/h and maximum wind speeds varied between 70 and 100 km/h. Unlike in the Paarl region, the predominant wind direction was south to south-west (SAWS 2007).

3.3 Methods

Seven reservoirs were monitored monthly (Reservoir 1 to 7, Table 3.1) over two years and an additional nine reservoirs (Reservoir 8 to 16, Table 3.1) were visited three times per year over the study period (Figure 1.2). Site specific information was recorded and relevant chemical and physical water quality parameters were measured. Phytoplankton composition and biomass was intensively studied in seven reservoirs (Reservoir 1 to 7) and zooplankton composition and biomass was intensively studied in five reservoirs (Reservoir 1, 2, 3, 5 and 6). Chlorophyll a levels (derived from biomass data) were calculated for all sixteen reservoirs. Trophic status was determined for all study sites according to Carlson (1977).

3.3.1 Morphometry and hydrology

The length and width of each reservoir at full capacity was measured using Google Earth software (http://earth.google.com) and surface area was estimated via the nearest geometrical shape (usually ellipse or rectangle). The deepest region of five reservoirs was determined through transect measurements of the horizontal and vertical central lines (Wetzel & Likens 2000) and was estimated at all other sites, while the depth at each specific sampling location within the reservoirs was recorded.

3.3.2 Water quality

The seven intensively studied sites were visited every four weeks from August 2005 to October 2007, and the other nine sites were visited on three occasions in both 2006 and 2007 (March, July and October). After preliminary depth measurements by means of a weighted line, a buoy was set in the deepest area to ensure a consistent sampling location. The sampling sites were visited in a fixed sequence so that the time of day at which the samples were taken at any one site remained within a two hour window of the previous sampling effort. The sites were accessed by boat (Appendix 10.9, Picture 2). At each of the seven sites, water samples were collected at a depth of 2 m, 6 m and near the sediment interface using a 1.5-L water sampler with a single line trigger mechanism (The Science Source, Waldoboro, ME, USA) and stored and transported according to Wetzel and Likens (2000). At the nine sites that were visited on three occasions per year, only the 2 m and bottom samples were collected. Turbidity, oxygen content and temperature were measured on site (Appendix 10.9, Picture 3), with all other parameters being determined the following day in the laboratory (Apppendix 10.9, Picture 4). For a complete list of all physical and chemical water parameters monitored, as well as analytical methods applied refer to Table 3.2.

On four occasions (December 2005, April and July 2006, January 2007), samples of the seven reservoirs were externally analysed at the DWAF laboratories in Pretoria, which served as an external quality control and provided additional information on total nitrogen content, presence of major anions (SO_4, F, Cl) and cations (Na, Mg, K and Ca), as well as several trace metals.

Water quality status was described with the Carlson Trophic State Index (TSI) (Carlson 1977). The formula introduced by Carlson depends on phosphorus content, Secchi depth (visibility) and chlorophyll a content, values that were readily vailable. Chlorophyll a content was calculated from the phytoplankton biomass determination. Vörös and Padisak (1991) calculated that the relative chlorophyll a content of phytoplankton biomass varied between 0.1 and 2 %. An average of 1 % was applied for the conversion in this study. Both, Carlson's average TSI calculated as an average from three TSI values and TSI(TP), the value calculated from the total phosphorus concentrations, were used in this study.

Table 3.2: Parameters measured with methods applied (n.a. = not applicable).

Parameter	Unit	Method	Reference
Turbidity	***cm***	Secchi disc	Wetzel & Likens (2000)
Temperature	***°C***	Oxyguard MK III, OxyGuard Polaris	OxyGuard International A/S, Birkerød, Denmark
Oxygen	***mg/L***	Oxyguard MK III, OxyGuard Polaris	OxyGuard International A/S, Birkerød, Denmark
pH	***(n.a.)***	Hanna pH 211 microprocessor	Hanna Instruments, Woonsocket, RI, USA
Conductivity	***mS/m***	HACH CO 150 Conductivity meter	Hach Company, Loveland, CO, USA
Alkalinity	***mg/L***	Complexometric titration with 0.02N H_2SO_4 (method 2320)	APHA et al. (2005), USEPA approved
Hardness	***mg/L***	Complexometric titration with 0.01M EDTA (method 2340c)	APHA et al. (2005), USEPA approved
Silicate	***mg/L***	Heteropoly Blue Method	APHA et al. (2005)
Ferrous Iron	***mg/L***	1,10-Phenanthroline Method	Stucki (1981), APHA et al. (2005)
Sulfide	***mg/L***	Methylene Blue Method (method 4500d)	APHA et al. (2005) USEPA method 376.2
TSS	***mg/L***	Photometric Method	Hach (2005)
Soluble reactive P (srp)	***µg/L***	Molybdo-vanadate Method (filtered sample)	APHA et al. (2005), USEPA approved
Total P (TP)	***µg/L***	Molybdo-vanadate Method (unfiltered, acid digested sample)	Adapted from APHA et al. (2005)
Nitrate	***mg/L***	Cadmium reduction Method	
Nitrite	***mg/L***	Diazotization Method	
Ammonia	***mg/L***	Nesslerization Method & Salicylate Method	APHA et al. (2005), USEPA method 350.2 (Nessler)
BOD	***mg/L***	Oxyguard MK III, OxyGuard Polaris	Closed bottle test (5 days)

DWAF data on phosphorus content and pH were accessible for a selection of six large reservoirs within the same area of the Western Cape (DWAF 2007). Average values for the period of January 2003 to March 2007 were determined, although the sampling dates and total number of samplings varied between the reservoirs (12 to 83 values per reservoir were available).

3.3.3 Phytoplankton

Unfiltered water samples were collected at a depth of 2 m, 6 m and at the bottom at seven sites (Reservoir 1 to 7). Lugol's iodine solution was immediately added as a preservation method. Species identification and enumeration was possible by means of Utermöhl counting chambers (Utermöhl 1958) using an inverted microscope (Appendix 10.9, Picture 5). Identification volumes of Pascher's "Süßwasserflora von Mitteleuropa" (Komarek & Anagnostidis 1998, Krammer & Lange-Bertalot 1991a und b, Popovsky & Pfiester 1990, Ettl & Gärtner 1988, Krammer & Lange-Bertalot 1988, 1986, Kadlubowska 1984, Häusler 1982) and Thienemann's volumes "Das Phytoplankton des Süßwassers" (Huber-Pestalozzi & Förster 1982, Huber-Pestalozzi & Fott 1972, Huber-Pestalozzi 1961, 1955, 1950, Huber-Pestalozzi & Hustedt 1942, Huber-Pestalozzi 1941, 1938), as well as single keys such as Entwisle et al. (1997), van den Hoek et al. (1995), Yunfang (1995) and Prescott (1978) were used. The biovolume of each specimen was taken from the literature or calculated via the nearest geometrical shape. Biomass was calculated from the volume data using factors of 1.02 to 1.30 kg/m^3 (Sommer 1996).

The Shannon-Weaver index describes the stability within phytoplankton communities at species level and increases as the phytoplankton community increases in species diversity (Shannon & Weaver 1949). More eutrophic reservoirs would therefore usually have decreased H' values. The formula applied from the Shannon and Weaver (1949) publication was:

$$H' = -\sum_{i=1}^{s} \frac{N_i}{N_0} \cdot \log_2 \frac{N_i}{N_0}$$

where H' = Shannon-Weaver index, s = number of species, N_i = number of individuals of species i, N_0 = sum of all individuals of all species.

3.3.4 Zooplankton

At five sites (Reservoirs 1, 2, 3, 5 and 6), zooplankton was collected with a 10-L self-constructed Schindler-Patalas plankton trap. The samples were preserved in 5 % formaldehyde and transferred into a phenoxetol medium called Steedman's solution within 24 hours (Steedman 1976). Protozoans and some rotifera were already established with the phytoplankton section in the Utermöhl chambers, whereas the crustaceans, (cladocera and copepoda), as well as the majority of the rotifera were quantified and identified to genus level with support of a modified Bogorov counting tray and a Leica stereomicroscope (6.3x to 50x magnification) (Appendix 10.9, Picture 6). Staining of organisms with diluted Lugol's solution supported the counting process. Identification was based on Day et al. (1999, 2001), Thirion (1999) and Yunfang (1995).

The biovolume and biomass of rotifers and protozoans were estimated via the nearest geometrical shape. The biomass of cladocerans and copepods were estimated via dry biomass estimations of other studies (Gonzalez et al. 2008, Sendacz et al. 2006).

3.3.5 Statistical analyses

Statistical analyses were undertaken using the Statistica 7.0 program (StatSoft, Inc.). The student's t-test was employed with data sets showing a normal distribution. Differences were considered statistically significant if $p<0.05$ and strongly significant if $p<0.01$. Correlations were analysed using the Spearman's rank correlation coefficient. Coefficient values were considered statistically significant with $p<0.05$ and strongly significant with $p<0.01$.

3.4 Results & discussion of individual water quality parameters

The discussion of individual parameters in individual reservoirs was drawn to the results section for clarity of structure. The general discussion (Chapter 3.5) was dedicated to the analysis of the reservoirs as integrated systems in their respective backgrounds.

3.4.1 Temperature and oxygen distribution - mixing patterns

The water temperature of the examined reservoirs differed between the epilimnion and hypolimnion from August to March, with the greatest differences from August to January (Figure 3.3, Reservoirs 1 and 7, also Appendix 10.1). Stratification was established in six of the seven reservoirs, even in reservoirs of only 6 m in depth.

In Reservoir 1, one of the deepest reservoirs with a <10 ha surface area, the thermocline established in September 2005 and sat above a depth of 6 m. In November, the thermocline moved below the 6 m mark (Figure 3.3). In February, when the water level of the reservoir dropped to 5.5 m (from 7 m the previous month), the thermocline disappeared with an increase in temperature throughout the entire water mass. In Reservoir 7, an exemplary shallow reservoir with a <10 ha surface area, the maximum depth at full supply was 6 m. Nonetheless, a thermocline developed from October to February in the first study year, as well as from October to early January in the second study year, with the water level decreasing to 3.5 and 5.5 m in overall depth from September to May 2006 and 2007, respectively.

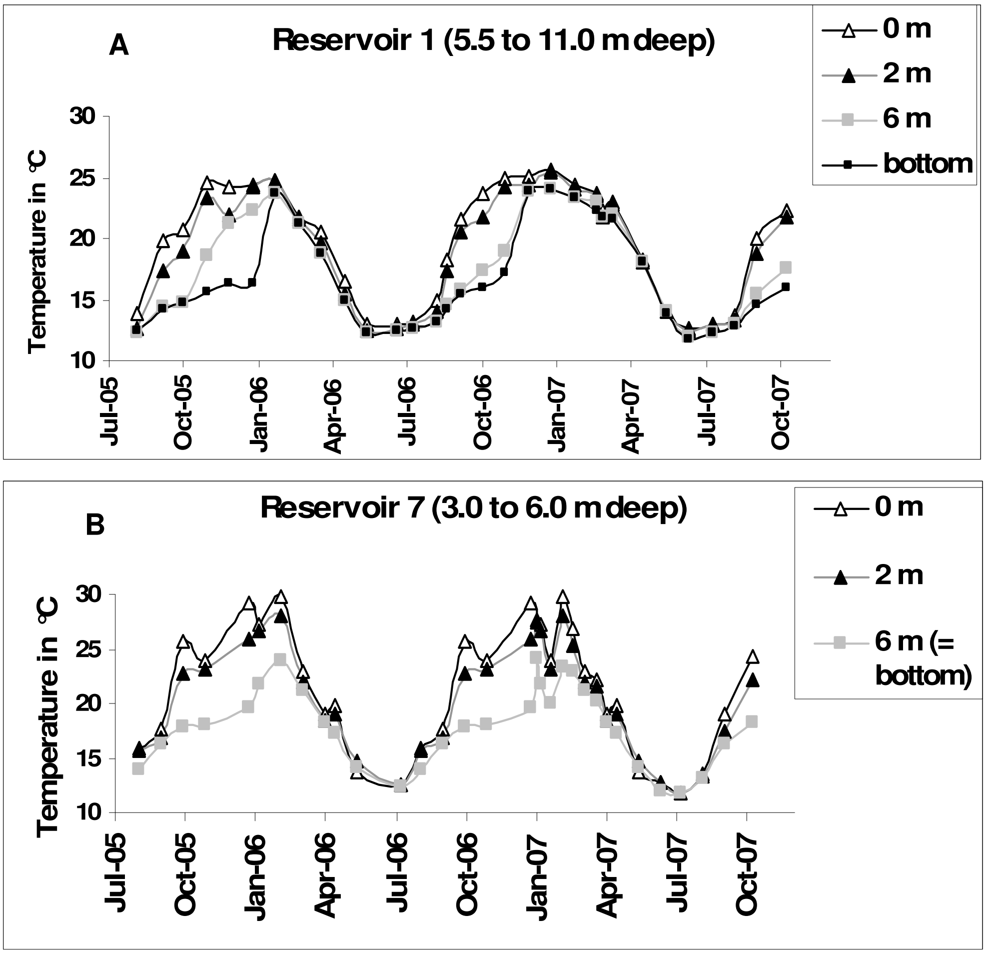

Figure 3.3: Water temperature development at Reservoir 1 (A), a deeper lake of 11 m maximum depth situated in Elgin, and Reservoir 7 (B), a lake with a maximum depth of 6 m at full supply level situated in the Stellenbosch area. Both reservoirs had the lowest water level in late summer.

In general, the reservoirs in the Stellenbosch area showed longer periods of temperature differences between depths, in shallower, but especially in reservoirs deeper than 7 m on average throughout the year. In these deeper reservoirs differences of at least 6 °C were stable from August through to March. In Elgin and the Drakenstein district, the reservoirs could only develop divergences from September to January, with a core period in October and November.

In summer 2006/07, the thermocline in Reservoir 1 lasted from September to November, which can be explained by the overall basin water level that dropped below 6 m by December. Wind action was also stronger in late 2006 when compared to 2005 (more days of wind, higher average wind speed, higher maximum wind speed).

It seems possible that with days of strong wind action during stratification, the physical barrier can be disrupted (January 2007 in Reservoir 7). With a wind speed of approximately 40 km/h, a 6 to 12 m water column can be fully mixed in 10 °C water, and shear forces to be overcome are weaker in warmer water (Tomczak & Godfrey 1994). Therefore, wind episodes of 70 to 90 km/h, as they occurred between September and November, disturbed the weak thermocline with a 3 to 5 °C temperature difference between the epi- and hypolimnion.

Together with temperature differences and wind action, another driving force of stratification and mixing processes in closed water bodies is the hydrodynamic regime in a water body (Schwoerbel 2005, Bohle 1995). In Reservoir 3, a very short or no thermocline developed during the study period, probably due to a year-round water inflow entering the hypolimnion or stronger wind action in that specific area (funnelled by the shape of the valley and documented in the field reports). In general, in- and outflowing water did not affect thermal stratification in the studied reservoirs. The bulk of inflowing water entered the reservoirs during the winter period (April to September) which is a time of heterogeneous temperature distribution, stronger winds and observed reservoir mixing (distribution of phosphorus and nitrogen concentrations).

The same seven reservoirs were studied for their oxygen distribution patterns. Six had periods of no to low (<2 mg/L) oxygen in bottom waters and there was a trend of an increasing number of months of oxygen depletion with increasing eutrophication (no statistical evaluation possible with data of two study years; Figure 3.5).

Two reservoirs of different trophic status were chosen as a classic example of oxygen distribution (Figure 3.4A to D). Results for other reservoirs can be found in Appendix 10.2. Reservoir 1 (highly eutrophic) was depleted of oxygen in the bottom waters from October to December 2005. The temperature graph (Figure 3.3A) suggests that a thermocline was established from September to January in the first year, therefore temperature and oxygen overlap with a month's delay in oxygen reduction in the bottom waters. In the second year, the temperature stratification only occurred from September to November, while oxygen levels were diminished until February and March 2007. Neither extreme phytoplankton nor zooplankton biomasses developed in Reservoir 1 in order to explain oxygen depletion by decomposition. Therefore, the best explanation for the decrease in oxygen is the ceased wind action and water depths of >6 m (Figure 3.4C), with concurrent oxygen consumption by breakdown of sedimented organic material.

Reservoir 6 (hypereutrophic) was potentially 9 m deep with a very low water level in the first year (2.5 to 3 m) (Figure 3.4D). The low water level permitted wind action to mix the water masses to the bottom, thereby oxygenating the entire water mass. A thermocline was present from September to January in the same year, but intermediate mixing must have taken place (Figure 3.4B). In the second year, the thermocline was stable from September to November 2006 only, but low oxygen levels in the basin prevailed from August 2006 to May 2007, primarily due to a water level which always exceeded 6 m.

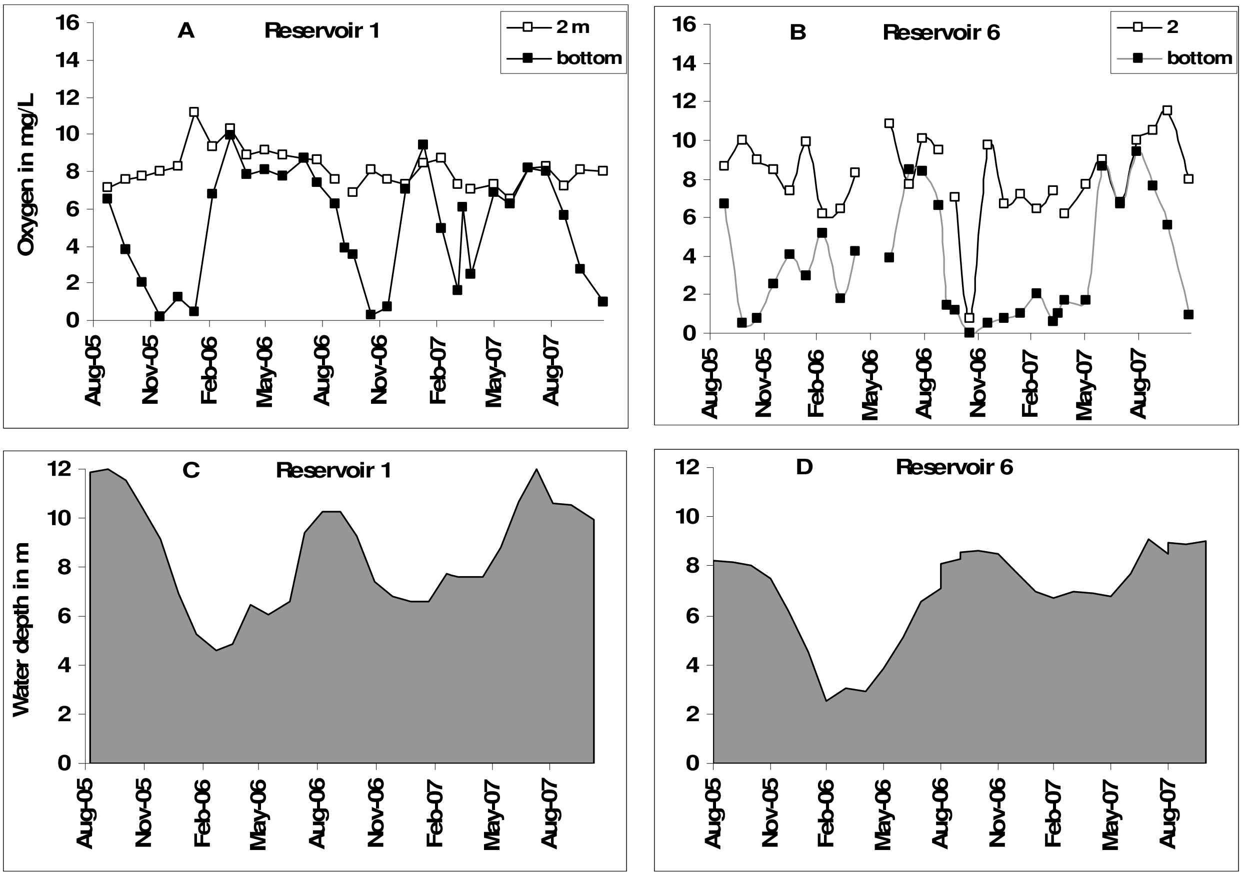

Figure 3.4: Graphs A and B show the oxygen distribution at two depths for Reservoirs 1 and 6 while graphs C and D show the corresponding maximum depth over time (representing the water level of the reservoir).

Concerning <u>bottom water oxygen depletion</u>, at some sites, the duration of oxygen concentrations <2 mg/L in the hypolimnion did not exceed three months (Reservoir 1, 2, 3 and 4 in Elgin, Drakenstein and Franschhoek), whereas at the Stellenbosch site Reservoir 5 oxygen levels below 2 mg/L lasted for four to five months (Figure 3.5). In the second study year, Reservoir 6 had a 10 month period of low oxygen in its bottom water, while in the previous year low oxygen concentrations only prevailed for 4 months, probably due to the relative shallowness of the reservoir in the first year (Figure 3.4B and D). Reservoir 7 experienced 6 months of low oxygen concentrations. The oxygen concentration of Reservoir 3 never dropped below 4 mg/L during the two year study period.

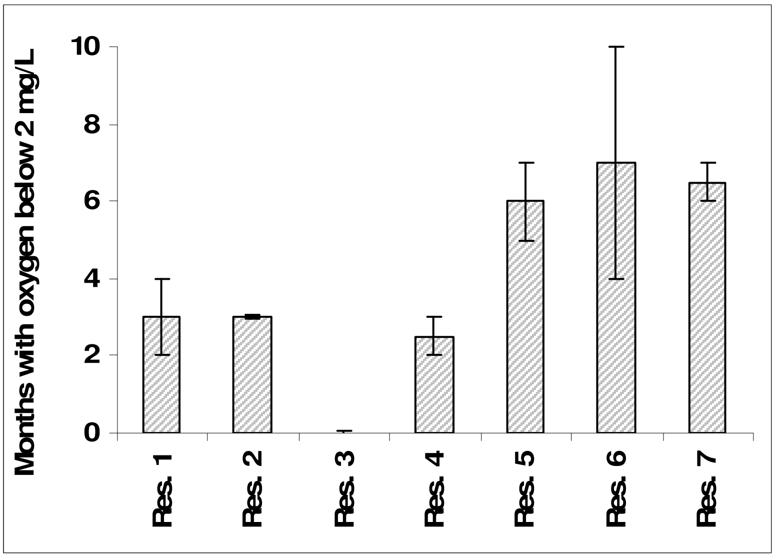

Figure 3.5: Average number of months in which the bottom water of the reservoirs had oxygen concentrations of less than 2 mg/L (n=2). The error bars represent the two values that could be determined from two stagnation periods in a two year observation period.

3.4.2 Physical and chemical water properties

Table 3.3 shows water quality reflected by typical parameters at the intensively studied sites (average values or ranges shown). Fifteen reservoirs had surface areas of less than 20 ha, with five reservoirs having surface areas below 2.5 ha. Reservoir 8 (Eikenhof Reservoir) had a surface area of approximately 240 ha.

Water temperatures ranged from a lower, quite homogenous temperature of 12 °C in winter to an upper temperature of 26 – 30 °C in summer, with the highest summer water temperatures recorded in the Stellenbosch area. Oxygen at 2 m depth varied from a minimum of around 6.5 mg/L (only exception was Reservoir 6 with a minimum of 4 mg/L, directly after the phytoplankton peak collapsed) to an upper limit of 11.5 - 14 mg/L, with Reservoirs 4 to 7 representing the reservoirs with the highest maximum oxygen concentrations. The periods of oxygen levels higher than 10 mg/L in most reservoirs concurs with the phytoplankton peaks, except for Reservoir 6 where the peaks coincide with the periods of dominance of filamentous green algae.

Table 3.3: Water quality parameters monitored for Reservoir 1 to 7 from August 2005 to October 2007 shown as a range over the study period or an average (n=29). TP = total phosphorus, SD = Secchi depth, srp = soluble reactive phosphorus, TN = total nitrogen, TIN = total inorganic nitrogen, Chl a = chlorophyll a, TSI = trophic state index, Cond = conductivity, TSS = total suspended solids. (n.a. = not applicable – dimensionless variable).

Parameter	Unit	1	2	3	4	5	6	7
Surface area	ha	1.7	6.9	7.4	7.2	1.8	11	1.4
Depth	m	11	16	9.5	11	9.5	9	6
Average TSI	(n.a.)	67	67	69	71	63	74	79
TSI (Chl a)	(n.a.)	76	65	63	69	43	69	81
TSI (TP)	(n.a.)	64	68	73	73	74	77	76
TSI (SD)	(n.a.)	62	68	71	72	72	76	79
Range of Water temp. (2 m)	°C	12.6 to 25.7	12.2 to 26.6	12.0 to 28.0	12.0 to 24.8	12.0 to 29.0	12.0 to 25.8	11.8 to 30.0
Range of Oxygen (2 m)	mg/L	6.5 to 11.5	6.3 to 12.1	6.4 to 11.6	7.1 to 14.2	6.5 to 14.6	4.0 to 14.4	6.4 to 14.0
Secchi	cm	40 to 450	18 to 350	80 to 350	80 to 350	80 to 300	20 to 230	20 to 300
Average Secchi	cm	145	113	136	156	163	151	80
Range of pH 2m	(n.a.)	6.1 to 9.2	6.3 to 8.4	5.8 to 8.8	6.0 to 9.8	5.7 to 8.6	6.6 to 9.8	6.7 to 9.5
Average pH	(n.a.)	7.1	7.2	7.5	7.3	7.4	8.5	7.9
Cond 2m	µS/cm	157 to 460	132 to 277	135 to 201	61 to 80	86 to 150	158 to 463	96 to 128
TSS (2m)	mg/L	1 to 40	1 to 27	1 to 19	1 to 29	1 to 28	1 to 48	1 to 49
Average TSS (2m)	mg/L	9.0	11.7	8.4	9.0	8.4	9.5	14.5
Range of Alkalinity (2m)	mg/L	2 to 41	6 to 18	19 to 30	3 to 12	11 to 27	50 to 108	12 to 28
Range of Hardness (2m)	mg/L CaO	21 to 37	16 to 28	11 to 17	2 to 6	7 to 12	26 to 55	--
Average TP as P (2m)	µg/L	63	83	119	121	129	153	142
Average srp as P (2m)	µg/L	61	29	73	52	51	81	64
Range of TN as N (2m)	mg/L	0.3 to 5.5	0.6 to 1.0	0.3 to 0.5	--	0.5 to 0.8	0.7 to 2.8	0.7 to 1.2
Average TIN as N (2m)	mg/L	1.21	0.26	0.19	0.42	0.15	0.49	0.22
Ammonia as N (2m)	mg/L	0.0 to 0.4	0.0 to 0.4	0.0 to 0.5	0.0 to 0.8	0.0 to 1.1	0.0 to 0.5	0.0 to 1.1
Ammonia as N (bt)	mg/L	0.0 to 1.2	0.0 to 0.8	0.0 to 1.4	0.0 to 0.2	0.0 to 0.8	0.0 to 1.6	--
Nitrate as N (2m)	mg/L	0.0 to 5.2	0.0 to 0.5	0.1 to 0.2	0.0 to 0.7	0.0 to 0.1	0.0 to 2.3	0.0 to 0.2

Secchi depths ranged from 18 to 450 cm with periods of less visibility (<150 cm) during the rainy period. Additionally, when small phytoplankters of sufficient cumulated biomass/m^3 (usually more than 1 mg/L) appeared, the Secchi depth was impacted upon. The yearly average was around 150 cm. Reservoir 2 and 7 had lower averages with results from Reservoir 7 being considerably lower than all other sites. Low results from Reservoir 2 could be due to inflowing water with a high sediment load and results from Reservoir 7 probably due to the higher algal biomasses. Table 3.4 shows the average Secchi depths (>3 measurements per reservoir) of the additional 9 sites. Five reservoirs had visibilities between 80 and 165 cm, with the exception of two reservoirs with a Secchi visibility of only 30 cm. These two reservoirs were tea coloured, influenced by fynbos material.

Secchi depth was influenced by the presence of suspended solids (TSS) and algal growth. Algal peaks were not reflected in concurrent low visibility, possibly due to the lack of smaller algae and the presence of larger species such as *Ceratium sp.*, *Volvox sp.* and *Eudorina sp*. Total suspended solids were influenced by lake turnovers (sediment resuspension) and strong rain events causing soil runoff, therefore with peaks coinciding with the wet season (Appendix 10.3). However, in some reservoirs, suspended solid presence peaked in January and February 2006 (summer).

Table 3.4: Secchi depth, pH, TSS, total phosphorus, chlorophyll a and Carlson index for Reservoirs 8 to 16 (average of three samples in 2006 or 2007). The reservoirs are ranked according to the TSI (TP), corresponding to the surface (2 m) concentrations of total phosphorus. TP = total phosphorus, SD = Secchi depth, srp = soluble reactive phosphorus, TIN = total inorganic nitrogen, Chl a = chlorophyll a, TSI = trophic state index, TSS = total suspended solids. (n.a. = not applicable – dimensionless variable).

Reser-voir	TSI average	TSI (Chl a)	TSI (TP)	TSI (SD)	pH (2m)	Secchi	TSS (2m)	TP (2m)	srp (2m)	TIN (2m)	Chl a (2m)
	(n.a.)	(n.a.)	(n.a.)	(n.a.)	(n.a.)	in cm	in mg/L	in µg/L	in µg/L	in mg/L	in µg/L
12	41	24	45	53	7.88	165	5	17	7	0.04	0.5
8	49	34	54	58	4.89	118	12	31	22	0.04	1.4
16	49	37	56	54	6.3	155	8	36	19	0.12	1.9
13	52	24	58	75	4.56	35	32	42	25	0.02	0.5
14	61	47	59	77	6.64	30	36	45	20	0.46	5.0
9	59	49	65	63	7.12	80	14	66	44	0.06	6.5
15	64	64	65	62	8.9	85	9	67	13	0.12	30.8
11	54	39	66	56	6.94	135	10	73	14	0.32	2.3
10	65	72	68	54	7.36	150	8	83	11	0.02	66.2

Table 3.5: DWAF data for six large reservoirs within the study area (DWAF 2007). Average of 12 to 57 values for TP and 27 to 83 values for pH, srp and Silica. TSI = Trophic State Index, TP = total phosphorus, srp = soluble reactive phosphorus. (n.a. = not applicable - dimensionless variable).

Reservoir	TSI (TP)	pH	TP	srp	Silica
	(n.a.)	(n.a.)	in mg/L	in mg/L	in mg/L
Kogelberg	51	7.21	26	24	1.66
Eikenhof	54	5.15	32	23	1.31
Steenbras	55	6.03	35	22	1.58
Voelvlei	63	7.23	61	29	1.8
Kraaibos	66	7.33	71	35	2.57
Misverstand	77	7.4	157	35	1.77

The average pH as well as the maximum pH were higher in the more eutrophicated reservoirs, with values varying from 5.7 to 9.8 with the highest average pH in Reservoirs 6 and 7. Two reservoirs had pH values below 5 (Reservoirs 8 and 13). Another three reservoirs (11, 14 and 16) had slightly acidic values and four reservoirs had a tendency to alkaline pH values with Reservoir 15 rising to a pH value of 8.9 (Appendix 10.4).

The six large reservoirs from the region (Table 3.5) had slightly alkalinic pH levels with two reservoirs maintaining a naturally slightly acidic (Steenbras Reservoir) or strongly acidic pH (Eikenhof Reservoir).

Conductivity depends primarily on underlying soil and inflowing water quality. Overall, the conductivity of waters in the study region was low with minimum levels of 61 µS/cm and maximum levels of 463 µS/cm (equivalent to 6.1 mS/m and 46.3 mS/m). According to the South African Water Guidelines for Aquatic Ecosystems (DWAF 1996) it can be concluded that the water flowing into Reservoir 4 (Franschhoek area) was in contact with granite or siliceous material with a maximum conductivity value of 80 µS/cm (equivalent to a total dissolved salts level of 40 mg/L). Most other reservoirs with conductivity values above 130 (Reservoirs 1 to 3 and Reservoir 6) were characterized by shale. The conductivity of Reservoirs 5 and 7 was at an intermediate level of between 80 to 150 µS/cm. The nine additional reservoirs (not in table) included four that were very low in dissolved salts content (Reservoirs 8, 10,11 and 16), all less than 80 µS/cm and with Reservoir 16 showing a conductivity of <30 µS/cm. Reservoirs 13 to 15 had conductivity values around 100 µS/cm and Reservoirs 9 and 12 had values >200 µS/cm. The total suspended solid levels varied between 1 and 50 at all 16 sites (one outlier, Reservoir 2, with TSS levels around 90). The TSS levels showed single peaks that settled to normal background levels within one or two months. The peaks were primarily found from July to November, with higher TSS values in the reservoirs with finer sediments. Elevated TSS levels also occurred around February and March when water levels decreased and bottom sediment was re-suspended. After heavy downpours the TSS values in the reservoirs were temporarily increased by fine particles in the runoff water (soil erosion).

Hardness and alkalinity of the water were primarily defined by the calcium and magnesium content for hardness and the carbonate and bicarbonate content for alkalinity. All sixteen reservoirs had water with a CaO concentration of <70 mg/L which is also referred to as very soft water (0 to 40 mg/L CaO). The low alkalinity of the water may also explain the high variability in pH since the low buffering capacity of the water makes it sensitive to pH shifts by phytoplankton, rain and evaporation as well as inflowing water quality.

3.4.3 Nutrients

Average total phosphorus ranged from 63 to 153 µg/L as P in Reservoirs 1 to 7 (Table 3.3). In Reservoirs 8 to 16, total phosphorus ranged from 17 to 83 µg/L as P (Table 3.4). Average total phosphorus in six large reservoirs in the area ranged from 26 to 157 µg/L as P (Table 3.5).

The yearly cycle of total phosphorus and soluble reactive phosphorus in Reservoirs 1 to 7 is shown in Figures 3.6 and 3.7. With total phosphorus, Reservoirs 5, 6 and 7 showed an annual cycle with elevated concentrations in winter, while with soluble reactive phosphorus a seasonal pattern is only apparent for Reservoir 6 (find detailed graphs in Appendix 10.5). Generally, elevated soluble reactive phosphorus was observed in the summer of 2005/2006, a very dry summer.

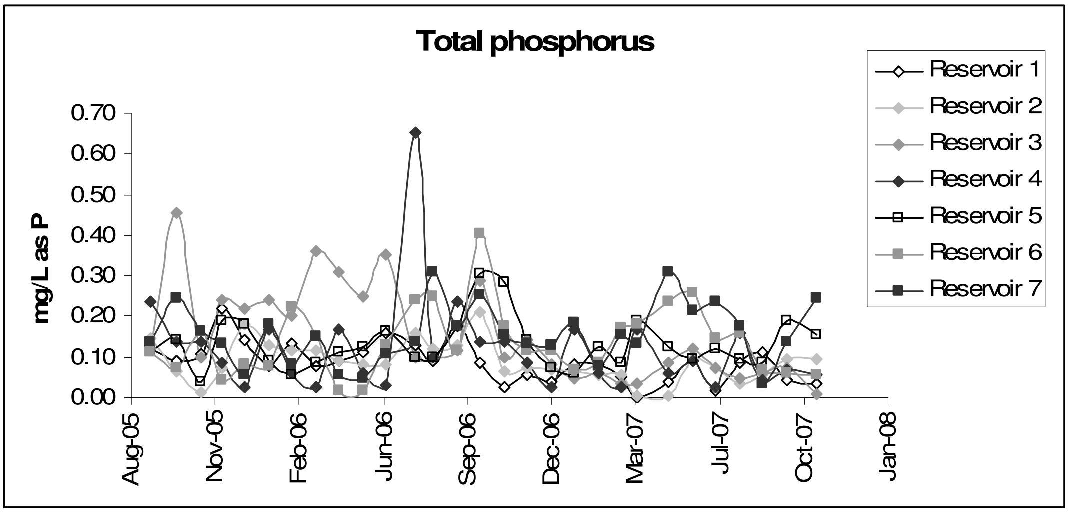

Figure 3.6: Total phosphorus concentration in mg/L as P from August 2005 to October 2007 at the depth of 2 m.

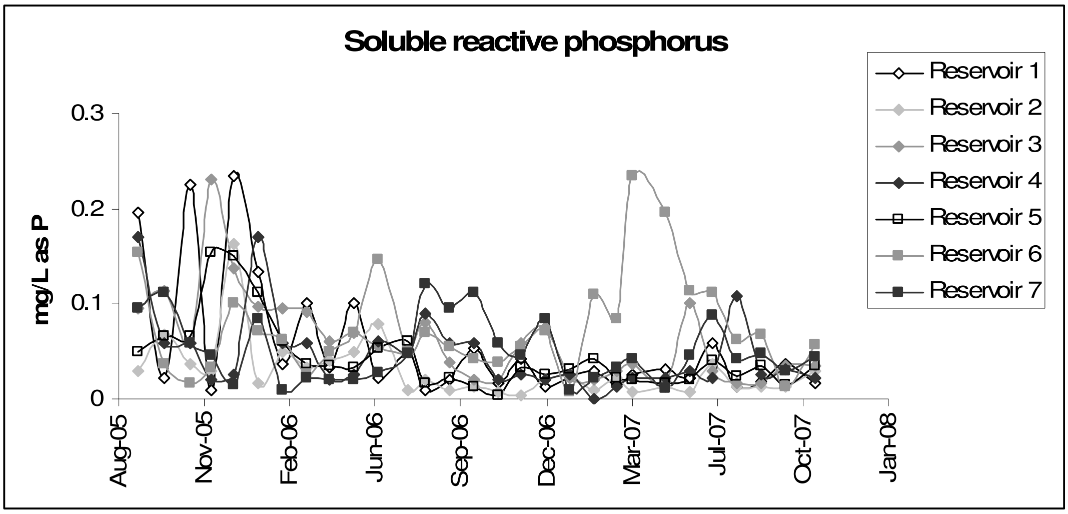

Figure 3.7: Soluble reactive phosphorus concentration in mg/L as P from August 2005 to October 2007 at the depth of 2 m.

Total nitrogen (TN) concentrations , sampled at a depth of 2 m, varied between 0.3 and 5.5 mg/L. Ammonia can be most harmful to aquatic life, depending on temperature and pH which determines the balance between ammonium and free ammonia (toxic ammonia). The maximum total ammonia concentrations ranged from 0.4 to 1.1 mg/L in summer, when the daily fluctuation of pH allowed concentrations of free ammonia of up to 0.50 mg/L. With Reservoir nutrient status (Reservoir 1 to Reservoir 7), a divergence in ammonia concentrations between the epi- and hypolimnion was commonly observed (see Figure 3.8, Reservoir 7 as an example. Data for the other reservoirs is shown in Appendix 10.6.

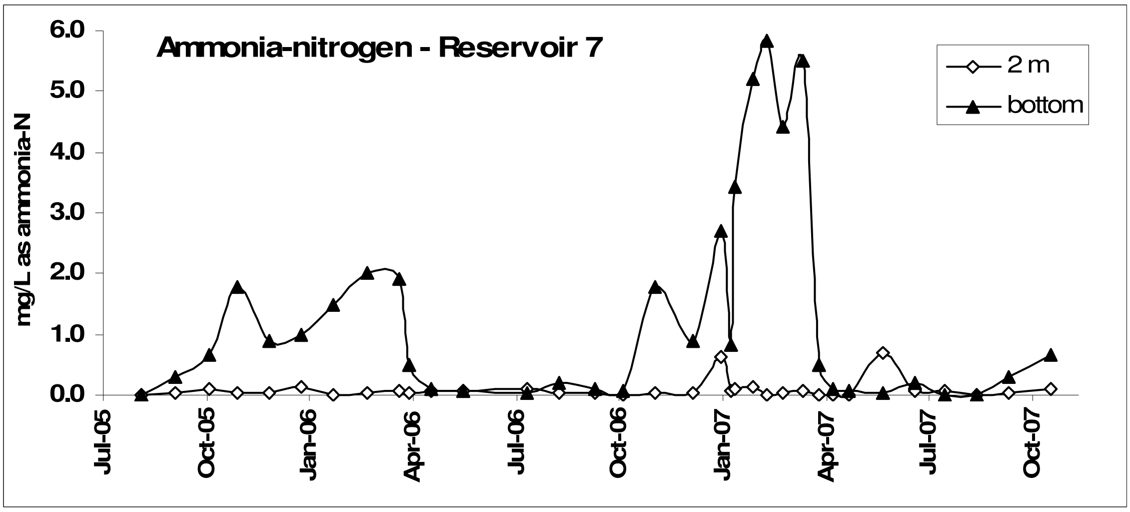

Figure 3.8: Ammonia-nitrogen (equivalent to TAN = total ammonia nitrogen) in mg/L as N from August 2005 to October 2007 at a depth of 2 m and at the bottom.

Nitrate is the most readily available nitrogen source for algae. In Reservoirs 1 and 6, nitrate concentrations reached levels of above 5 mg/L at times, while in the other reservoirs the concentrations remained at <0.7 mg/L. The average total inorganic nitrogen (TIN) content of the reservoirs was approximately <0.5 mg/L which represents the conditions in an oligotrophic lake according to the South African Water Quality Guidelines (DWAF 1996). Reservoir 1's average inorganic nitrogen level reflected mesotrophic conditions, where nuisance growth of aquatic plants and blooms of blue-green algae occurred. The nitrogen to phosphorus ratio is very important in determining final algal growth. The inorganic nitrogen results suggested a nitrogen limited nutrient situation in most of the reservoirs.

3.4.4 Elevation and total phosphorus concentrations

With analyses among nutrient concentrations and parameters defining and describing the reservoirs, a strong correlation was found between total phosphorus content and elevation, and also geological background. The difference among reservoirs situated at higher (in this study defined at >500 m a.m.s.l.) and lower elevations (<500 m a.m.s.l.), was due to differences in land use. No agricultural practices influenced the reservoirs referred to as "highland sites", in comparison to predominantly agricultural influences on the "lowland sites".

Total phosphorus showed a significantly negative correlation (Pearson correlation/squared correlation) with elevation of the sampling stations ($R^2 = -0.71$, $p<0.01$, $n=16$), with lowest total phosphorus concentrations at the highest elevations and visa versa. In a more detailed differentiation of the phosphorus levels versus elevation, differences were found among three experimentally determined groups (Figure 3.7). Within group 1, Reservoirs 3, 4, 5, 6 and 7 (subgroup granite) were underlain by granite, whereas Reservoirs 1, 2, 9, 10, 11 and 15 (subgroup shale) were characterized by sandstone and shale (Figure 3.6).

All three groups differed significantly when analysed for phosphorus levels at a depth of 2 m using a student's t-test (Group 1 shale vs group 2, $t=-4.92$, $p<0.05$; group 1 granite vs group 2, $t=-9.03$, $p<0.01$; group 1 shale vs group 1 granite, $t=-8.55$, $p<0.01$) (also see Box-Whisker plot, Figure 3.10). Group 3 consisted of a new reservoir (younger than one year when sampled) and Eikenhof Reservoir which was 20-fold larger in surface area than all other 14 reservoirs). When comparing surrounding land use (Table 3.1) to the groups in Figure 3.9, the results underline the selected grouping. Agriculture had a great influence on runoff water entering the reservoirs of group 1, whereas the reservoirs at a higher elevation (group 2 and 3) were less impacted by agricultural runoff.

The portion of srp (with total phosphorus) varied from 13 to 96 % of TP (total phosphorus) with no distinct pattern. The sites at higher altitude (highland sites) showed srp fractions of 44 to 59 %, similar to the lowland shale sites with 43 to 61 %. The lowland granite sites were very heterogeneous in their srp portions ranging from 13 to 96 % with other factors obviously influencing the presence of srp (phosphorus directly available to algae). The srp concentrations ranged from 7 to 81 µg/L in the 15 small reservoirs, with very uniform values in the six large reservoirs where the srp remained between 22 and 35 µg/L.

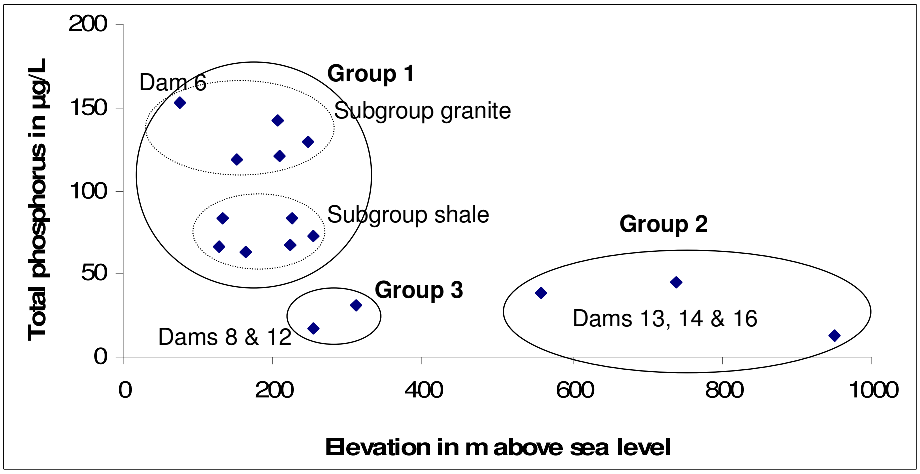

Figure 3.9: Total phosphorus concentrations versus elevation (a.m.s.l.). Lines show groups 1 to 3, dotted lines refer to the subgroups within group 1. Reservoir numbers refer to group 2 and group 3 and subgroups within group 1.

With nitrogen, the grouping of “altitude” versus group 1 shale and group 1 granite could also be verified (Figure 3.10).

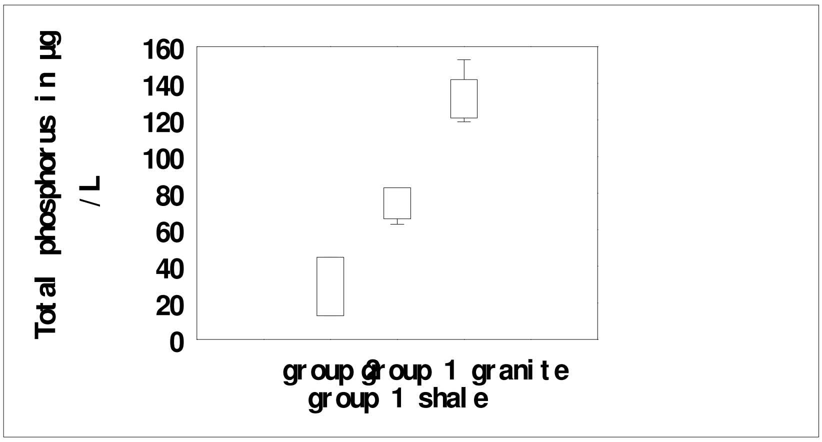

Figure 3.10: Box-Whisker plot of groups 1 and 2 with subgroups with median, percentiles and minimum and maximum values.

3.4.5 Trace metals and mineral salts

Understanding lake ecology requires a basic understanding of an extensive set of constituents affecting water quality. Next to physical and chemical properties as well as nutrient contents, mineral salts and trace metals also play a role in overall ecosystem functioning (Table 3.6).

Aluminium (Al) is potentially toxic at levels above 0.01 mg/L in acidic water and at 0.02 mg/L in alkaline water (DWAF 1996). Reservoirs 1 and 7, but in particular Reservoir 3, showed a higher total aluminium concentration, of which a portion was bioavailable. The reason for the elevated levels in Reservoir 3 may be due to the construction process of a stilt house in 2007.

Iron (Fe) and manganese (Mn) occurred in low concentrations and therefore did not affect gills of fish or reservoir colour. A manganese concentration of 0.15 mg/L produces an undesirable taste in drinking water, but only reached concentrations near this limit in Reservoir 5. Iron concentrations would be elevated (above 0.2 mg/L) near industrial sites but did not reach concentrations higher than 0.13 (Reservoir 3).

Table 3.6: Average trace metal contents in mg/L (four values per reservoir) in Reservoir 1 to 3 and 5 to 7.

	Al	*Fe*	*Ba*	*B*	*Mo*	*Mn*
Reservoir 1	0.029	0.070	0.017	0.024	0.003	0.001
Reservoir 2	0.010	0.019	0.022	0.018	0.027	0.001
Reservoir 3	0.089	0.130	0.026	0.018	0.003	0.023
Reservoir 5	0.014	0.069	0.020	0.018	0.003	0.112
Reservoir 6	0.011	0.031	0.019	0.101	0.003	0.006
Reservoir 7	0.036	0.066	0.021	0.018	0.003	0.001

Barium (Ba) is not an essential metal for higher organisms and occurred at fairly stable concentrations in all reservoirs varying from 0.017 to 0.026 mg/L.

Boron (B) is a ubiquitous metal which is essential to plant and animal growth. A concentration <0.1 mg/L is considered to be fairly low. Boron levels <0.3 mg/L do not pose a health hazard (WHO 1993).

Molybdenum (Mo) concentrations should remain <0.07 mg/L (WHO 1993). Only Reservoir 2 had a higher molybdenum concentration (0.027 mg/L) than all other reservoirs.

Table 3.7: Four anions and three cations in mg/L (average of 4 values per reservoir) present in Reservoir 1 to 3 and 5 to 7.

	Na	*Mg*	*Ca*	*K*	*SO_4*	*F*	*Cl*
Reservoir 1	37	8	10	3.5	11	0.1	72
Reservoir 2	25	6	10	2.6	7	0.1	53
Reservoir 3	21	5	6	3.6	5	0.3	39
Reservoir 5	15	3	4	2.2	4	0.2	26
Reservoir 6	38	6	24	6.3	11	0.1	43
Reservoir 7	19	4.5	7.5	3.5	8	0.2	29

The total concentration and ratio of mineral salts to each other was defined by the underlying rock type and additional salt introduction, primarily from agricultural production, however the overall salinity of the reservoirs can be regarded as low (Table 3.7).

Magnesium (Mg) and calcium (Ca) are essential nutrients for algal production. Magnesium can be a limiting component for plant growth at levels below 1.5 mg/L, however this was not the case for any of the six studied reservoirs. Water with elevated calcium levels was more eutrophic. In Reservoir 6 the Ca concentration exceeded 24 mg/L, however, this reservoir was also identified to be of lowest trophic status (highest TSI value).

Potassium (K) is not regarded as a health hazard (in drinking water) at levels <12 mg/L.

Sulfate (SO_4) is usually the second most common anion. When the hypolimnion becomes anoxic, sulfate can be reduced to sulfide which is toxic to bottom dwellers. Levels of 5 to 50 mg/L occur in natural waters and therefore the results from this study (4 to 11 mg/L) were considered to be well below the norm.

A fluoride concentration of <1.5 mg/L is desirable in drinking water and also has no negative effect on the environment. All reservoirs in this study had fluoride averages of less than 0.3 mg/L.

3.4.6 Carlson trophic state index (TSI)

In Table 3.3, Reservoir 1 to 7 were sorted according to the outcome of the Carlson's trophic state index (TP) with Reservoir 1 being in a lesser eutrophicated state and Reservoir 7 in the most eutrophicated state. The ranges and categories were applied according to studies of temperate area lakes due to the lack of globally valid categories for subtropical areas. The Carlson TSI (TP) ranged from a highly eutrophic category (TSI between 60 and 70, for example: Reservoirs 1 and 2) to a hypereutrophic category (with TSI values higher than 70, for example: Reservoirs 3 to 7) (Table 3.3). Carlson ascribes a TSI value of >70 to cyanophyte or excessive macrophyte growth, as well as algal scums. In this study, Reservoir 7 had cyanophyte dominated periods and algal scum and Reservoir 6 was dense with floating filamentous green algae, almost impenetrable by boat (accumulated by wind action at one side of the reservoir). Reservoir 5 had dense macrophyte stands, however in comparison to Reservoir 6 and 7, no cyanophytes or filamentous algae were present. No macrophytes or filamentous algae were observed in Reservoirs 1 to 4 which showed only slightly lower TSI values (TP).

The additional nine reservoirs studied were of better average trophic status, with average TSI values ranging from 41 to 65. Three reservoir were regarded as mesotrophic (TSI 41 to 50), three as eutrophic (TSI 51 to 60) and three as highly eutrophic (TSI 61 to 70). The TSI for chlorophyll a was in most cases below average. This result, however, could be due to the reduced number of sampling sessions and the fact that not all phytoplankton peaks were possibly recorded. When using the TSI for total phosphorus instead of the average TSI, the reservoirs' TSI (TP) ranged from 64 to 77 for Reservoirs 1 to 7 and from 45 to 68 for Reservoirs 8 to 16. In summary this indicated: one mesotrophic, four eutrophic, six highly eutrophic and five heterotrophic reservoirs.

3.4.7 Carlson index and correlations

Comparisons among morphometric data, water quality parameters and the trophic status index were evaluated for 16 reservoirs using correlation matrices. The Carlson average TSI correlated significantly with pH at a depth of 2 m (R^2=0.77, $p<0.01$), with the total phosphorus level near the bottom (R^2=0.80, $p<0.01$) and with chlorophyll a levels (R^2=0.67, $p<0.01$). Of the above three parameters, chlorophyll a concentration was not an independent parameter since it feeds into the Carlson TSI calculation. Additionally, TSI (Chl a) had the strongest influence on the Carlson average, followed by TSI(TP) and TSI(SD). Other parameters that correlated significantly with the Carlson average were the total depth of the reservoir (R^2=-0.055, $p<0.05$), total phosphorus levels at a depth of 2 m (R^2=0.62, $p<0.05$), ammonium levels near the bottom (R^2=0.56, $p<0.05$), alkalinity (R^2=0.58, $p<0.05$) and hardness (R^2=0.52, $p<0.05$). The total phosphorus concentration was an dependent parameter.

Results for seven intensively studied reservoirs, only showed two strong correlations, namely chlorophyll a levels (R^2=0.99, $p<0.01$) and the surface area of the reservoir (R^2=0.99, $p<0.01$). Total phosphorus near the bottom also correlated with the average Carlson TSI (R^2=0.95, $p<0.05$).

Considering all water quality parameters at all 16 sites, the two parameters that correlated with most other parameters were pH and total phosphorus at a depth of 2 m. Therefore, these two parameters can be used to predict water quality. Interestingly enough, there was no correlation ($p>0.05$) between Secchi depth and chlorophyll a levels, however a strong correlation existed between Secchi depth and total suspended solids (R^2=-0.72, $p<0.01$). This indicates that inflowing sediment material strongly influences and regulates the transparency of the open water systems even when compared to excessive algal growth.

3.4.8 Phytoplankton

Most estimations of trophic state postulate a strong relationship between phosphorus concentrations and algal growth. This relationship was also found for the seven intensively studied sites (Figure 3.11A). Using an exponential curve to describe the data points (best R^2 values), results indicted that the higher the phosphorus levels exceeded a certain limit, the greater the phytoplankton peaks. Algal growth exploded exponentially when phosphorus concentrations exceeded 100 µg/L. A doubling (2.5-fold) of phosphorus concentrations from Reservoir 1 to Reservoir 6 meant a 16-fold higher algal biomass. Reservoir 7 (as indicated by the square symbol) was an exception with reduced chlorophyll a levels which could be explained by the higher TSS average in this reservoir. Figure 3.11B shows the results of total phosphorus versus chlorophyll a for all 16 studied sites.

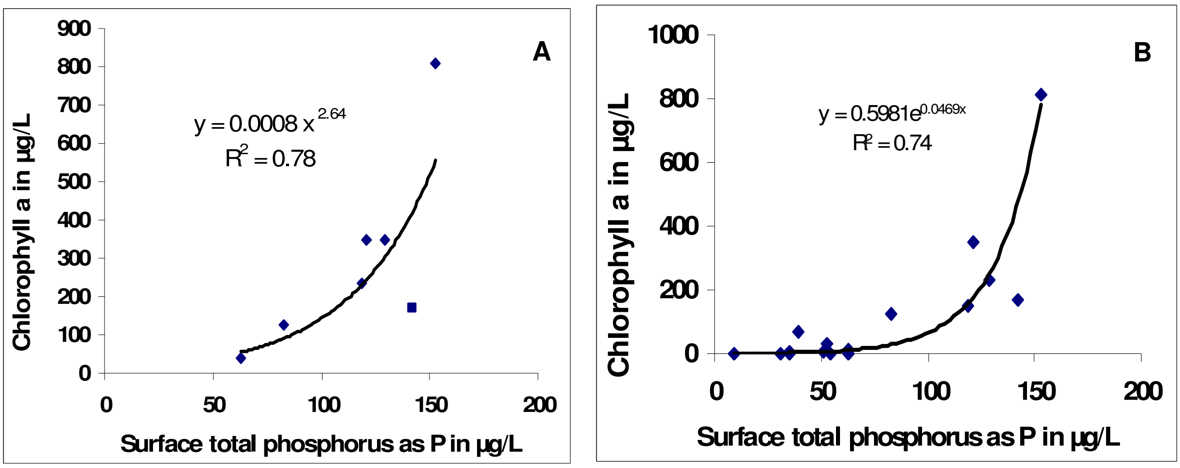

Figure 3.11 (A) Plot of surface total phosphorus against chlorophyll a at the seven intensively studied sites (R^2=0.78) (Diamonds represent Reservoir 1 to 6 and the square represents Reservoir 7). (B) Plot of surface total phosphorus against chlorophyll a at all 16 sites (R^2=0.74).

The main factors that influence phytoplankton growth are nutrient availability, light intensity and temperature conditions. Within the Western Cape reservoirs, nutrient availability was highest during the winter period (late March to September) when the reservoirs were inclined to mixing. The longest sunshine hours per day and highest sunlight radiation occurred during the summer period, from October to early March. However, high UV-b radiation can negatively influence phytoplankton growth. The levels around Cape Town were highest from November to February ranging from 20 and 34 MED (minimal erythemal dose) (DEAT 2006). At water temperatures higher than 22 °C, algae will be primarily nutrient limited (in contrast to temperature and light limited). This surface water temperature was exceeded from October to March, while in winter the water temperature decreased to 10 to 14 °C with minimum levels in June and July. Ideal periods for phytoplankton growth would therefore be the periods when light intensity starts to increase and nutrients become available by mixing (around August or September) and probably again when nutrients are still available and sunlight radiation decreases from the most destructive levels (around March and April).

Table 3.8: Additional parameters used for phytoplankton description (DIN = dissolved inorganic nitrogen, TN = total nitrogen, TP = total phosphorus, TIN = total inorganic nitrogen, TIP = total inorganic phosphorus, Chl a = chlorophyll a).

Parameter	Unit	R 1	R 2	R 3	R 4	R 5	R 6	R 7
Silica (2m)	mg/L	7.4	2.6	14.2	--	10.4	4.7	--
DIN: silica		0.16	0.10	0.01	--	0.01	0.10	--
TN:TP (weight)		13.5	10.6	3.4	--	4.8	9.4	2.1
TIN:TIP (weight)		19.8	9.0	2.6	8.0	3.0	6.0	1.1
Chl a max	µg/L	40	127	235	350	350	810	170

Other factors used to describe phytoplankton presence can be seen in Table 3.8. Silica levels are important in triggering the presence of Bacillariophytes (diatoms). In all reservoirs where silica levels were determined, a limitation of silica was excluded due to the low dissolved inorganic nitrogen against silica ratio. In these reservoirs, nitrogen was a limiting factor instead of silica. If Bacillariophytes are low in number, other factors will suppress or outcompete their presence.

The TN:TP ratio showed that nitrogen, as opposed to phosphorus, was the likely limiting factor to phytoplankton growth with the ratio of the inorganic components verifying this finding (Table 3.8). Nitrogen can be limiting at TN:TP ratios from 15 or less, while phosphorus limiting can occur from ratios of seven and higher. Algal growth in Reservoir 1 was therefore affected by phosphorus limitation.

Species from eight classes of the Phylum Algae were found in the reservoirs in the Western Cape (in some reservoirs the xanthophyceae occurred as a ninth class, however, only in low numbers and biomass). The contribution of each of these classes per reservoir can be seen in Figure 3.12. The reservoirs were sorted according to the average Carlson TSI. Reservoir 1 and Reservoir 2 showed the most even distribution among the classes, thereby verifying that the average Carlson TSI could generate reliable information on the trophic status of the studied subtropical reservoirs. In both reservoirs, bacillariophytes, chlorophytes, cryptophytes and cyanophytes, and additionally dinophytes in Reservoir 2, were quite balanced. In Reservoirs 3 and 4, the chlorophytes dominated the overall biomass with a 15 % proportion of cyanophytes present in Reservoir 3. Reservoirs 5 to 7 showed a strong trend towards dinophyte dominance (more than 40 % of the biomass), while other important groups were the chlorophytes and cyanophytes. The dominance of dinophytes rather than cyanophytes in the reservoirs of lower trophic state could be partly explained by a presumed carbon limitation. The given nitrogen limitation would enhance cyanophyte dominance due to their capability to fixate aerial nitrogen. The bulk biomass of algae occurred in August and September with a peak dominated by *Ceratium hirundinella*. During this period, mixing ceased, but nutrients were in easy reach for a species practicing vertical migration. In Reservoir 7, the dominating dinophyte species was replaced by *Peridinium sp*. Chlorophyte species that peaked during early spring were *Eudorina sp*. and *Volvox sp*., both migratory species with diel patterns.

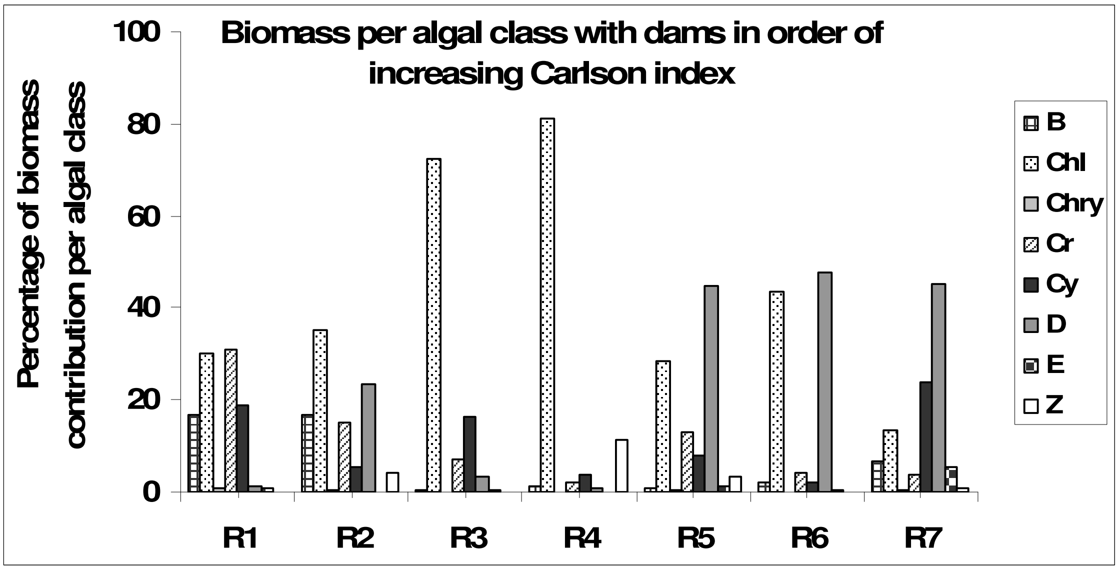

Figure 3.12: The relative contribution of each algal class to the total phytoplankton biomass per reservoir (Reservoir 1=R1 to Reservoir 7=R7; B=Bacillariophyceae, Chl=Chlorophyceae, Chry=Chrysophyceae, Cr=Cryptophyceae, Cy=Cyanophyceae, D=Dinophyceae, E=Euglenophyceae, Z=Zygnematophyceae).

In contrast to the tremendous differences in overall biomass among the classes, the number of species per class was fairly similar at all seven sites (Figure 3.13). Overall, species numbers found in the seven reservoirs varied from around 100 species (89 to 122) in Reservoirs 1 to 6 to 44 species in Reservoir 7.

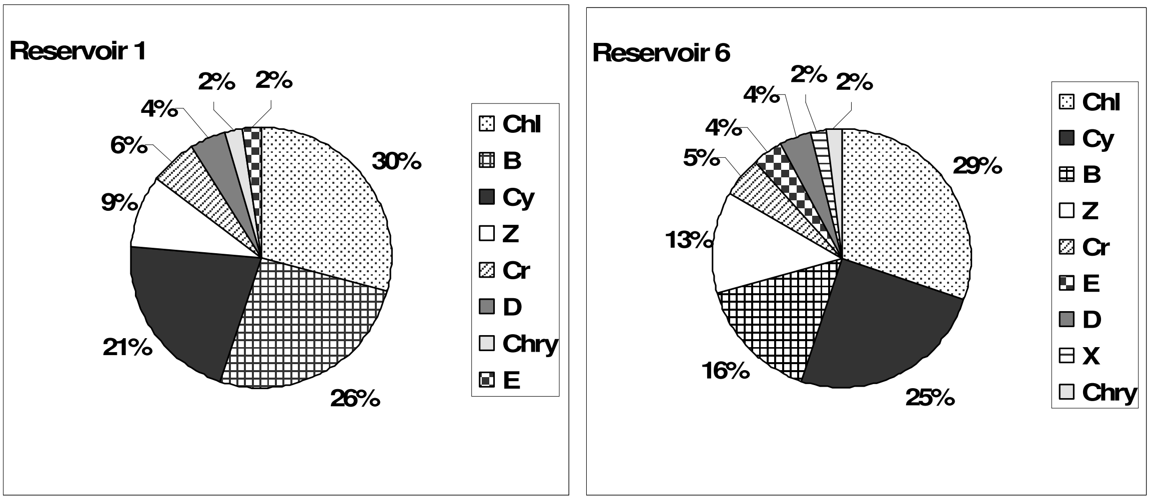

Figure 3.13: The relative contribution of species number per algal class in (A) Reservoir 1 and (B) Reservoir 6. (B=Bacillariophyceae, Chl=Chlorophyceae, Chry=Chrysophyceae, Cr=Cryptophyceae, Cy=Cyanophyceae, D=Dinophyceae, E=Euglenophyceae, X=Xanthophyceae, Z=Zygnematophyceae).

At Reservoir 1, the reservoir with the best trophic status according to Carlson, the biomass peaks during the course of the year amounted to a maximum of 1.3 mg/L (Figure 3.14). Some classes occurred concurrently during the peaks, but were primarily represented by one dominant species. In Reservoir 1 species included *Cryptomonas sp*. and *Rhodomonas sp*. for the cryptophytes, *Eudorina sp*., *Chlorococcum sp*., *Ankistrodesmus falcatus* and *Chlamydomonas sp*. for the chlorophytes, *Cyclotella sp*. for the bacillariophytes as well as *Aphanothece nidulans* and *Microcystis minutissima* for the cyanophytes. When comparing species number counted during a sampling period with the biomass during the same period (Figure 3.14 with 3.15), the highest species numbers occurred concurrently with the biomass peaks.

Overall biomass peaked in August/September (winter) or February/March (summer) (Reservoirs 3 to 7). In Reservoir 1 and 2, the number of peaks during the year was higher and showed an almost even distribution. Reservoir 2's highest biomass peak during the study period was 5 mg/L, in Reservoir 3 the highest peak was at 13 mg/L, in Reservoir 4 the peak was at 36 mg/L, in Reservoir 5 the peak was at 17 mg/L, in Reservoir 6 the peak was at 50 and 90 mg/L and in Reservoir 7 biomass peaked at 17 mg/L. The low biomass relative to phosphorus levels in Reservoir 7, already mentioned with the chlorophyll a levels, was probably reduced by suspended matter present specifically during the typical peaking period.

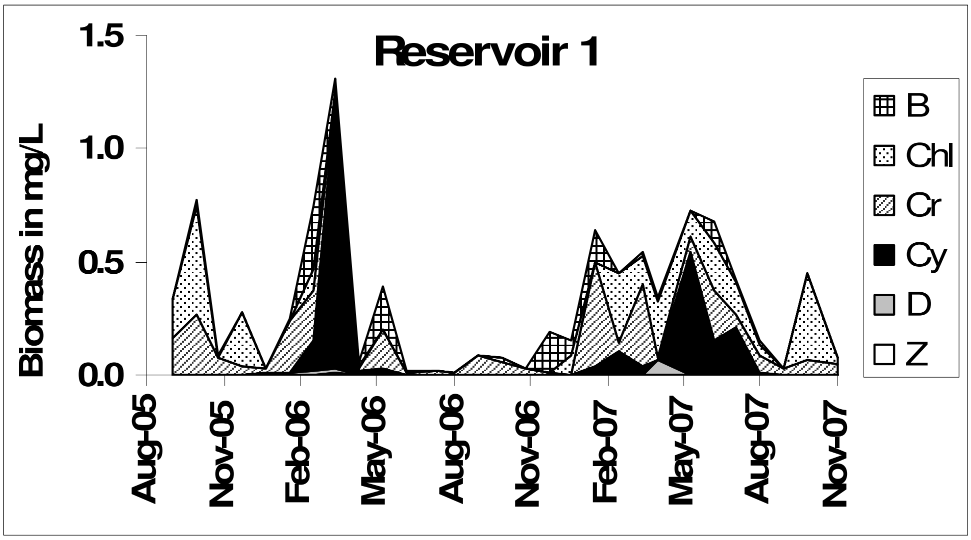

Figure 3.14: Six classes of typical phytoplankton representatives and their biomass distribution over time in Reservoir 1 (August 2005 to October 2007). Three phyla were neglected due to low biomasses (euglenophyceae, chrysophyceae and xanthophyceae). B=Bacillariophyceae, Chl=Chlorophyceae, Cr=Crysophyceae, Cy=Cyanophyceae, D=Dinophyceae, Z=Zygnematophyceae).

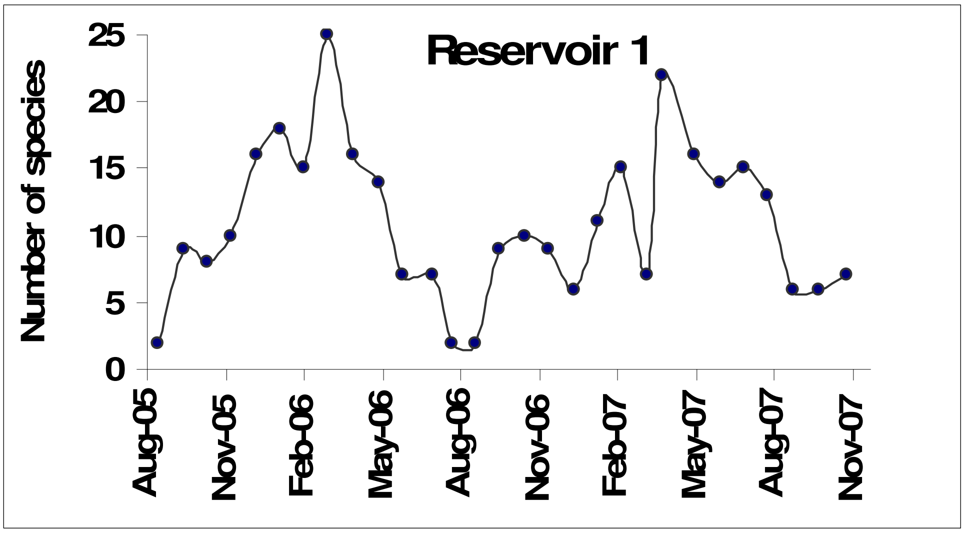

Figure 3.15: Number of phytoplankton species in Reservoir 1 from August 2005 to late October 2007.

Reservoir 6 was chosen as representative for a reservoir of elevated trophic status. In this reservoir biomass peaked at 50 and 90 mg/L with chlorophyte and dinophyte dominance (Figure 3.16). When peaking, concurrent species numbers were low (Figure 3.17), particularly when the dinophytes peaked. Less than 5 species were present when the dinophytes peaked in biomass. The same observation was made at Reservoir 2 and 5. Only two species dominated in Reservoir 6, namely *Volvox spermatosphaerum* or *C. hirundinella*.

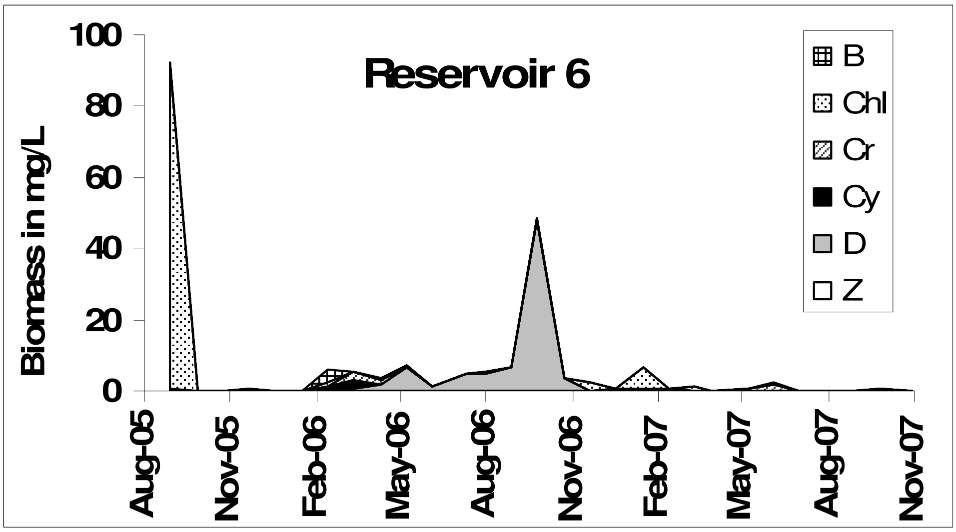

Figure 3.16: Six phytoplankton classes and their biomass distribution in Reservoir 6 from August 2005 to October 2007. Three classes were neglected due to low biomasses (euglenophytes, chrysophytes and xanthophytes). B=Bacillariophyceae, Chl=Chlorophyceae, Cr=Chrysophyceae, Cy=Cyanophyceae, D=Dinophyceae, Z=Zygnematophyceae).

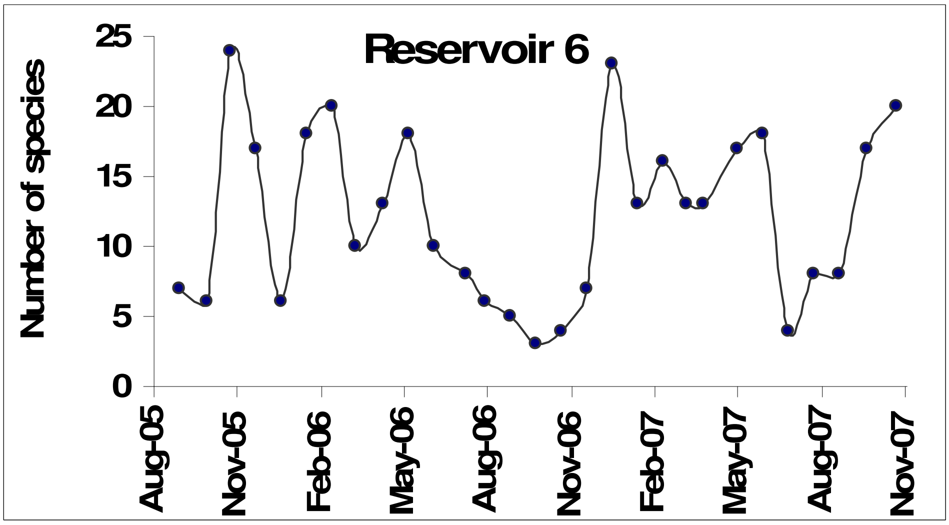

Figure 3.17: Number of phytoplankton species in Reservoir 6 from August 2005 to October 2007.

In addition to the phytoplankton biomass, Reservoir 6 was largely covered with a filamentous green algae starting in September with thick mats present by March.

The Shannon-Weaver index was calculated for all sites and varied between 1.43 and 1.52, almost equal at all sites. An exception was Reservoir 5 with a Shannon-Weaver average value of 1.97. The deviation of this index was highest in Reservoir 6 with 0.9 and at 0.7 at all other reservoirs. The Shannon-Weaver index describes the stability within phytoplankton communities on the species level and increases as more nutrients become available for algal growth (with higher eutrophication levels). Species compositions in all reservoirs can change extremely quickly and are relatively unstable.

3.4.9 Zooplankton

The zooplankton analysis did showed slight differences in the number of peaks among the five sites where zooplankton was sampled (Reservoirs 1, 2, 3, 5, 6). The standing biomass of zooplankton was higher than the phytoplankton biomass, with peaks of 40 mg/L in Reservoir 1, 80 mg/L in Reservoir 2, 20 mg/L in Reservoir 3, 90 mg/L in Reservoir 5 and 160 mg/L in Reservoir 6. Maximum biomass increased with trophic state of the reservoirs, with the exception of Reservoir 3 which had a relatively low standing biomass. Reservoir 3 was the only reservoir where many *Chaoborus sp*. larvae were found. It is possible that *Chaoborus sp*. controlled the residual zooplankton, in particular, the copepods in this reservoir.

The occurring peaks of zooplankton coincided with the phytoplankton peaks in August and September (especially in Reservoir 5 and 6 with only 1 peak per year). Reservoir 3 had additional intermediate peaks in March and April. In Reservoir 1 and 2, the zooplankton showed 3 to 5 peaks per season. The main food source for the copepods could have been smaller zooplankton groups (rotifers and protozoans) as well as particulate nutrients mixed into the surface water during the beginning of the overturn period. Another food source will certainly include bacteria (picoplankton) which were not considered in the current study.

Figure 3.18A shows the biomass contribution per zooplankton group. Quite apparently, the copepods dominated the biomass of all reservoirs, primarily consisting of Nauplius stages and cyclopoid copepodites. Cladocerans accounted for the second largest proportion, while dipteran larvae (*Chaoborus sp.*) replaced the latter in Reservoir 3. The cladocerans were primarily represented by *Bosmina longirostris*, *Daphnia sp*. and *Diaphanosoma sp*. Rotifers contributed to approximately 5 % of the total biomass of Reservoir 5 and 6.

With regards to species numbers (Figure 3.18B), rotifers exceeded all other groups in variety in all reservoirs followed by the protozoans. Cladocerans and copepods were similar in species numbers with slightly more species of cladocerans present (two to four species per reservoir).

Rotifer species abundance in five reservoirs varied between nine and 14 species. Most abundant species in terms of biomass were *Keratella quadrata*, *Brachionus calyciflorus* and *Pompholyx sulcata*. Species more abundant in Reservoir 1, with decreasing numbers towards Reservoir 6 were *Synchaeta* sp. and *Monostyla quadridentata*. Species with increasing abundance from Reservoir 1 to 6 were *Testudinella patina* and *Hexarthra mira*.

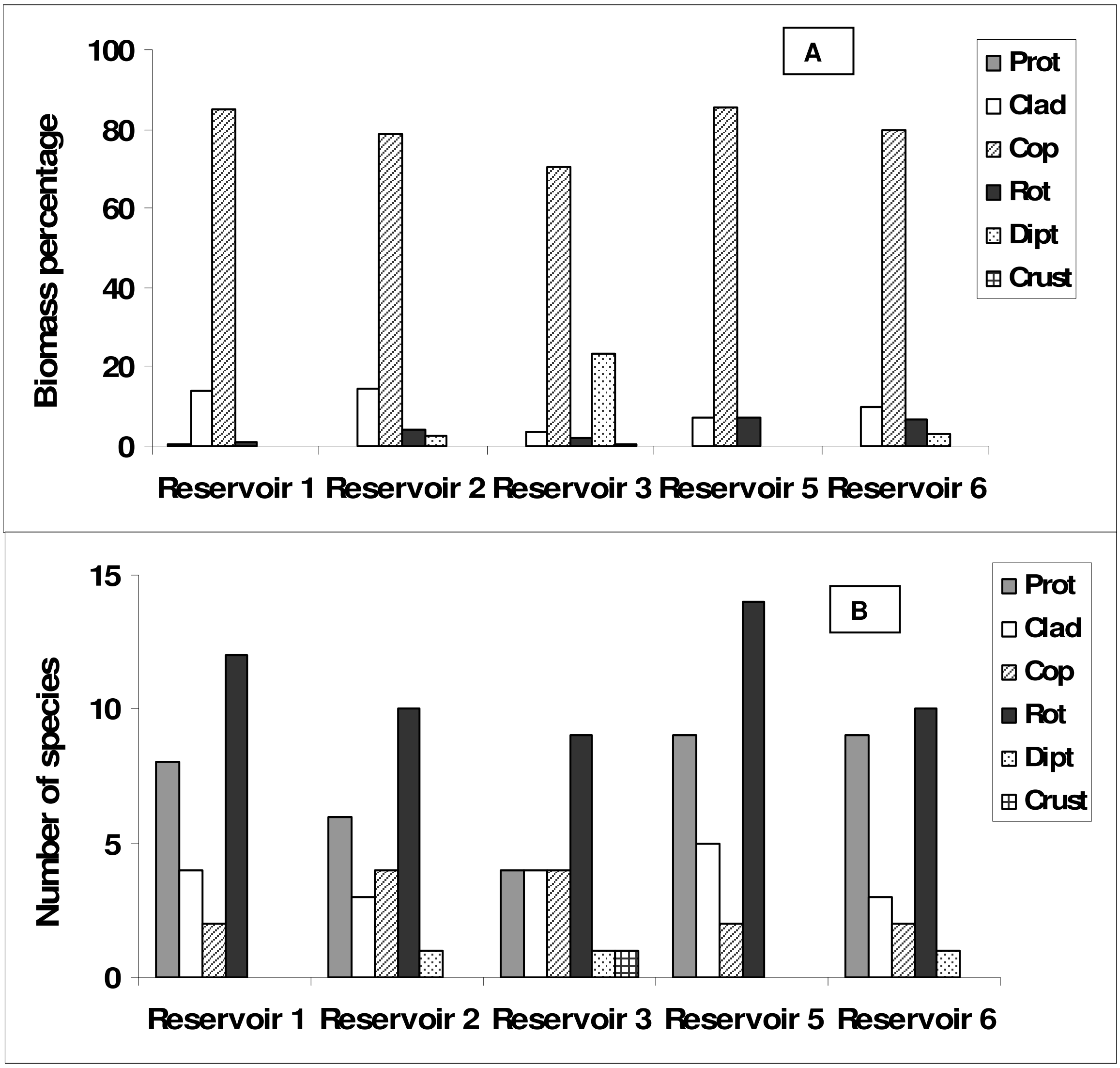

Figure 3.18: (A) The relative zooplankton biomass per group and (B) the absolute number of species per group per reservoir. Prot=Protozoa, Clad=Cladocera, Cop=Copepoda, Rot=Rotifera, Dipt=Dipteran larvae, Crust=Crustaceans.

The peaks of copepod and large phytoplankton species were simultaneous in Reservoirs 1 and 2 and rather independent in Reservoirs 3, 5 and 6. The annual peaks occurred from August to October and if more peaks occurred, they occurred in March. In Reservoir 6, a *C. hirundinella* bloom was followed by peaking copepod presence with a one month lag. The phenomenon of concurrent biomass peaks enhances the dinophyte and large chlorophyte dominance when copepods diminish any appearance of smaller edible species. During August to October, nutrients are consistently available due to strong winds and mixed water bodies and temperatures increase. Particulate matter and bacterial development nourished the copepods, while nutrients and particulate matter were the main food source for heterotrophic algal species. In Reservoir 1, *Euglenia sp.*, *Cyclotella sp.* and *Chlorococcum sp.* were the major phytoplankton species feeding copepods.

3.5 General discussion

The general discussion is dedicated to the analysis of the reservoirs as integrated systems. Oxygen distribution and content, physico-chemical conditions, nutrients and phyto- and zooplankton are important parameters when describing limnological ecosystems such as, for example, Western Cape irrigation reservoirs.

3.5.1 Thermal characteristics and oxygen distribution

The reservoirs in the study area are all warm monomictic which confirmed findings for reservoirs with >100 ha surface area in the north-eastern area of South Africa (Allanson & Gieskes 1961) and findings in lakes in Florida, USA (Beaver et al. 1981), Chile (Ramos et al. 1997) and New Zealand (Mitchell & Burns 1980). Interestingly, the 170 ha Bon Accord Dam in Gauteng, with similar water depths as the Western Cape Reservoirs (maximum depth 7.4 m), was described as warm dimictic (Van Ginkel et al. 2007), probably due to the high elevation of the reservoir (>1600 m a.m.s.l.) which is situated in the Highveld. With the monomictic Western Cape sites, different durations for the mixing period were established. Primarily dependent on reservoir minimum depth and respective weather conditions, a thermocline that inhibited mixing emerged from September to May in the deeper reservoirs (depth >9 m throughout summer) and from September to November with wind exposed shallow sites. Consequently, turnover phases can last from June to August and maximally from December to August. Circulation during the period when physical conditions permit lake turnover, is not necessarily a continuous process, but can be restricted to shorter periods or single events (instigated by heavy rainfall or strong winds), also described by Townsend (1999). In-between, “pseudo stabilisation” phases establish when the hypolimnion turns anoxic. This phenomenon was observed in three of seven reservoirs between January and May. In large reservoirs studied in KwaZulu-Natal (Hart 2001, Hart 1999), the period of lake mixis lasted from May to October, and therefore longer into spring than in the Western Cape. The difference in elevation levels and subsequently air temperature will be the reason for this, and the extended circulation period can presumably be transferred to sites at higher elevations in the Western Cape. The overall variations in timing and duration of water column circulation of the Western Cape sites reflect findings for 24 lakes in Florida, USA, with surface water temperature ranges similar to the Western Cape sites (Beaver et al. 1981). In summary, factors influencing differences among the studied sites were: reservoir size (maximum depth), reservoir management (minimum depth in summer, location and type of water inflow) and microclimate (exposure to wind, wind velocities and air temperatures).

Closely linked to water circulation processes is oxygen content and distribution in different water layers. In surface waters of Western Cape reservoirs, oxygen levels ranged from 6 to 14 mg/L, with lower oxygen levels as an exception in one highly eutrophicated reservoir (4 mg/L directly after breakdown of an algal peak). Surface oxygen levels above 12 mg/L were categorized as a sign of excessive algal growth and were observed in six of sixteen reservoirs. The supersaturation with oxygen coincided with large abundances of algae (oxygen production in the course of photosynthesis with concurrent carbon dioxide fixation). Phytoplankton triggered oxygen saturation at the surface and oxygen depletion at the lake bottom are joint processes and their association with eutrophication were recognized early (Vollenweider 1968).

Hypolimnetic decomposition processes (strongly dependent on organic material abundance and water temperature at the lake bottom) consume oxygen and whenever water circulation ceases, deplete oxygen resources. With most reservoirs in this study, oxygen reduction in the hypolimnion to levels below 1 mg/L occurred for up to 3 months during summer (between October and January). The only reservoir showing no oxygen depletion had a deep water supply system (pipe) and the highest wind velocities recorded for the period of the study. The reservoir in the most eutrophicated condition maintained a state of hypolimnetic anoxia for up to ten months. Five and nine months were also the periods of anoxia in two shallow Australian reservoirs (Townsend 1999) while phosphorus concentrations in these reservoirs were considerably low (6 and 10 µg/L). Hypolimnetic oxygen depletion is also a common phenomenon of large reservoirs studied in KwaZulu-Natal (Hart & Hart 2006). However, the 16 reservoirs in this study showed a clear trend towards prolonged anoxia with higher eutrophication levels, an observation often made with temperate systems (Bohle 1995). The fact that hypolimnetic anoxia can not be used as a global indicator for trophic status (due to temperature dependent processes – lower phosphorus concentrations will cause hypolimnetic anoxia in subtropical lakes when compared to temperate lakes), does not deny the support of the extent and duration of anoxia for regional comparisons (Townsend 1999).

3.5.2 Physical and chemical constituents

Periods of low water clarity (Secchi disk) occurred primarily in winter coinciding with elevated TSS levels, both influenced by strong rainfall events and the grain size of predominant soils in the catchment. Phytoplankton blooms influenced Secchi depth to a lesser extent in reservoirs of lower trophic status. A water clarity range of 100 and 200 cm found in the study reservoirs would, according to temperate studies correspond to phosphorus levels of 50 to 200 µg/L (eutrophied waters) (Jeppesen et al. 2005) which were concentrations found in this study. However, the given impact of suspended solids on Western Cape reservoirs impedes most comparisons and has implications on the strength of the TSI value for Secchi depth in this study (Havens 2000, Carlson 1983). Within South Africa, soil erosion in the Western Cape is categorized on a lighter scale, however a countrywide average of 2.5 t of soil are lost per hectare per year (DEAT 2006). Within the catchment areas of the study sites, most soil lay bare (vine, olives, orchards) which increases the risk of soil degradation. Soil erosion is also influenced by the amount of rainfall per hour (increasingly affecting from 25 mm/hr) (Zachar 1982). Additionally, a trend towards a decreased number of rain days, no matter what increase in precipitation intensity, was discovered for many global areas (Nel 2008, SAWS 2007, Brunetti et al. 2001) which increases the pressure on soil protection.

The pH values showed strong correlations to lake trophic status. The reservoirs were primarily alkalinic with measured values up to 9.8 in summer. These extreme values were accompanied by peaks of algal biomass (photosynthesis and carbon dioxide/carbonate ratio), but levels up to 9 occurred throughout in the warmer period. Davies (1997) studied 28 sites with surface areas ranging from 0.02 to 5.70 ha near Elgin and found pH values of 6.8 to 9.3.

Phytoplankton growth consumes carbon dioxide and shifts the pH towards extremely alkaline conditions (biologically produced alkaline pH) during the day, while dropping at night. When phytoplankton growth is reduced (e.g. in winter in mesotrophic conditions) and the pH levels are moderately alkaline, the mixing of acidic hypolimnion and alkaline epilimnion causes almost neutral to even slightly acidic conditions. Reservoirs with primarily natural fynbos surroundings were acidic throughout the year, influenced by the organic acids in the fynbos material. Acidic surface pH is the natural pH for fynbos influenced Western Cape water bodies of low nutrient status (DWAF 1996).

In contrast, the pH in bottom waters was always acidic, especially during lake stratification. Superfluous nutrient conditions can serve as an explanatory model. They enhance acidic conditions at the lake bottom through the activity of anaerobic bacteria decomposing organic material. This would also explain why the reservoir with no anoxic hypolimnion was the only reservoir with alkaline hypolimnetic water conditions. Differences between the epilimnion and hypolimnion in a reservoir were highest with elevated levels of eutrophication.

The low alkalinity and hardness (very low buffering capacity) of the Western Cape reservoirs, leaves these ecosystems exposed to extreme pH oscillations and with ongoing eutrophication this can have severe effects on aquatic biota (DWAF 1996). In conclusion it can be stated that the temperature and pH conditions in the study area favoured the presence of generalistic, tolerant species that coped with rapid changes in their environment (TSS) and in fact, primarily cosmopolitan phytoplankton species were found.

In accordance with the low hardness and alkalinity values and low conductivity, all quantified trace metals and mineral salts occurred at rather limiting levels and no readings of critically elevated amounts occurred.

3.5.3 Nutrients and TSI

The phosphorus levels in the reservoirs varied with the surrounding land use and the underlying rock type. The reservoirs located on shale and sandstone formations (Reservoirs 1, 2, 6, 9, 11 and 15) had lower phosphorus levels than the reservoirs located on ground dominated by granite material (Reservoirs 3, 4, 5, 7, 10). A newly constructed reservoir (Reservoir 12) and a reservoir of much larger surface area than the other sites (Reservoir 8) were exceptions. Reservoirs situated at higher altitudes (Reservoirs 13, 14 and 16) with no other land use in the catchment other than natural vegetation (primarily fynbos) were defined by relatively low total phosphorus levels between 36 and 45 µg/L as P, which counts as lightly eutrophic according to OECD (1982). The shale sites had TP levels between 60 and 80 µg/L and the granite sites between 120 to 150 µg/L. According to total phosphorus concentrations, these agriculturally influenced sites were categorized as medium to highly eutrophic (shale) and hypertrophic (granite) (Willen 2000, OECD 1982). In a hypothetical comparison of the sites without fish production at higher altitudes to the corresponding lowland sites of the same rock type (shale), the influence of agriculture on phosphorus content amounts is an approximate 65 to 75 % increase within 10 years (average minimum age of the reservoirs). This is a very conservative estimation since many reservoirs were even older than this and the estimation includes natural enrichment of organic material and nutrients from highland to lowland sites. A nutrient increase of 2 to 10 µg/L surface TP in the reservoirs was estimated per year. The addition ranges from an increase of 1 to 30 % to surface TP per year (with an average of 4 to 8 % TP increase for 16 reservoirs).

In particular, shale as parent rock weathers into a loam or clay loam texture of higher clay content than soils derived from granite and sandstone rocks (Gray & Murphy 1999). The finer sediment particles have better capacities to bind phosphorus. This could explain why the reservoirs of the subgroup granite jointly have higher water phosphorus concentrations than the reservoirs of the subgroup shale. Reservoir 6 of the subgroup granite is part of an additional cycle, where the reservoir water is used for cellar and storage facility cleaning and is pumped back into the reservoir. Usage of cleaning detergents could explain the higher phosphorus content in this reservoir.

The portion of soluble reactive phosphorus against total phosphorus was often shown to increase with higher phosphorus levels (Kerekes 1983) an effect which was not visible in the current study. Inflowing water quality is probably influencing inorganic phosphorus concentrations (including srp) so that srp experiences a greater variability throughout the year. The high variance of soluble reactive phosphorus fractions suggests that srp values alone would be a weak indicator for overall ecosystem status in these reservoirs.

Overall, the influence of agricultural practises on the eutrophication status of the reservoirs seems to be enormous and most reservoirs influenced by agriculture are at least eutrophic (3 of 16), often highly eutrophic (5 of 16) or hypereutrophic (5 of 16). The effect of the runoff of nutrients from overfertilised or eroding soils (as well as supposedly other factors) on water quality should be studied further and made a priority issue of surface water protection in South Africa. A publication on the status of the Berg River (de Villiers 2007) indicates agricultural runoff, overflowing municipal sewage treatment plants and un-sewered communities as main source of nutrients to surface waters. From the location of the reservoirs and surrounding land use, agricultural runoff will be the main single source of nutrients to most reservoirs included in this study.

Nitrogen levels were often above the suggested minimum levels required to maintain healthy algal growth and to avoid nuisance growth at 0.1 to 0.5 mg/L respectively according to ANZECC and ARMCANZ (2000). In Florida lake systems (USA), 0.7 mg/L is suggested as an upper limit, with 1.0 to 1.1 mg/L in coloured lakes (Florida DEP 2003).

With free ammonia, a concentration of 0.015 mg/L would be the chronic effect value and a concentration of 0.100 the acute effect value, according to the South African Water Guidelines for Aquatic Ecosystems (DWAF 1996). Hence, the toxic ammonia levels occurring in the reservoirs exceeded the chronic effect value. Maximum concentrations were reached in warm summer conditions with concurrent high pH values. Ammonia is toxic to the respiratory system of fish and other aquatic life (ANZECC & ARMCANZ 2000). Ammonia concentrations increased at the reservoirs' hypolimnia during stagnation, especially in the reservoirs of higher trophic status. This signal of eutrophication has often been described in the literature (Beutel 2001, Wetzel 2001, Jones et al. 1982, Schindler 1981).

In relation to the uptake ratio of nutrients by plants, however, nitrogen was the limiting component in most reservoirs (see TN:TP). Total phosphorus present in concentrations higher than 100 µg/L is most likely no limiting factor (Brown et al. 2000, Loeb & Verdonschot 2007). Nitrogen control was therefore most common (four of six reservoirs) and the remaining two reservoirs were nitrogen or phosphorus controlled. Other likely controlling factors of phytoplankton growth in these reservoirs were: light conditions, UV radiation and TSS levels, as well as physiologically important constituents (e.g. Mg, Si).

The average biomass of Swedish lakes (Willen 2000) corresponds to the average biomass found in the 16 reservoirs as well as the maximum biomass found in Florida lakes, USA (Agusti et al. 1990). Brown et al. (2000) studied 360 temperate and Florida lakes and compared two models describing the phosphorus-chlorophyll relationship. Up to phosphorus levels of 100 µg/L, temperate and Florida lakes did not differ in biomass production. However, at higher levels the models predicted higher biomasses per unit phosphorus for temperate lakes. The current limitation of the South African dataset does not allow enough detail for such predictions, but verifies that the average biomass produced in the 16 reservoirs revolved around the same central values than the Swedish study (Table 3.9). In comparison to the Florida data, the South African maximum biomasses showed lower ranges, especially in the group of 50 to 100 µg/L total phosphorus (primarily shale group).

Table 3.9: Comparison of the relationship between phosphorus and phytoplankton biomass in Swedish, Florida (USA) and South African lakes. Average phytoplankton biomass was available for 60 Swedish lakes and maximum phytoplankton biomass for 165 lakes in Florida (USA). In parentheses: the number of reservoirs within that range.

TP	**Corresponding average biomass (May to Oct) according to Willen (2000) in 60 Swedish lakes**	**Average biomass (August to March) in 16 South African reservoirs**	**Maximum phytoplankton biomass at given phosphorus levels (regression analysis of 165 Florida Lakes) according to Agusti et al. (1990)**	**Maximum phytoplankton biomass in 16 South African reservoirs**
µg/L	**mg/L**	**mg/L**	**mg/L**	**mg/L**
< 6	< 0.1	--	0.08 - 0.15	--
6-12.5	0.1-0.5	--	0.15 - 0.63	--
12.5-25	0.5-1.5	0.05 (1)	0.63 - 1.0	0.2 (1)
25-50	1.5-2.5	0.05 - 3.65 (4)	1.0 - 6.3	0.2 – 6.6 (4)
50-100	2.5-5	0.19 - 3.08 (6)	6.3 – 63	0.6 – 10.3 (6)
> 100	> 5	1.23 - 9.36 (5)	> 63	15 – 81 (5)

These results suggest that the comparison of South African data with trophic state classifications primarily derived in temperate regions (OECD 1982, Vollenweider 1968, Carlson 1977, Willen 2000) can deliver adequate status descriptions without over- or underestimations.

According to the correlation analyses, the trophic state of Western Cape irrigation reservoirs is primarily defined by the surface area (representing overall size and volume), pH and the chlorophyll a levels as well as surface and hypolimnetic phosphorus values.

Carlson's indices were confirmed as a useful measure for trophic status, when single water quality parameters and TSI values delivered coherent assessments of trophic status. Carlson (1983) actually preferred a prioritized use of the three indices instead of an averaged value applied in regional studies (Osgood 1982). Phosphorus was regarded as the best predictor of trophic status which would, in this study, suggest a hypertrophic status for all seven reservoirs. According to TSI (TP), the current eutrophication status of 16 reservoirs was manifested as mesotrophic (6 %), eutrophic (25 %), highly eutrophic (38 %) or hypertrophic (31 %). It has been discussed as to whether subtropical reservoirs produce less phytoplankton biomass at similar phosphorus levels when compared to temperate waters, and therefore need a different scaling of the trophic state than the Carlson average. In fact, in many reservoirs (three of seven) TSI (Chl a) stayed below the TSI (TP) or the average TSI value. Havens (2000), using the concept formalized by Carlson (1991), suggested TSI (Chl a)<TSI (TP) as an indication that phosphorus is not the limiting nutrient in the system (confirmed by the TN:TP ratios). In contrast, Reservoir 1 was phosphorus limited with TSI (Chl a)>TSI (TP) and was in fact near the TN:TP ratio, suggesting phosphorus limitation. However, with cyanobacterial dominance, the P limitation can be overestimated (Havens 2000). Similarly, when TSI (Chl a)<TSI (SD), Havens (2000) postulated that colour (or TSS) influences the Secchi depth or chlorophyll a concentrations were underestimated due to the abundance of picoplankters (Reservoirs 3, 5 and 6), while Reservoir 1 with TSI (Chl a)>TSI (SD) was dominated by large phytoplankton species (which were large patches of *Aphanothece sp.*). However, Reservoir 6 was also often dominated by large species (*C. hirundinella* and *V. spermatosphaerum*), which suggests that water clarity (and the secchi depth reading) was probably influenced by suspended solid presence.

3.5.4 Phytoplankton and zooplankton

As for the predictive power of chlorophyll a levels from total phosphorus concentrations, an exponential fit was established in this study, for the range of 0 to 150 total phosphorus µg/L as P. Brown (2000) showed a sigmoid dependency of chlorophyll a levels on phosphorus when using Florida (USA) and north-temperate lake data. In Brown's study there was an almost linear section between 8 and 76 µg/L TP and the bend towards the horizontal asymptote was only attained at total phosphorus values between 200 and 300 µg/L. It was probably not yet reached with a maximum phosphorus level of 153 µg/L computed in this study, which is why the best fit was exponential, not sigmoid.

Differences among reservoirs of different trophic status were number, timing and extent of biomass peaks and the relative occurrence of certain algal classes. More eutrophicated reservoirs tended towards higher maximum biomass levels (4 to 81 mg/L). The reservoirs of higher nutrient status had only had one biomass peak per year, whereas the mesotrophic reservoirs had patterns of three to four almost even peaks. The controlling algal groups also decreased to one at a time, either chlorophytes or dinophytes, instead of an even mixture of species from several taxonomic classes. Additionally, the number of species occurring during maximum peaks per group decreased from two or three to one. The physico-chemical water conditions of reservoirs with fluctuating water levels and large pH oscillations favour ubiquitous, euryecological species. Overall, the resilience of the phytoplankton community to changes decreases and dominance of single species is favoured in more eutrophic conditions.

A low abundance of cyanophytes was observed in the reservoirs of highly eutrophic and hypertrophic status. The nitrogen limitation indicated by the TN:TP ratios of most reservoirs would also suggest a small advantage for cyanophytes since they can fixate up to 10 % of their nitrogen content from aerial nitrogen (Ferber et al. 2004). However, all sites (except Reservoir 6) had short periods of cyanobacterial domination independent of phosphorus or nitrogen limitation. Xie et al. (2003) found that a bloom of *Microcystis sp*. can cause rather than result from phosphorus limitation. However, all reservoirs (5 out of 12) with an elevated nutrient status and algal biomass peaks of 10 mg/L and higher, show the same pattern of dinophyte dominance. Despite the advantage of cyanophytes with N limitation, there are obviously advantages of dinophyceae (phytoflagellates) over species of other algal classes. The species with most presence and biomass is *C. hirundinella*, followed by *Glenodinium sp*. and *Gymnodinium sp*.

One advantage will be the dinophytes' ability to obtain nutrients through phagotrophy in times of inorganic nutrient shortage (*Peridinium sp., Ceratium hirundinella*). Also, in intensive light conditions they will profit from the capability to acclimate rapidly in terms of chloroplast accumulation. Once populations of dinoflagellates established, blooms recurred which might be due to their resting stage strategy which was also observed in Argentine lakes (Mac Donagh et al. 2005). Large quantities of temporary cysts were found near the sediment of reservoirs with dinoflagellate blooms which allow the respective species an early start in the overall population dynamics. Jeppesen et al. (2005) in their study of 35 Swedish lakes observed dinoflagellates dominating in deeper lakes (deep enough to allow stratification) when the hypolimnion was rich in phosphorus. High initial phosphorus levels could be a favourable condition for cells to emerge from cysts or for young cells to proliferate. Lastly, it is discussed as to whether low dissolved inorganic carbon (DIC) concentrations favour dinoflagellates. The DIC content was not evaluated in this study, but can be assumed to be relatively low due to the low alkalinity. In marine dinoflagellates, their advantage proved to be the facultative carbon acquisition from carbonate cations as well as carbon dioxide and with higher pH levels this advantage increases (Rost et al. 2006). In conclusion, there are more factors that need to be studied to understand the dinoflagellate dominance in most highly eutrophicated reservoirs in the Western Cape. However, what can be stated is that once dinophyceae dominate the population structure, they successfully suppress coexisting algal species to low numbers and abundance.

The phenomenon of *C. hirundinella* dominance in highly eutrophic impoundments has been observed in South Africa (Hart 2006, Van Ginkel et al. 2001). In Spain (Perez-Martinez & Sanchez-Castillo 2001) a preference of water bodies with higher Mg and Ca contents and elevated alkalinity was found, while in Australia, *C. hirundinella* blooms were associated with warm water conditions (>20 °C) and peaked in January (Whittington et al. 2000). Within the Western Cape reservoirs, *C. hirundinella* blooms occurred from September to November (lake turnover) and therefore concurred with the availability of nutrients. However, *C. hirundinella* bloomed in cooler water conditions of 10 to 15 °C in waters with nutrient availability in winter (aquaculture production), which stresses the importance of nutrient availability rather than temperature. The optimum temperature for *C. hirundinella* ranges widely (5 to 30 °C) according to Buck (1989). Van Ginkel et al. (2007) found that the initial conditions for *C. hirundinella* growth were temperatures between 5 and 17 °C as well as inorganic phosphorus concentrations below 82 µg/L, while total nitrogen levels controlled the final biomass of the bloom. Dinophyte blooms cause filter clogging and odour problems which has no implications on health risks to consumers, but causes financial losses (Van Ginkel et al. 2007).

The zooplankton structure was clearly dominated by copepod species, which was also observed in subtropical Florida reservoirs (Havens et al. 2007). Reservoirs 1 and 2 showed more peaks in the course of one year than the more eutrophicated Reservoirs 5 and 6, and the absolute biomass peaks increased with a more elevated nutrient status which represents a typical development with eutrophication (Bohle 1995, Sommer 1994, Bays & Crisman 1983). The linkage between zooplankton and phytoplankton was synchronized, not consecutive. Most dominant phytoplankton species were large, motile chlorophytes or dinophytes and thus inedible for zooplankton. Particulate matter and bacterial biomass probably nourished the copepods, while nutrients and particulate matter were the main food sources for other species. Edible phytoplankters occurred only in the less eutrophicated reservoirs.

The species of rotators identified in these reservoirs indicated higher trophic status (Sládeček 1983) with the most abundant species *Keratella quadrata, Brachionus calyciflorus and Pompholyx sulcata* as indicators for intermediate trophic state (Baião & Boavida 2005). Overall, rotators seem not as suitable (species similar in all reservoirs) for trophic state determination in reservoirs as they are in river systems.

3.6 Conclusions

The small reservoirs (<20 ha) in the Western Cape are primarily of highly eutrophic to heterotrophic status (combined 62.5 %), according to average water quality parameters, Carlson TSI and phytoplankton succession and biomass. The natural phosphorus concentrations of the reservoirs without fish production at higher altitude ranged from 35 to 45 µg/L, while all other sites were significantly influenced by further phosphorus input. Without fish production, agriculture activities are postulated to be the main influence (deducted from the location of the sites and exclusion of other known phosphorus sources). Non-point pollution (nutrient introduction) from runoff is most likely the main source. Reservoirs situated on a granite base (rock type) tended to significantly develop higher total phosphorus concentrations in the surface water (>100 µg/L as P) than shale and sandstone based basins (60 to 80 µg/L as P).

The best predictors of trophic status in Western Cape reservoirs were extent and duration of hypolimnetic oxygen during stagnation, total phosphorus levels in the surface water (especially during turnover), summer ammonia concentrations in the hypolimnion, the difference between epilimnetic and hypolimnetic pH during stagnation, the differences among Carlson TSI values and phytoplankton composition.

The low natural alkalinity and hardness enhance symptoms of eutrophic waters. Missing buffer mechanisms allow wide fluctuations of maximum pH levels (9.8 was common) during phytoplankton growth periods, especially when combined with warm surface water temperatures. With low nitrogen content, episodes of chronically or acutely toxic ammonia are rare, but short periods of acute toxicity in surface waters were monitored in two hypertrophic sites. The physico-chemical conditions and nutrient conditions in eutrophied water favour ubiquitous phytoplankton species as well as the dominance of single species.

Only in a minority of the reservoirs (approximately 19 %) were conditions attained that impact upon the health of aquatic biota (fish, water birds) and irrigation and drinking water quality. These are: surface water oxygen depletion, elevated free ammonia levels and algal blooms.

Dominance of cyanobacteria is rare (and no species known to release toxins were found), however, dinophyte blooms were common and caused more frequent filter clogging and odour problems.

Reservoirs undergo constant eutrophication even without anthropologenic influences, however very slowly (decades to centuries). Most reservoirs in the current study were estimated to be 10 to 20 years of age. In this time frame they often reached a highly eutrophic to hypertrophic state (> 250 µg surface TP/L). The nutrient additions into the studied reservoirs originated primarily from agricultural runoff and to a lesser degree from informal housing developments, and an approximation of 2 to 10 µg TP/L was added to the surface water of the lowland reservoirs per year, dependent on the steepness of the slope, the nature of the inflowing water structures and nutrient loading of the inflowing water (which is a collection of runoff from natural landscape and agricultural land). Therefore, the contribution of agriculture to the eutrophication of the reservoirs is high.

3.7 References

Agusti, S., Duarte, C. M., and Canfield, D. E. (1990). Phytoplankton abundance in Florida lakes: Evidence for the frequent lack of nutrient limitation. Limnology and Oceanography 35(1): 181-188.

Allanson, B. R. and Gieskes, J. M. T. M. (1961). An investigation into the ecology of polluted inland waters in the Transvaal. Hydrobiologia 18: 79-94.

ANZECC and ARMCANZ (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality. National Water Quality Management Strategy Paper No 4. Canberra, Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand.

APHA, AWWA and WEF (2005). Standard Methods for the Examination of Water and Wastewater. Standard Methods online. American Public Health Association, American Water Works Association, Water Environment Federation. http://www.standardmethods.org.

Baião, C. and Boavida, M. J. (2005). Rotifers of Portuguese reservoirs in river Tejo catchment: Relations with trophic state. Limnetica 24(1-2): 103-114.

Bays, J. S. and Crisman, T. L. (1983). Zooplankton and Trophic State Relationships in Florida Lakes. Canadian Journal of Fisheries and Aquatic Sciences 40(1813): 1819.

Beaver, J. R., Crisman, T. L., and Bays, J. S. (1981). Thermal regimes of Florida lakes. Hydrobiologia 83(2): 267-273.

Berg, R. R., Thompson, R., Little, P. R., and Görgens, A. H. (1994). Evaluation of farm dam area-height-capacity relationships required for basin-scale hydrological catchment modelling. Water SA 20(4): 265-272.

Beutel, M. W. (2001). Oxygen consumption and ammonia accumulation in the hypolimnion of Walker Lake, Nevada. Hydrobiologia 466(1-3): 107-117.

Bohle, H. W. (1995). Limnische Systeme. Berlin: Springer-Verlag.

Brown, C. D., Hoyer, M. V., Bachmann, R. W., and Canfield, D. E. (2000). Nutrient-chlorophyll relationships: an evaluation of empirical nutrient-chlorophyll models using Florida and north temperate lake data. Canadian Journal of Fisheries and Aquatic Sciences 57(8): 1574-1583.

Brunetti, M., Maugeri, M., and Nanni, T. (2001). Changes in total precipitation, rainy days and extreme events in Northeastern Italy. International Journal of Climatology 21: 861-871.

Buck, H. (1989). Ecology of selected planktonic algae causing water blooms. Acta Hydrobiol. 31: 207-258.

Carlson, R. E. (1977). A trophic state index for lakes. Limnology and Oceanography 22: 361-369.

Carlson, R. E. (1983). Discussion on the article "Using differences among Carlson's trophic state index values in regional water quality assessment", by Richard A. Osgood. Water Resources Bulletin 19: 307-309.

Carlson, R. E. (1991). Expanding the trophic state concept to identify non-nutrient limited lakes and reservoirs. In Carpenter.L. (ed.), Proceedings of a National Conference on Enhancing the States' Lake Management Programs. United States Environmental Protection Agency, Chicago.

Davies, H. (1997). An Assessment of the Suitability of a Series of Western Cape Farm Dams as Waterbird Habitats. Thesis, Cape Town, University of Cape Town, Conservation Biology Department.

Day, J. A., Stewart, B. A., de Moor, I. J., and Louw, A. E. (1999). Crustacea I - Notostraca, Anostraca, Conchostraca and Cladocera. WRC Report No: TT 121/00. Guides to the Freshwater Invertebrates of Southern Africa - Volume 2. Pretoria, Department of Water Affairs and Forestry.

Day, J. A., de Moor, I. J., Stewart, B. A., and Louw, A. E. (2001). Crustacea II - Notostraca, Anostraca, Conchostraca and Cladocera. WRC Report No. TT 148/01. Guides to the Freshwater Invertebrates of Southern Africa - Volume 3. Pretoria, Department of Water Affairs and Forestry.

DEAT (2006). South Africa Environment Outlook. A report on the state of the environment. Pretoria, Department of Environmental Affairs and Tourism.

DEADP (2007). Spatial Data Catalogue: geology, geology lithos and soil types. Pretoria, Department of Environmental Affairs and Development Planning.

DWAF (1996). South African Water Quality Guidelines. Volume 7: Aquatic Ecosystems. Pretoria, Department of Water Affairs and Forestry.

DWAF (2007). Unpublished data on some major irrigation dams in the Western Cape. Pretoria, Department of Water Affairs and Tourism.

Entwisle, T. J., Sonneman, J. A., and Lewis, S. H. (1997). Freshwater Algae in Australia - a guide to conspicuous genera. Potts Point, Australia: Sainty and Associates Pty Ltd..

EPA (2003). National Management Measures to Control Nonpoint Source Pollution from Agriculture. US EPA 841-B-03-004. Washington D.C., US Environmental Protection Agency.

Ettl, H. and Gärtner, G. (1988). Chlorophyta II (Tetrasporales, Chlorococcales, Gloeodendrales). Stuttgart, New York: Gustav Fischer Verlag.

Ferber, L. R., Levine, S. N., Lini, A., and Livingston, G. P. (2004). Do cyanobacteria dominate in eutrophic lakes because they fix atmospheric nitrogen? Freshwater Biology 49(6): 690-708.

Florida Department of Environmental Protection (2003). Development of Florida Lake Nutrient Criteria: Summary and Synthesis. Tallahassee, USA.

Gonzalez, E. J., Matsumura-Tundisi, T., and Tundisi, J. G. (2008). Size and dry weight of main zooplankton species in Bariri reservoir (SP, Brazil). Brazilian Journal of Biology 68(1): 69-75.

Gray, J. M. and Murphy, B. W. (1999). Parent Material and Soils: a guide to the influence of parent material on soil distribution in Eastern Australia. DLWC Technical Report Number 45. Department of Land and Water Conservation. Sydney, Department of Land Water and Conservation.

Hart, R.C. (1999). On the limnology of Spioenkop, a turbid reservoir on the upper Thukela River, with particular reference to the structure and dynamics of its plankton community. Water SA 25: 519-528.

Hart, R.C. (2001). A baseline limnological study of Wagendrift Dam (Thukela basin, KwaZulu-Natal). Water SA 27(4): 507-516.

Hart, R. C. (2006). Phytoplankton dynamics and periodicity in two cascading warm-water reservoirs from 1989 to 1997 - taxonomic and functional (C-S-R) patterns, and determining factors. Water SA 32(1): 81-92.

Hart, R. and Hart, R. C. (2006). Reservoirs and their management: a review of the literature since 1990. WRC Report No. KV 173/06. Pretoria, Department of Water Affairs and Forestry.

Häusler, J. (1982). Schizomycetes. Gustav Fischer Verlag, Stuttgart.

Havens, K. E. (2000). Using Trophic State Index (TSI) Values to Draw Inferences Regarding Phytoplankton Limiting Factors and Seston Composition from Routine Water Quality Monitoring Data. Korean Journal of Limnology 33(3): 187-196.

Havens, K. E., Beaver, J. R., and East, T. L. (2007). Plankton biomass partitioning in a eutrophic subtropical lake: comparison with results from temperate lake ecosystems. Journal of Plankton Research 29(12): 1087-1097.

Hillbricht-Illkowska, A. and Pieczynska, E. (1993). Nutrient dynamics and retention in land/water ecotones of lowland, temperate lakes and rivers. Dordrecht: Springer Netherland Journals.

Huber-Pestalozzi, G. (1938). Das Phytoplankton des Süßwassers - 1. Teil Allgemeiner Teil, Blaualgen, Bakterien, Pilze. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Huber-Pestalozzi, G. (1941). Das Phytoplankton des Süßwassers - 2.Teil,1. Hälfte Chrysophyceen, farblose Flagellaten, Heterokonten. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Huber-Pestalozzi, G. (1950). Das Phytoplankton des Süßwassers - 3. Teil Cryptophyceen, Chloromonadinen, Peridineen. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Huber-Pestalozzi, G. (1955). Das Phytoplankton des Süßwassers - 4. Teil Euglenophyceen. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Huber-Pestalozzi, G. (1961). Das Phytoplankton des Süßwassers - 5. Teil Chlorophyceae, Volvocales. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Huber-Pestalozzi, G. and Fott, B. (1972). Das Phytoplankton des Süßwassers - 6. Teil Chlorophyceae, Tetrasporales. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Huber-Pestalozzi, G. and Förster, K. (1982). Das Phytoplankton des Süßwassers - 8. Teil, 1. Hälfte Conjugatophyceae. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Huber-Pestalozzi, G. and Hustedt, F. (1942). Das Phytoplankton des Süßwassers - 2. Teil,2. Hälfte Diatomeen. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Jeppesen, E., Soendergaard, M., Jensen, J. P., Havens, K. E., Anneville, O., Carvalho, L., Coveney, M. F., Deneke, R., Dokulil, M. T., Foy, B., Gerdeaux, D., Hampton, S. E., Hilt, S., Kangur, K., Köhler, J., Lammens, E. H. H. R., Lauridsen, T. L., Manca, M., Miracle, M. R., Moss, B., Noges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Schelske, C. L., Straile, D., Tatrai, I., Willen, E., and Winder, M. (2005). Lake responses to reduced nutrient loading - an analysis of contemporary long-term data from 35 case studies. Freshwater Biology 50: 1747-1771.

Jones, G. B., Simon, B. M., and Horsley, R. W. (1982). Microbial sources of ammonia in freshwater lake sediments. Journal of general microbiology 128(12): 2823-2831.

Kadlubowska, J. Z. (1984). Conjugatophyceae I (Chlorophyta VIII - Zygnemales). Gustav Fischer Verlag, Stuttgart.

Kerekes, J. (1983). Predicting Trophic Response to Phosphorus Addition in a Cape Breton Island Lake. Proceedings of the Nova Scotian Institute of Science 33: 7-18.

Komarek, J. and Anagnostidis, K. (1998). Cyanoprokaryota - 1. Teil: Chroococcales. Stuttgart, New York: Gustav Fischer Verlag.

Krammer, K. and Lange-Bertalot, H. (1986). Bacillariophyceae - 1. Teil: Naviculaceae. Stuttgart, New York: Gustav-Fischer-Verlag.

Krammer, K. and H., Lange-Bertalot (1988). Bacillariophyceae - 2. Teil: Bacillariaceae, Epithemiaceae, Surirellaceae. Stuttgart, New York: Gustav Fischer Verlag.

Krammer, K. and H., Lange-Bertalot (1991a). Bacillariophyceae - 3. Teil: Centrales, Fragilariaceae, Eunotiaceae. Stuttgart: Gustav Fischer Verlag.

Krammer, K. and Lange-Bertalot, H. (1991b). Bacillariophyceae - 4. Teil: Achnanthaceae, Kritische Ergänzungen zu Navicula und Gomphonema. Stuttgart: Gustav Fischer Verlag.

Loeb, R. and Verdonschot, P. (2007). Processes determining limitation in aquatic systems - a review. Alterra, Wageningen University and Research Centre, Netherlands. http://www.cost869.alterra.nl/greece/Loeb.pdf.

Mac Donagh, M. E., Casco, M. A., and Claps, M. C. (2005). Colonization of a Neotropical Reservoir (Córdoba, Argentina) by Ceratium hirundinella (O. F. Müller) Bergh. Annales de Limnologie - International Journal of Limnology 41(4): 291-299.

Mason, C. (2002). The Biology of Freshwater Pollution. Upper Saddle River, New Jersey: Prentice Hall.

Mitchell, S. F. and Burns, C. W. (1980). Oxygen consumption in the epilimnia and hypolimnia of two eutrophic, warm-monomictic lakes. New Zealand Journal of Marine & Freshwater Research 14(2): 215.

Nel, W. (2008). Observations on daily rainfall events in the KwaZulu-Natal Drakensberg. Water SA 34(2): 271-274.

OECD (1982). Eutrophication of Waters. Monitoring, Assessment and Control. Organisation for Economic Co-operation and Development. Paris, Organisation for economic co-operation and development.

Osgood, R. A. (1982). Using differences among Carlson's trophic state index values in regional water quality assessment. Water Resources Bulletin 18: 67-74.

Pérez-Martínez, C. and Sanchez-Castillo, P. (2001). Temporal occurrence of *Ceratium hirundinella* in Spanish reservoirs. Hydrobiologia 452(1-3): 101-107.

Popovský, J. and Pfiester, L.A. (1990). Dinophyceae (Dinoflagellida). Stuttgart: Gustav Fischer Verlag.

Prescott, G. W. (1978). How to know the freshwater algae. C. Brown Company Publishers.

Ramos, R., Flores, F., Trapp, C., Siebeck, O. and Zuniga, L. (1997). Thermal, light and oxygen characteristics in a small eutrophic warm monomictic lake (El Plateado, Valparaiso, Chile). Verhandlungen - Internationale Vereinigung für theoretische und angewandte Limnologie 26(2): 256-260.

Rost, B., Richter, K.-U., Riebesell, U., and Hansen, P. J. (2006). Inorganic carbon acquisition in red tide dinoflagellates. Plant, Cell and Environment 29: 810-822.

SAWS (2007). Weather data retrieved for 2000 to 2007. Cape Town, South African Weather Service.

Schindler, D. E. (1981). Interrelationships Between the Cycles of Elements in Freshwater Ecosystems. In G. E. Likens (ed.), Some Perspectives of the Major Biogeochemical Cycles. Scientific Committee on Problems of the Environmen (SCOPE) 17.

Schwoerbel, J. (2005). Einführung in die Limnologie. München, Heidelberg: Elsevier, Spektrum Akademischer Verlag.

Sendacz, S., Caleffi, S., and Santos-Soares, J. (2006). Zooplankton biomass of reservoirs in different trophic conditions in the state of Sao Paulo, Brazil. Brazilian Journal of Biology 66(1): 337-350.

Sequi, P., Ciavatta, C., and Antisari, L. V. (1989). Phosphate fertilizers and phosphorus loadings to rivers and seawater. In I. IFA (ed.), The role of phosphates in balanced fertilisation. Marrakesh, Morocco.

Shannon, C. and Weaver, W. (1949). The Mathematical Theory of Communication. Urbana: University & Illinois Press.

Sladecek, V. (1983). Rotifers as indicators of water quality . Hydrobiologia 100(1): 169-201.

Sommer, U. (1994). Planktologie. Heidelberg: Springer.

Sommer, U. (1996). Plankton ecology: The past two decades of progress . Naturwissenschaften 83(7): 293-301.

Steedman, H. F. (1976). Zooplankton Fixation and Preservation. Paris: UNESCO Press.

Steyn, D. J., Toerien, D. F., and Visser, J. H. (1976). Eutrophication Levels of some South African Impoundments III. Roodeplaat Dam. Water SA 2(1): 1-6.

Thirion, C. (1999) Introduction to the practice of identifying Zooplankton. Volume 1: Rotifera. IWQS Report number N/0000/00/DEQ0799. Pretoria, Institute for Water Quality Studies, Department of Water Affairs and Forestry.

Thornton, J. (1987). A review of some unique aspects of the limnology of shallow Southern African man-made lakes. Geojournal 14(3): 339-352.

Thornton, J. A. (1989). Aspects of the phosphorus cycle in Hartebeespoort Dam (South Africa). Hydrobiologia 183: 87-95.

Tiessen, H. (1995). Phosphorus in the global environment: Transfers, cycles and management.
Toerien, D. F., Hyman, K. L., and Bruwer, M. J. (1975). A Preliminary Trophic Status Classification of some South African Impoundments. Water SA 1(1): 15-23.

Toerien, D. F., Hyman, K. L., and Bruwer, M. J. (1975). A Preliminary Trophic Status Classification of some South African Impoundments. Water SA 1(1): 15-23.

Tomczak, G. and Godfrey, J. S. (1994). Regional Oceanography: An introduction. Oxford: Pergamon Press.

Townsend, S.A. (1995). Metalimnetic and hypolimnetic deoxygenation in an Australian tropical reservoir of low trophic status. In: Timotius,K.H. and F.Goltenboth (eds), Tropical Limnology. Vol II. Tropical Lakes and Reservoirs.

Townsend, S. A. (1999). The seasonal pattern of dissolved oxygen, and hypolimnetic deoxygenation, in two tropical Australian reservoirs. Lakes & Reservoirs: Research & Management 4(1): 41-53.

Utermöhl, H. (1958). Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. Mitteilungen der internationalen Vereinigung der theoretischen und angewandten Limnologie 5: 567-596.

van den Hoek, C., Mann, D. G., and Jahns, H. M. (1995). Algae - an introduction to phycology. Cambridge: Cambridge University Press.

van Ginkel, C. E., Hohls, B. C., and Vermaak, E. (2001). A *Ceratium hirundinella* (O.F. Müller) bloom in Hartbeespoort Dam, South Africa. Water SA 27(2): 269-276.

van Ginkel, C. E., Cao, H., Recknagel, F., and du Plessis, S. (2007). Forecasting of dinoflagellate blooms in warm-monomictic hypertrophic reservoirs in South Africa by means of rule-based agents. Water SA 33(4): 531-538.

Vörös, L. and Padisak, J. (1991). Phytoplankton biomass and chlorophyll-*a* in some shallow lakes in central Europe. Hydrobiologia 215(2): 111-119.

Vollenweider, R. A. (1968). Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. OECD Technical Report DA5/SU/68-27. Paris: Organisation for economic co-operation and development (OECD).

Wetzel, R. G. and Likens, G. E. (2000). Limnological Analyses. New York: Springer.

Wetzel, R. G. (2001). Limnology: Lake and River Ecosystems. San Diego: Elsevier.

Whittington, J., Bradford, S., Green.D., and Oliver, R. L. (2000). Growth of *Ceratium hirundinella* in a subtropical Australian reservoir: the role of vertical migration. Journal of Plankton Research 22(6):1025-1045.

WHO (1993). Guideline for Drinking Water Quality. Geneva: World Health Organization (WHO).

Willen, E. (2000). Phytoplankton in water quality assessment - an indicator concept. In P. Heinonen, G. Ziglio, and A. Van der Beken (eds.). Hydrological and limnological aspects of lake monitoring. John Wiley and Sons.

Xie, L., Xie, P., Li, S., Tang, H., and Liu, H. (2003). The low TN:TP ratio, a cause or a result of *Microcystis* blooms? Water Research 37(9): 2073-2080.

Yunfang, H. M. S. (1995). Atlas of Freshwater biota in China. Beijing: China Ocean Press.

Zachar, D. (1982). Soil erosion. Amsterdam: Elsevier.

CHAPTER 4 INFLUENCE OF TROUT CAGE AQUACULTURE (ONCORHYNCHUS MYKISS) ON WATER QUALITY AND PHYTOPLANKTON COMMUNITIES IN WESTERN CAPE IRRIGATION RESERVOIRS

Abstract

The water quality changes in 26 irrigation reservoirs with 5 t production units of rainbow trout in floating cage culture (mean surface area of reservoirs = 5.6 ha) were monitored and compared with the conditions in 16 reference reservoirs (mean surface area of reservoirs = 6.9 ha). A before-and-after study undertaken at one site strengthened the comparative analysis of reservoirs with trout production versus reservoirs without a production history. The environmental stocking density varied between 600 and 5000 kg/ha, with a maximum stocking density of 8.1 kg/m^3. The sites varied in altitude, underlying geology, surface area, water exchange rate and production history.

The differences between fish production and non-aquaculture reservoirs included increased surface phosphorus levels (68 to 144 µg/L as P; 112 % increase) at production sites, development of more pronounced pH fluctuations and increased maximum pH, especially during the summer period. In the hypolimnion the changes were most prominent during the summer period (stagnation) with increased duration of hypolimnetic hypoxia (2.3 to 4.6 months), increased total phosphorus concentrations in water collected at the sediment-water interface (98 to 263 µg/L as P, 168 %) and increased ammonia concentrations near the bottom (118 to 474 µg/L as N, 302 %). All production sites were classified as highly eutrophic to hypertrophic according to the Carlson TSI (TP).

The effects of aquaculture on phytoplankton composition were: increased algal biomass, fewer, but higher peaks, a shift from chlorophyte, dinophyte, cyanophyte and cryptophyte presence (90 % biomass) to chlorophyte and dinophyte dominance (80 %). Small species (10 to 30 µm) were replaced by larger species (> 100 µm). Accumulated *Ceratium hirundinella* occurrence characterised the most hypertrophic sites.

Reliable predictors for water quality change (eutrophication) were identified as bottom ammonia and bottom TP concentrations. Low impact (<15 % increase) regarding variance from conditions of the average reservoirs without fish production can be concluded for 15.3 % (four of 26) of the current production sites. Medium impact (approximately 50 % increase) occurred with production in 42.3 % of the reservoirs (11 of 26 sites). High impact with a doubling effect of

nutrients within one year (>100 % increase) occurred in 42.3 % of the current production sites (11 of 26) when looking at TP and total ammonia.

4.1 Introduction

Cage aquaculture, especially of carnivorous species, has been shown to influence water quality in open water systems (Gale 1999, Axler et al. 1996, Marsden et al. 1995, Stirling & Dey 1990). Generally, negative impacts of cage aquaculture derive from the introduction of artificial structures and fouling, the spread of diseases and parasites, the ecological impact of escapees, and the input of feed and chemicals (Beveridge 2004, Pillay 2004, Davenport et al. 2003). This paper concentrates on the latter, the introduction of nutrients into water systems with the most important source: feed and its particulate and soluble residues (Midlen & Redding 1998).

As a consequence of uneaten feed and faeces introduction, phosphorus and nitrogen concentrations increase in the water column and in the sediments. Water quality parameters can be primarily or secondarily affected, such as water transparency, pH and oxygen levels. Phytoplankton growth is often stimulated as well as communities changed. If and to what extent effects occur in cage aquaculture operations depends greatly on the underlying geology, intensity of production, the water volume and the water exchange rate (Boyd et al. 2001). Grain size and composition of sediment in reservoir basins can strongly affect nutrient binding capacities and overall nutrient budgets (Hayes & Anthony 1958).

The occurrence of many small water bodies mainly used for storage of summer irrigation water in a winter rainfall area, instigated the idea for multiple use of these water resources. A project was initiated in 1992, which was later privatised as “Hands-On Fish Farmer’s Co-operative Limited”. The goal was to develop and propagate rainbow trout (*Oncorhynchus mykiss*) cage farming in irrigation reservoirs of the Western Cape, to raise income and to empower previously disadvantaged groups of the population. With low capital investment, a ready domestic market for trout and lucrative market prices for salmonids, the project operates very successfully. The project supported one production unit from 1993 to 1995, three to four production units from 1996 to 2004, ten in 2005, 21 in 2006, six in 2007 and 23 in 2008 (a problem with fingerling supply caused the decrease in 2007).

The 5 t production units of rainbow trout in floating cages in small reservoirs (<20 ha) of the Western Cape were initially thought to be comparatively extensive (Rouhani & Britz 2004). However, some sites had to be abandoned due to emerging oxygen deficiencies and algal problems, while other production sites operated with continuing success. A severe impact on surface phosphorus was found with 28 ha and 56 ha mine pit lakes and a 15 t production of rainbow trout in Minnesota (USA) (Axler et al. 1996). However, a 20 t production unit in the 250 ha Lake Menteith in Scotland showed a low discernable impact (Marsden et al. 1995). The most related study concerned three South African reservoirs in Mpumalanga (former Eastern

Transvaal) where 30 to 40 t of rainbow trout were produced in 12 and 25 ha reservoirs. As a result, toxic algal blooms (*Anabaena* sp. and *Microcystis* sp.) and diurnal and seasonal oxygen fluctuations developed (Heath 1990).

To understand the ecological processes in more detail and to be able to quantify short-term and long-term impacts on the environment at the Western Cape sites, trout production reservoirs and non-production reservoirs (also referred to as production and reference sites) were compared. The main factors influencing the velocity and degree of change under the specific conditions were described. The results will help to support ecologically, economically and socially sustainable aquaculture (FAO 1995).

4.2 Study Sites

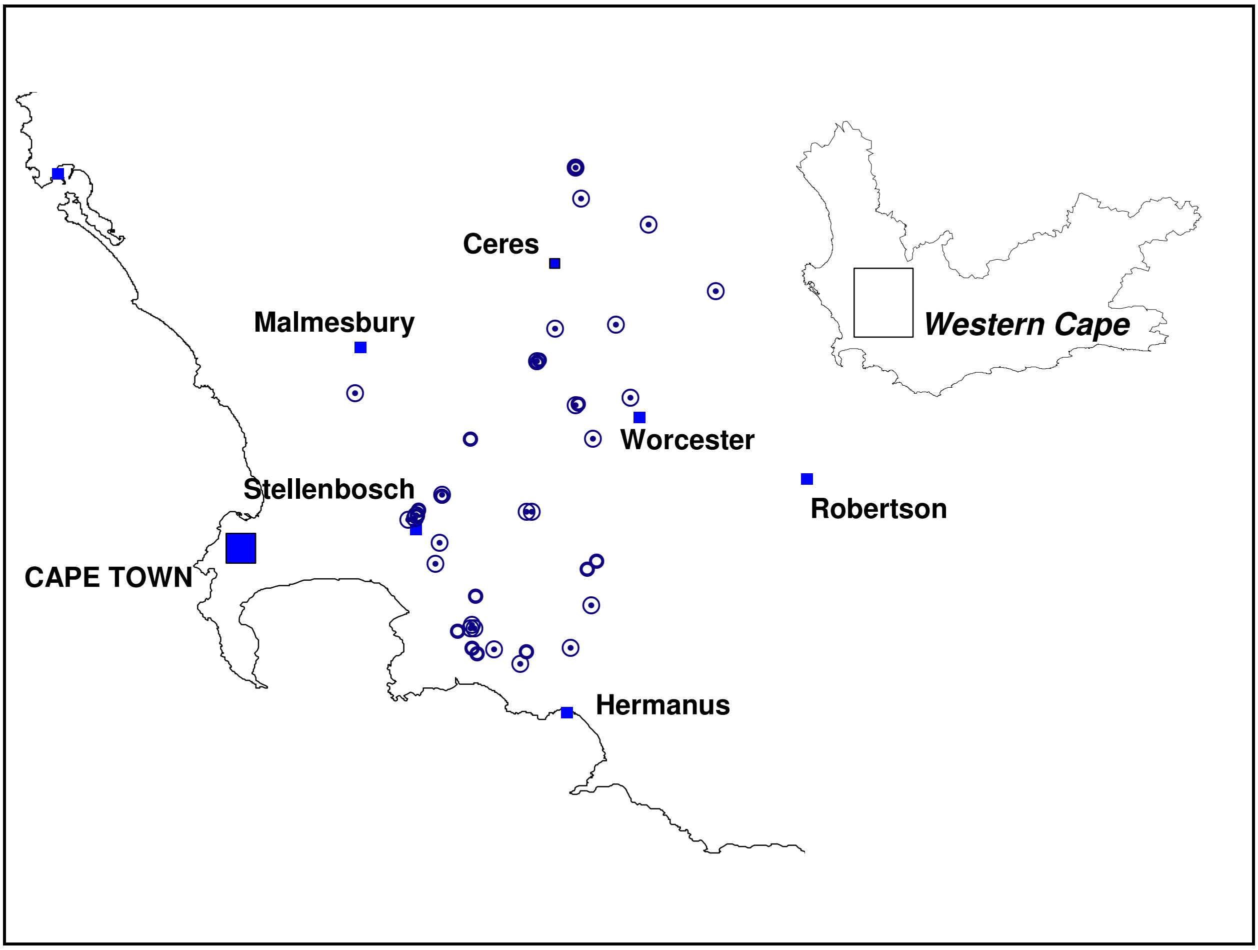

Figure 4.1: Forty-two sampling locations within the Western Cape Province, South Africa. Main cities and towns are marked with squares. The production sites are represented by circles with central dots (26 circles), the reference sites by empty circles (16 circles). Some circles overlap.

Twenty-six production sites (Reservoirs 21 to 46) with aquaculture production of 5 to 15 t per winter season and 16 reference sites (Reservoirs 1 to 16) with no aquaculture production were investigated in this study. All 42 sites (one reservoir served as a reference site in the first year and a production site in the second year) lie within the south-western area of the Western Cape Province , South Africa. The sampling area extends from 32°59'59''S to 34°17'35''S and from 18°43'22''E to 19°39'46''E and covers 150 by 100 km (Figure 4.1).

Cool, rainy winters and hot, dry summers (Mediterranean climate) characterise the region, with the conditions and changes between seasons becoming more extreme further inland, where total annual rainfall also declines. As a consequence, Ceres has hotter summers and cooler, drier winters than Hermanus and Elgin (maximum of 400 mm annual rainfall in Ceres, 1300 mm and more in Hermanus and Elgin).

4.2.1 Production reservoirs

Twelve reservoirs with trout production were monitored monthly, while another 14 sites were visited three times per year (autumn, winter and spring) (Figure 1.2). These three seasons were chosen for the following reasons: autumn - to measure the parameters after stratification was fully established and continued for some months (March); end of winter (August) to measure the oxygen and temperature distributions and nutrient concentrations during turnover and to describe phytoplankton composition at a time of nutrient supply and inset of warmer temperatures; spring (November) to estimate the water quality situation at an early stage of stratification (e.g. hypolimnetic deoxygenation).

The surface areas of the reservoirs ranged from 0.9 to 16.8 ha, with an average of 5.6 ha and three reservoir surface areas were >10 ha (Table 4.1). Four sites were situated >500 m a.m.s.l. All sites of higher altitude were dominated by shale or sandstone rocks. The 22 lowland sites ranged from 42 to 390 m a.m.s.l. with an average of 217 m. Fourteen of these sites were situated in a shale and sandstone dominated underlying rock type (Breede, Olifants-Doring and Gouritz Water Management Area (WMA)) and seven in a granite dominated watershed (Berg WMA). The underlying rock type of one site near the coast was uniquely dominated by ferricrete, silcrete and sand. The surrounding landscape within the watershed area above the site was characterized by natural landscape (primarily fynbos) at 30 % of the sites. All other sites were influenced by vineyards, orchards, olive plantations, pastures, forests, crop production and poultry farming. Except for all locations at higher elevation, another five reservoirs were not influenced by agricultural activities, four of which were based on shale dominated ground.

Nine reservoirs were directly supplied by a river, all of which were lowland sites with equal distribution between granite and shale dominated basins of which six re-feed their water directly into a river. The other reservoirs relied entirely on runoff water, water from other reservoirs or groundwater. The water exchange rate varied between 0.3 and 2 times per year, with two exceptional reservoirs with water exchange rates of 10 and 50 times the full capacity per year.

In the Western Cape, the reservoirs are stocked with 100 g juvenile fish in early winter (April) and harvested in spring (October/November) before summer temperatures increase and negatively effect successful trout production. The net cages inserted into the reservoirs usually hold 5 t of trout (up to 15 t) at harvest with a stocking density of a maximum of 8.1 kg trout/m^3 (Appendix 10.9, Picture 1). Most study sites already had a production history when this study commenced. Only one site could be monitored as a reference reservoir for one year, with the first production season starting the following year. The production history of the sites varied from 1 to 11 years in 2007, with 67 % of reservoirs having produced trout for 1 and 2 years, 19 % having produced for 3 to 5 years and 15 % having produced for more than 7 years (Appendix 10.11). For further details on the production process consult Maleri et al. (2008).

A before-and-after study was undertaken in Reservoir 32, a site influenced by shale as the underlying rock type, with a surface area of 6.9 ha and a water exchange rate of 2.1 times the full supply volume per year. Production started in April 2006 (sampling began in May 2005 until May 2007).

4.2.2 Non-production reservoirs

The 16 reference sites were monitored monthly (seven sites) or three times per year (nine sites). The surface area ranged from 1.4 to 16.5 ha (with one exception of 240 ha), with an average of 6.9 ha (excluding the largest reservoir) and three reservoirs covered an area >10 ha (Table 4.1). Three sites were >500 m a.m.s.l.. These sites were dominated by sandstone. The 13 lowland sites ranged from 75 to 312 m a.m.s.l. with an average of 199 m. Eight of these sites were situated in a shale and sandstone dominated underlying rock type (Breede and Olifants-Doring WMA) and five in a granite dominated watershed (Berg WMA).

The surrounding landscape within the watershed area above the site was characterized by a natural vegetation cover (primarily fynbos) at 13 % of the sites. All other sites were influenced by vineyards, orchards, olive plantations, pastures and forests.

Table 4.1: Comparison of non-production and production site characteristics (n=6 for all reservoirs), considering data of April, July and October samples.

	Non-production reservoirs	**Production reservoirs**
Number of sites	16	26
Surface area	1.4 - 16.5 ha	0.9 - 16.8 ha
Average surface area	6.9 ha	5.6 ha
Reservoirs >10 ha	3	3
Elevation upper sites (a.m.s.l.)	558 – 949 m	530 – 1233 m
Elevation lowland sites (a.m.s.l.)	75 – 312 m	42 – 390 m
Sites >400 m	3 sites	4 sites
Sites <400 m	13 sites	22 sites
granite dominated watershed	31 % of sites	32 % of sites
shale/sandstone dominated watershed	69 % of sites	64 % of sites
Influenced by natural vegetation cover only	13 % of sites	30 % of sites
Water exchange rates (times per year)	0.3 to 2.1; 5.5	0.3 to 2; 10 and 50

Five reservoirs were supplied by river water via channels or sluices, with controllable quantities entering the reservoirs. All others were supplied by groundwater, water pumped from another reservoir and runoff water. Only one reservoir re-fed its water into a river. The water exchange rates varied between 0.3 and 2.0 times per year, with one reservoir achieving a water exchange of 5.5 times the full supply volume per year.

4.3 Methods

Twenty reservoirs were monitored monthly over two years plus an additional twenty-five reservoirs were visited three times each in 2006 and in 2007. Site specific information was recorded and relevant chemical and physical water quality parameters measured. Phytoplankton composition and biomass were intensively studied in 19 reservoirs (12 production and seven reference reservoirs) (Appendix 10.12). Chlorophyll a levels (derived from biomass data) were calculated for all 42 sites. Zooplankton was recorded at 11 sites (six production and five reference reservoirs). Determinants required to calculate trophic status according to Carlson (1977) were recorded during every visit.

4.3.1 Morphometry and hydrology

The length and width of each reservoir at full capacity was measured using Google Earth software (http://earth.google.com) and the surface area estimated via the nearest geometrical shape. The deepest region of seventeen reservoirs was determined through transect measurements of the horizontal and vertical central lines (Wetzel & Likens 2000) and was estimated at all other sites. Sampling locations were fixated by buoys or the cages. The depth at each specific sampling location was recorded.

4.3.2 Water quality

The nineteen (12 production, 7 reference) intensively studied sites were visited every four weeks from August 2005 to October 2007; another nine sites were studied on three occasions per year in 2006 and 2007 (March, August, November). After preliminary depth measurements were determined by means of a weighted line, a buoy was set in the deepest area to ensure a consistent sampling location. The sampling sites were visited in a fixed sequence so that sampling hours at one site stayed within a two hour window throughout the study period.

Water samples were collected at depths of 2 m and 6 m and near the sediment-water interface using a 1.5-L water sampler with a single line trigger mechanism (The Science Source, Waldoboro, ME, USA) and stored and transported according to Wetzel and Likens (2000). At the seasonally visited sites (12 sites), only the 2 m and bottom sample were collected. Turbidity, oxygen content and temperature were measured on site, with all other parameters measured the following day. For a complete list of all physical and chemical water parameters monitored as well as analytical method applied refer to Table 3.2 in Chapter 3.

4.3.3 Phytoplankton

Unfiltered water samples were collected at the surface, a depth of 2 m and 6 m and at the bottom for the monthly sites (intensively studied sites) and at the surface and a depth of 2 m for the seasonal sites. Lugol's iodine solution was added as a preservation method. Species identification and enumeration was possible by means of Utermöhl counting chambers (Utermöhl 1958) and identification volumes of Pascher's "Süßwasserflora von Mitteleuropa" and Thienemann's volumes "Das Phytoplankton des Süßwassers" as well as single identification volumes (for detailed reference list refer to Chapter 3.3.3). The biovolume of each specimen was taken from the literature or calculated via the nearest geometrical shape. Biomass was calculated from the volume data using factors of 1.02 to 1.30 kg/m^3 (Sommer 1996).

4.3.4 Zooplankton

At ten sites (six production sites, four reference sites), zooplankton was collected with a 10-L self-constructed Schindler-Patalas plankton trap. The samples were preserved in 5 % formaldehyde and transferred into a phenoxetol medium called Steedman's solution within 24 hours (Steedman 1976). Protozoans and some rotifera were already established with the phytoplankton section in the Utermöhl chambers, whereas the crustaceans, cladocera and copepoda, and the majority of the rotifera were quantified and determined to genus level with support of a modified Bogorov counting tray and a Leica stereomicroscope (6.3x to 50x magnification). Staining of organisms with diluted Lugol's solution supported the counting process. Identification was based on Day et al. (1999, 2001), Thirion (1999) and Yunfang (1995).

The biovolume and biomass of rotifers and protozoans was estimated via the nearest geometrical shape. The biomass of cladocerans and copepods was estimated via dry biomass estimations of other studies (Gonzalez et al. 2008, Sendacz et al. 2006).

4.3.5 Statistical analyses

Statistical analyses were supported by the Statistica 7.0 program (StatSoft, Inc.). The Mann-Whitney U-Test was employed for comparisons of two non-parametric data sets, the Kruskal-Wallis ANOVA was applied for comparisons of more than two groups. Differences were considered statistically significant if $p<0.05$ and strongly significant if $p<0.01$. Correlations were analysed using the Spearman's rank correlation coefficient.

4.4 Results

Physico-chemical water properties, nutrients, Trophic State Indices (Carlson), and phytoplankton and zooplankton structure from 16 non-production (reference) and 26 production reservoirs (floating cage culture of rainbow trout) were statistically compared to each other.

4.4.1 Physico-chemical water properties and nutrients

In Table 4.2 all measured water quality parameters are listed with mean and standard deviation. Five parameters showed significant differences, two of which were highly significant.

When comparing morphometric and geological conditions, the two data sets were fairly similar. The average surface area of the reference reservoirs was slightly larger. The percentages of elevated sites were 15 and 19 % at production and reference sites respectively. With lowland sites, 54 and 50 % were dominated by shale respectively and 31 % of the sites (each with production and reference sites) were dominated by granite.

The total phosphorus concentrations in the reservoir surface water (0 and 2 m) were also significantly increased in production sites ($p<0.05$), from 68 µg/L to 144 µg/L. With the average production duration calculated as 2.96 y, the average total phosphorus increase per year per reservoir amounted to 25.3 µg/L (for single average TP values for the reservoirs consult Appendix 10.9). An increase by a factor 2 in the TP values can be postulated from the overall average as well as the single reservoir averages within a time frame of 3 years on average, one to two years for most sites. Nitrite concentrations were also significantly increased ($p<0.05$) in production reservoirs when compared to reference reservoirs, from 2 to 7 µg/L as N. There was a trend with the percentage of reservoirs developing a pH>9 at some point in summer.

Table 4.2: Nineteen water quality parameters with mean and standard deviation for non-production and production sites are listed. To equalize data of different sampling regimes, only three values per year (total of n=6) were averaged per reservoir (n=96 for the non-production sites, n=156 for the production sites). The highlighted (light grey) rows indicate parameters with significant differences, the degree of significance represented by asterices (* = p<0.05,** = p<0.01; Mann-Whitney-U-Test).

	Non-production sites (n = 16)		**Production sites (n = 26)**	
	Mean	**Stdev**	**Mean**	**Stdev**
Secchi visibility in cm	118.5	49.8	123.7	42.4
Oxygen min (2 m) in mg/L	6.2	1.0	5.9	1.4
Duration of anoxic hypolimnion in months	**2.3***	**2.1**	**4.6***	**2.6**
pH (2 m)	7.3	1.2	7.1	0.6
Reservoirs with pH(max)> 9 in %	**18.8***	**-**	**26.9***	**-**
pH range (mean min to mean max)	**6.17 to 8.87***	**0.41/0.57**	**6.57 to 9.38***	**0.38/0.51**
Delta pH (fluctuation)	**1.8***	**0.42**	**2.57***	**0.59**
Conductivity in µS/cm	136.6	76.5	144.2	123.0
Alkalinity in mg/L	15.1	13.8	15.6	16.9
Hardness in mg/L	12.0	9.8	15.1	15.1
Silicate in mg/L	7.9	4.5	6.7	4.6
Ferrous Iron in mg/L	1.0	1.3	0.9	1.4
Sulfide in mg/L	0.039	0.025	0.074	0.053
TSS in mg/L	14.2	9.3	15.3	7.2
SRP (2 m) as P in µg/L	28	17	44	24
Total P (2 m) as P in µg/L	**68***	**48**	**144***	**88**
Total P (bottom) as P in µg/L	**98****	**65**	**263****	**173**
Total N (2 m) in µg/L	804	528	1005	664
Nitrate (2 m) as N in µg/L	51	137	140	201
Nitrite (2 m) as N in µg/L	**2***	**2**	**7***	**6**
Ammonia (bottom) as N in µg/L	**118****	**90**	**474****	**280**
BOD_5 (2 m) in mg/L	**2.1***	**0.3**	**2.7**	**0.5**

There was no trend with regards to the overall pH means. However, when comparing the mean of the minimum and maximum pH in the reservoirs respectively and when comparing the delta pH values (maximum value in a reservoir minus minimum value in the same reservoir), there was a significant trend ($p<0.05$) towards higher maximum pH values in production reservoirs and a higher fluctuation (delta pH) in production reservoirs.

Oxygen consumption (BOD_5) increased with degree of eutrophication and was significantly higher in surface waters of production reservoirs in contrast to reference reservoirs.

The duration of anoxic hypolimnia was significantly different ($p<0.05$), defined by oxygen concentrations <2 mg/L. The average anoxic period lasted 2.3 months in the non-production reservoirs and 4.6 months in the production reservoirs. Total phosphorus in the bottom samples (30 cm above the sediment-water interface) showed highly significant ($p<0.01$) increments of TP from 98 to 263 µg/L as P. The same trend was apparent with the bottom ammonia concentrations that increased significantly ($p<0.01$) from a mean of 118 to a mean of 474 µg/L as N. Therefore, parameters of the sediment-near hypolimnion were the most significant indicators for aquaculture impact.

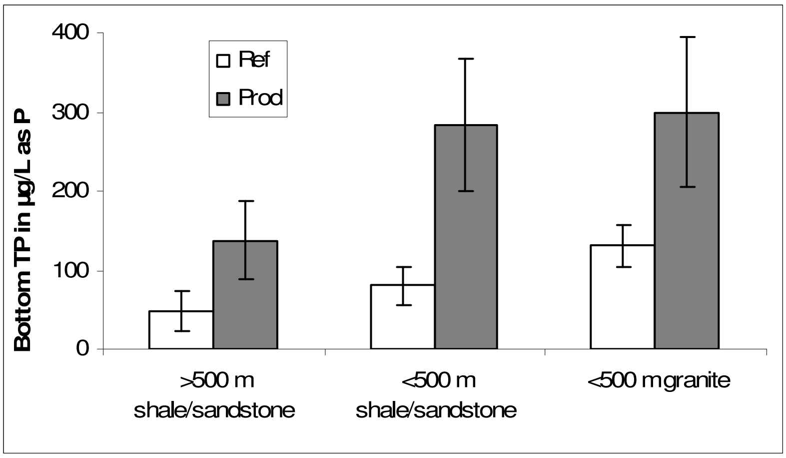

Figure 4.2: Mean TP concentrations (bottom) with standard deviations grouped according to altitude and rock type (Ref: n=3, n=8, n=5; Prod: n=4, n=14, n=8).

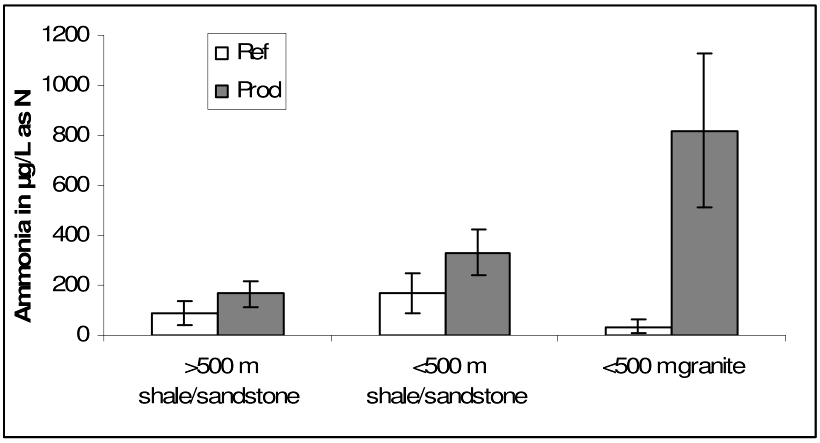

Figure 4.3: Mean ammonia concentrations (bottom) with standard deviations grouped according to altitude and rock type.

Mean phosphorus concentrations could have been dependent upon the years of production. A trend towards elevated phosphorus concentrations in reservoirs with longer production history was, however, not discernable. A similar data classification of hypolimnetic total phosphorus concentrations according to the production history, indicated an almost homogenous phosphorus concentration in all groups (247 to 297 µg/L), however, with high variations (standard deviations) within each group (143 to 271 µg/L).

When comparing sites according to altitude and dominant rock type in the catchment, the increase of certain nutrients in the production sites was again significant, as was the case with the non-production reservoirs (Chapter 3.4.4). This was tested (Mann-Whitney U Test) for total phosphorus at the surface, total phosphorus near the sediment (Figure 4.2) and ammonia near the sediment (Figure 4.3). The concentration of total phosphorus near the sediment was significantly increased in granite rock material in contrast to shale material at the reference sites. However, total phosphorus concentrations in the production sites were very similar between the two rock types at the sites <500 m a.m.s.l.. There was a significant difference ($p<0.05$) between shale/sandstone and granite sites at <500 m a.m.s.l. with regards to surface TP, with higher TP levels at granite sites. In each of the three groups, there was a significant increase of bottom TP in production versus non-production sites ($p<0.01$).

A lower ammonia level was measured at reference sites based on granite, than at reference sites based on shale/sandstone. In production sites, the same pattern as with phosphorus was followed, with significantly increased bottom ammonia levels in production sites based on granite, than in production sites with shale and sandstone as rock type.

When testing the interdependencies and correlations (Pearson correlation) of the various parameters amongst each other to identify indicators, differences between non-production and production sites became more apparent. Average surface and bottom TP correlated among the reference sites (R^2=0.58; p<0.05) as well as average surface TP and surface nitrate (R^2=0.83; p<0.01). Within the production sites, the central parameters with most linkages to other water quality parameters were bottom TP and bottom ammonia. They correlated with each other (R^2=0.684; p<0.01) as well as significantly (p<0.05) with surface TP, bottom soluble reactive phosphorus, maximum surface TP (in contrast to mean surface TP) and surface ammonia. The pH difference between the epilimnion and hypolimnion in summer correlated with TP (p<0.05), but absolute pH values did not.

Additionally, experimentally derived environmental and production factors were compared with water quality parameters. The factors were dimensionless and comprised of the water exchange rate (fraction of full capacity volume that is exchanged every year), the water exchange factor (water exchange rate * the surface area), a rock type factor (differentiating between shale/sandstone and granite rock types with sandstone/shale=1 and granite=0), the years of production and a production factor calculated as production (t) * production years (y) / surface area (ha). The correlations confirmed previously mentioned results, that the catchment environment (rock type) has a greater impact on water quality than production pressure.

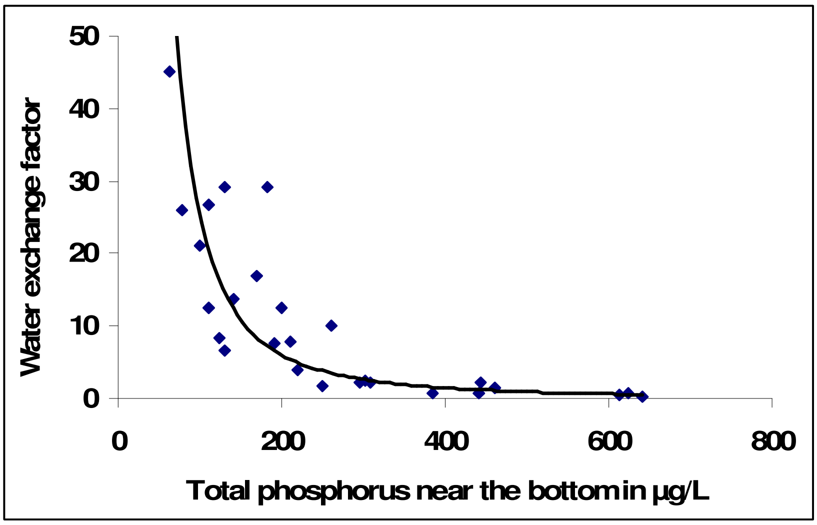

Figure 4.4: Total phosphorus concentrations near the bottom plotted against the water exchange factor (R^2=0.84) of 26 production reservoirs. The water exchange factor was calculated from the water exchange rate (Volume of inflowing water per year per volume of reservoir at full supply) * the surface area.

The water exchange rate, the years of production and the production factor, including years of production and production tonnage, did not correlate to the given water quality conditions. However, the geology factor differentiating between the shale/sandstone and granite based sites showed significant correlations with the bottom ammonia concentrations (R^2=-0.58, p<0.01). The more granite that was proportionally present in the water catchment area, the more ammonia accumulated in the hypolimnion of the production sites. This was, however, not directly dependent, but most likely an effect caused by different processes in the sediment water interface and consequent decomposition of organic material and corresponding deoxygenation. Similarly, the more granite present, the higher the TP concentration in the near-sediment water (R^2=-0.45; p<0.05) and in the surface water (R^2=-0.46, p<0.05).

Water exchange factor values correlated to bottom TP concentrations (R^2=-0.66, p<0.01) and the correlation was best shown using an exponential curve (Figure 4.4, R^2=0.84). The greater the surface area, combined with a greater exchange rate, the less total phosphorus and ammonia (R^2=-0.41, p<0.05) measured near the bottom.

4.4.2 Carlson TSI

When comparing reference site trophic states (Chapter 3 and Table 4.3) to production reservoir trophic states, production sites show the highest levels throughout. Using the TSI (TP) value in particular, the production sites ranged from 60 to 91, that is highly eutrophic to hypertrophic and showed a significant shift from the reference sites (p<0.05). The Secchi depth was not affected by aquaculture activities. The TSI (Chl a) readings showed a trend towards higher phytoplankton biomasses (p<0.01; Mann-Whitney-U-Test). Concurringly, the average TSI values of the production sites were significantly increased (p<0.05) when compared to the reference sites.

Table 4.3: Mean Carlson Trophic State Index (TSI) for production and reference sites, differentiated into the TSI for Secchi depth, total phosphorus concentrations, chlorophyll a concentrations and the average TSI.

		Mean	Stdev	min	max
Non-production sites	**Carlson TSI (SD)**	68	8	53	79
	Carlson TSI (TP)	65	11	45	77
	Carlson TSI (Chl a)	54	19	24	81
	Carlson average	62	11	41	79
Production sites	**Carlson TSI (SD)**	58	5	49	71
	Carlson TSI (TP)	74	8	60	91
	Carlson TSI (Chl a)	75	17	28	114
	Carlson average	68	8	48	89

4.4.3 Phytoplankton

The maximum biomass peaks of the 16 reference sites (single values, not time congruent) averaged at 3.40 ± 4.65 mg/L. At the 26 production sites, the average maximum biomass (single values, not time congruent) was 15.58 ± 14.27. Despite high variances, significantly higher maximum biomasses (referring to the highest biomass observed in each reservoir – one value per reservoir) occurred at the production sites ($p<0.01$; Mann-Whitney U-Test).

When averaging phytoplankton biomasses and classes of the production and reference sites over the study period, significantly higher biomass was recorded in the production reservoirs during each winter and spring (Figure 4.5). With ongoing stagnation, the nutrients within the epilimnion become depleted. Overall biomass at all sites established at concentrations of between 1 and 2 mg/L. There were different seasonal patterns of peaks for reference and production sites, with reference sites peaking in March and April and sometimes additionally in August, while production sites peaked from April to October throughout.

Generally, the biomass peaks of all reservoirs declined when comparing spring 2005 and 2006 results to those results of 2007. This phenomenon emphasizes the overall importance of climatic conditions (patterns of rainfall, sunshine hours, cloud cover and wind) enhancing or alleviating the given nutrient distribution patterns. For example, the sunshine hours in 2007 (September and October) were 2 and 1.5 hours below the average monthly sunshine hours of previous spring seasons (2005 and 2006), respectively. The average wind speed in the study area also decreased by 10 km/h on average in spring 2007 (SAWS 2007). The maximum wind speeds experienced from July to September 2005 and 2006 compared to July and September 2007, differed by 20 km/h in absolute values. Inhibited or delayed mixing limits the accessibility of nutrients and other elements to phytoplankton.

Generally, cryptophytes contributed to 14 % of the overall algal biomass (independent of seasonal distribution) in reference sites, which declined to 2 % in the production sites. The dinophyte presence in biomass increased from 22 to 40 % (reference versus production sites), while the proportion of cyanophyte biomass decreased from 15 to 9 % (reference versus production sites). In absolute terms, the average cyanophyte biomass increased from 5.27 mg/L in the reference sites to 6.15 mg/L in the production sites. The other classes contributed the same proportion in both reservoir types, namely: chlorophytes 40 %, bacillariophytes 5 %, euglenophytes and zygnematophytes each 2 %.

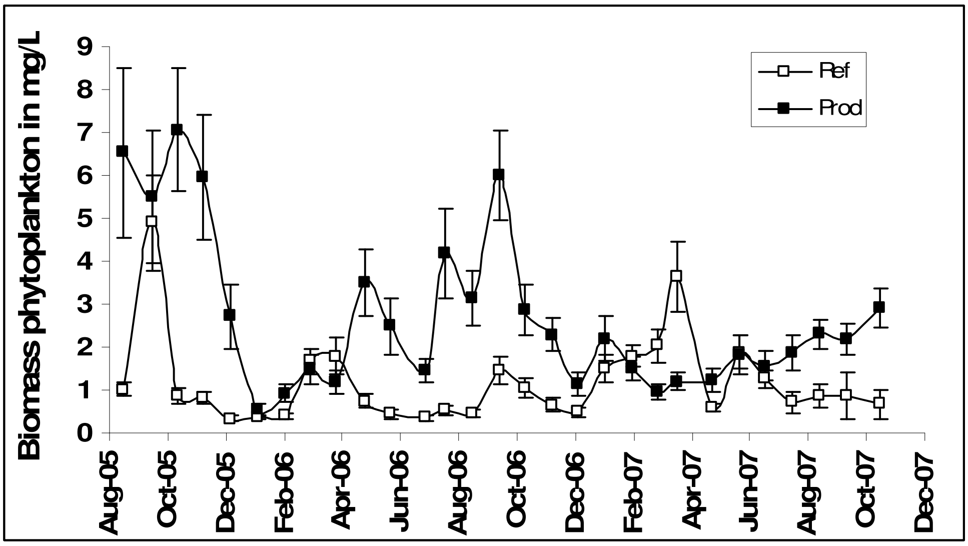

Figure 4.5: Total biomass comparison of reference and production reservoirs from August 2005 to October 2007.

Figures 4.6 and 4.7 illustrate the algal classes in temporal distribution. Dinophytes were more prominent throughout the year and were represented by larger species (*C. hirundinella*, *Peridinium* sp.) in the production reservoirs in contrast to most reference sites (*Glenodinium* sp.). The same shift was noticeable with the chlorophytes where large species such as *Eudorina* sp., Sp*haerocystis* sp. and *Volvox* sp. dominated the chlorophyte structure (only one species in each reservoir), while in reference reservoirs species such as *Ankistrodesmus falcatus*, *Chlamydomonas* sp., *Palmella mucosa* and *Chlorococcum* sp. dominated. Cryptophytes with small species like *Cryptomonas* sp. and *Rhodomonas* sp., were scarce in production sites in contrast to reference sites. With the cyanophytes, *Aphanothece* sp. was only found at one reference site, while *Merismopedia* sp., *Microcystis minutissima* and *M. robusta* were omnipresent. *Gloecapsa punctata*, *Woronichinia* sp., *Oscillatoria* sp., *Anabaena circinalis* and *Microcystis aeruginosa*, *M. flos-aquae* and *M. incerta* only occurred in production sites at biomass levels of 1 mg/L and higher. These four classes together represent 91 % of the biomass in reference and production sites. When bacillariophytes emerged, the primary species were centric diatoms in the mesotrophic sites and *Aulacoseira* sp. in the eutrophicated sites.

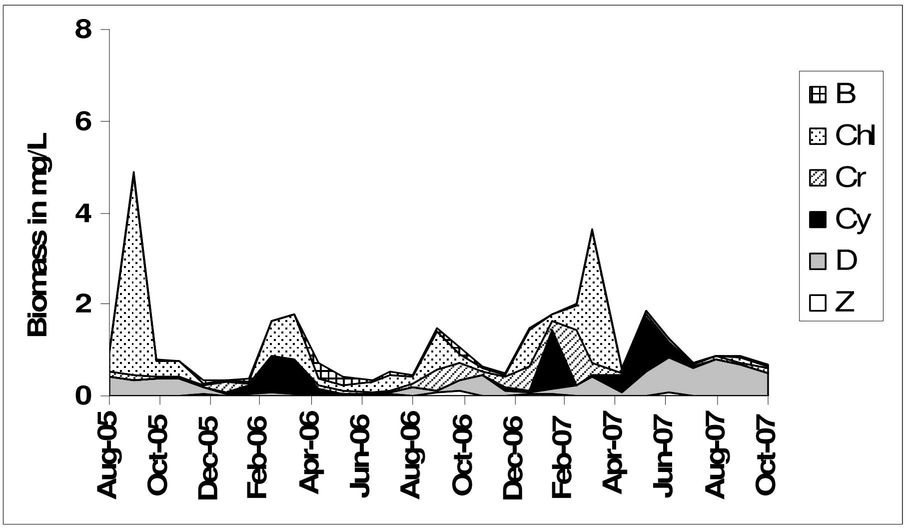

Figure 4.6: Averaged phytoplankton biomass and composition in the phototrophic zone (top 6 m water column) of 7 reference reservoirs. B=Bacillariophyceae, Chl=Chlorophyceae, Cr=Cryptophyceae, Cy=Cyanophyceae, D=Dinophyceae, Z=Zygnematophyceae.

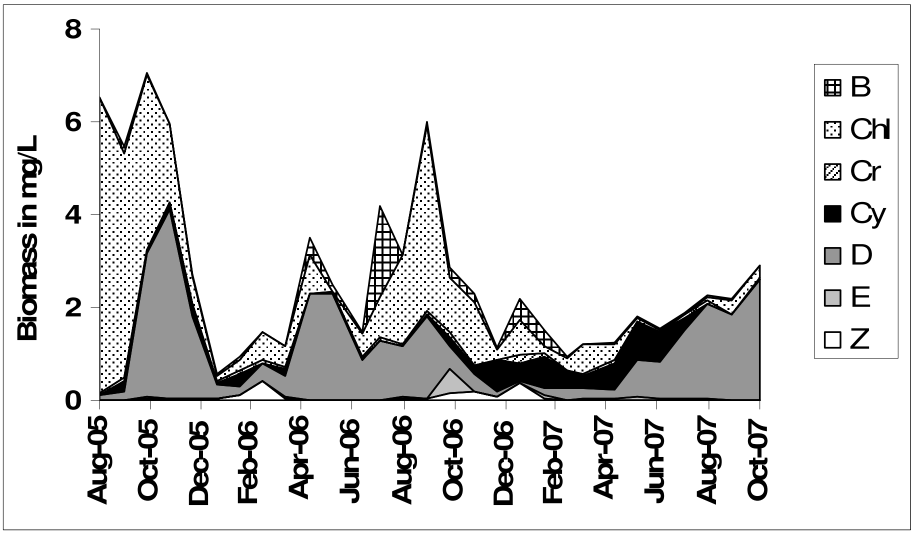

Figure 4.7: Average phytoplankton biomass and composition in the phototrophic zone (top 6m water column) of 12 production reservoirs. B=Bacillariophyceae, Chl=Chlorophyceae, Cr=Cryptophyceae, Cy=Cyanophyceae, D=Dinophyceae, E=Euglenophyceae, Z=Zygnematophyceae.

When comparing algal biomass with single water parameters (Pearson correlation), surface total phosphorus remained the best indicator and the bet-fitted curve to the surface phosphorus versus algal biomass distribution is shown in Figure 4.8 (and Appendix 10.7). In the non-production sites, algal biomass peaks and average algal biomass values (Chapter 2) were best depicted exponentially (R^2=0.78) where average total phosphorus levels fitted into the 15 to 153 µg/L as P window. In the production sites, a linear correlation ($p<0.01$, Pearson correlation) was found with total phosphorus at the surface versus phytoplankton biomass, with total phosphorus varying from 47 to 405 µg/L as P. In Figure 4.8, both fits were combined in one graph, with a freely drawn relationship at highest surface total phoshosphorus values as an approximated fit for highest phosphorus concentrations.

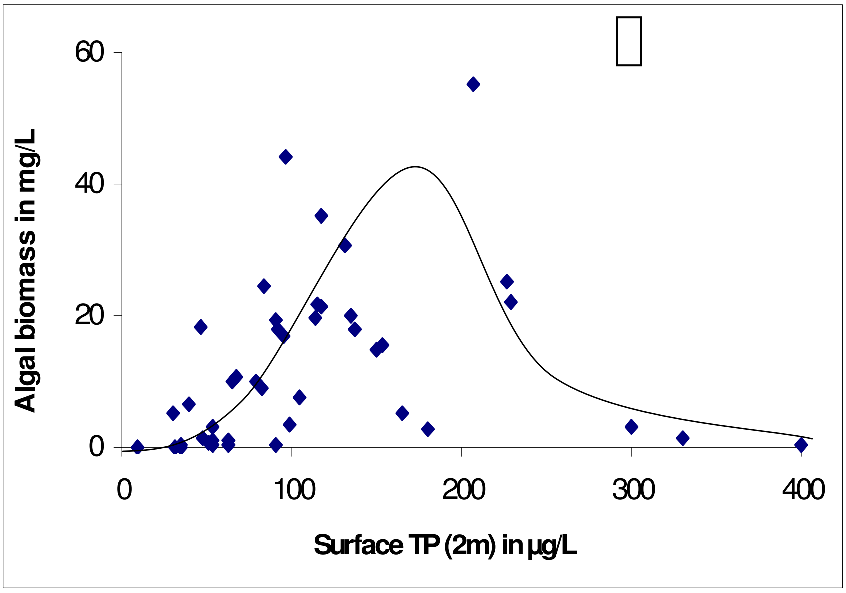

Figure 4.8: Surface TP and respective algal biomass in mg/L with combined exponential (reference sites, range of mean surface TP values from 17 to 153 µg/L) and primarily linear (production sites, range of mean surface TP values from 47 to 405 µg/L) relationship (Data from 41 reservoirs in the Western Cape). The increasing trend was generated with statistical software, the decreasing trend was added as freely drawn line.

4.4.4 Zooplankton

The zooplankton structure showed no significant differences or trends between reference and production sites (Figures 4.9 and 4.10), but a higher fluctuation in zooplankton biomasses did occur during winter 2006. The zooplankton biomasses in winter 2007 remained below the 2006 values, parallel to the findings in the phytoplankton biomass as illustrated in Figure 4.6.

The copepods (primarily calanoids) dominated the zooplankton biomass which indicates low top- down pressure from fish species (natural and introduced). Their main occurrence coincided with periods of high nutrient availability. Calanoids are primarily suspension feeders, feeding on small phytoplankton species, protozoans and bacteria.

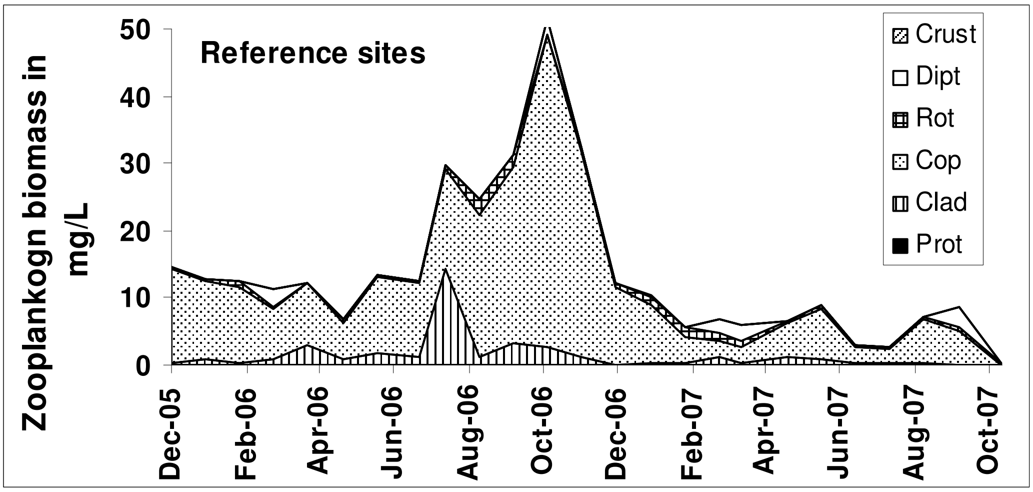

Figure 4.9: Average zooplankton biomass and composition in the phototrophic zone (top 2m water column) of 5 reference reservoirs. Crust=Crustaceans, Dipt=Dipterans, Rot=Rotifers, Cop=Copepods, Clad=Cladocerans, Prot=Protozoans.

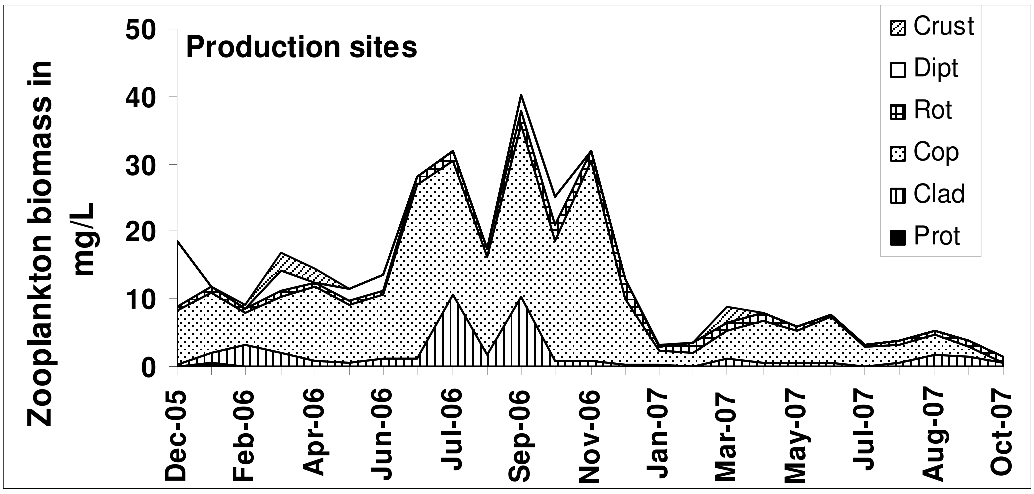

Figure 4.10: Average zooplankton biomass and composition in the phototrophic zone (top 2m water column) of 6 production reservoirs. Crust=Crustaceans, Dipt=Dipterans, Rot=Rotifers, Cop=Copepods, Clad=Cladocerans, Prot=Protozoans.

Rotifera complement the copepod dominance in July, with their nutrient source varying from bacteria to large particles (e.g. *Cryptomonas* sp.). The main rotator species at the reference sites were *Testudinella* sp., *Keratella quadrata* and *K. cochlearis*, while *Polyarthra* sp. and *Pompholyx* sp. were most prominent in production sites.

4.4.5 Before-and-after study

The before-and-after study of Reservoir 32, a 6.9 ha site with an upper third water exchange factor (surface area * water exchange rate), showed effects of aquaculture on the duration of an anoxic hypolimnion, the bottom total phosphorus and ammonia concentrations, and surface pH.

The duration of the hypolimnetic anoxia lasted 3 months in the summer, pre-production and eight months in the summer following the first production season. This indicated greater decomposition processes and an increased anoxic phase impacting sediment-water processes, TP content and ammonia accumulation.

The total phosphorus content in the hypolimnion increased continuously with no discernable difference between the summer and winter period (Figure 4.11). There was a significant ($p<0.05$) increase in total phosphorus from the reference reservoirs winter (79 ± 25 µg/L as P) towards the first production reservoirs winter (172 ± 82 µg/L as P) and a highly significant ($p<0.01$) difference between the reference reservoirs winter and the second production reservoir winter (218 ± 65 µg/L as P)(Mann-Whitney-U-Test). The values of another less intensively studied site increased from 63 µg/L as P before production to 130 µg/L after a 10 t production period commenced.

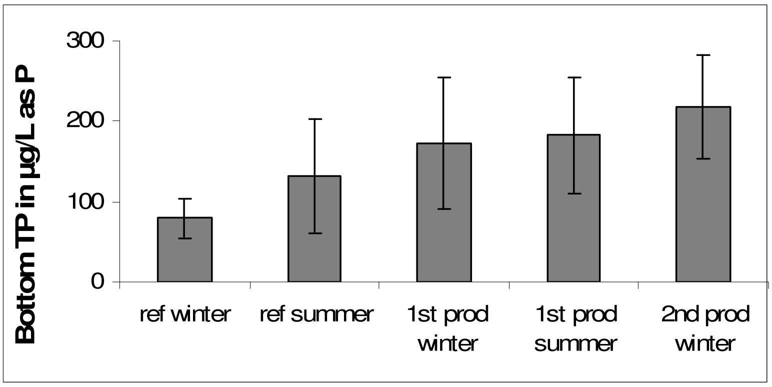

Figure 4.11: Total phosphorus concentrations near the sediment of five consecutive seasons in Reservoir 32 from May 2005 to October 2007 [ref(erence) winter = May to September 2005, ref(erence) summer = October to April 2005/06, 1st prod(uction) winter = May to September 2006, 1st prod(uction) summer = October to April 2006/07, 2nd prod(uction) winter = May to October 2007].

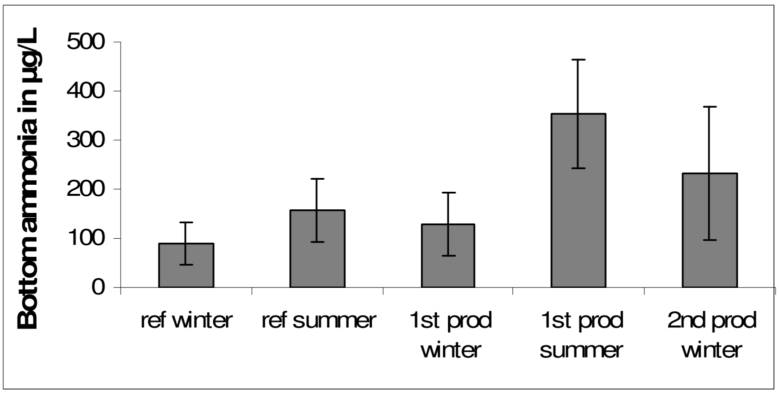

Figure 4.12: Total ammonia concentrations near the sediment of five consecutive seasons in Reservoir 32 from May 2005 to October 2007 [ref(erence) winter = May to September 2005, ref(erence) summer = October to April 2005/06, 1st prod(uction) winter = May to September 2006, 1st prod(uction) summer = October to April 2006/07, 2nd prod(uction) winter = May to October 2007].

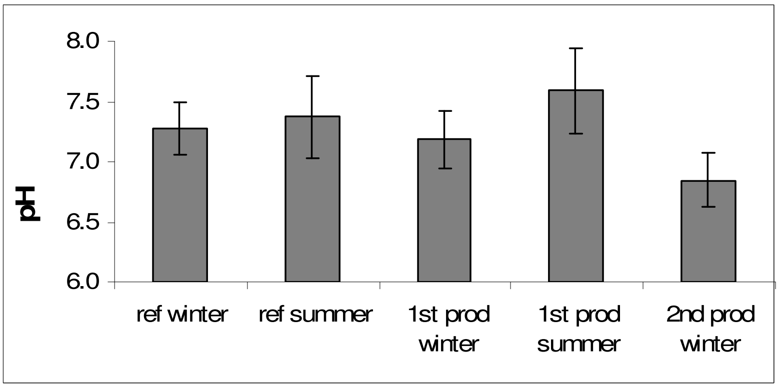

Figure 4.13: The pH value of five consecutive seasons in Reservoir 32 from May 2005 to October 2007 [ref(erence) winter = May to September 2005, ref(erence) summer = October to April 2005/06, 1st prod(uction) winter = May to September 2006, 1st prod(uction) summer = October to April 2006/07, 2nd prod(uction) winter = May to October 2007].

Total ammonia concentrations in the hypolimnion followed a similar trend in the winter period (Figure 4.12), however, only after the second production year did significant differences to the reference winter become discernable (<0.01). The summer ammonia concentrations between the reference summer and the first summer after production were significantly increased ($p<0.01$).

Surface TP increased significantly ($p<0.05$) from 75 ± 37 µg/L as P (winter 2005) to 120 ± 62 µg/L as P (winter 2006), however this then decreased in the second winter season of production to an average of 65 ± 35 µg/L as P (winter 2007), and up to only 20 µg/L can be explained by the elevated algal biomass.

The pH concentrations increased in summer and decreased in winter (Figure 4.13). The summer increase was not statistically significant ($p>0.05$), however, maximum pH was 7.95 in the first summer and 8.43 in the second summer. The winter decrease from the reference winter to the second winter after production began was significant ($p<0.01$).

The phytoplankton findings confirm the water quality trends (Figure 4.14) in that there was no biomass peak higher than 1 mg/L in the reference year (which could have been missed), while in spring following the first and second production winter, the average biomass increased and peaked at approximately 3.5 and 4.5 mg/L respectively (Figure 4.14).

Smaller species were replaced by larger species in the second and third year. In the first winter of production, there was a dominance of bacteriophyceae (Centrales) and chlorophyceae (*Westellopsis linearis*). Dominance of single species prevailed. Biomass may have already been influenced by the more readily abundant nutrients from fish production (September 2006). The first winter following production showed a shift towards cyanophyte and dinophyte dominance which continued throughout winter followed by dinophyte peaks in spring. The dominating species were *Microcystis robusta* (cyanophytes) and *C. hirundinella* (dinophytes). Dinophyte presence was previously common in the reference winter and spring, however, with a different species (*Glenodinium* sp.). *Glenodinium* sp. (20 µm) before production was therefore replaced by a species of greater size and biovolume, namely *C. hirundinella* (150 to 250 µm).

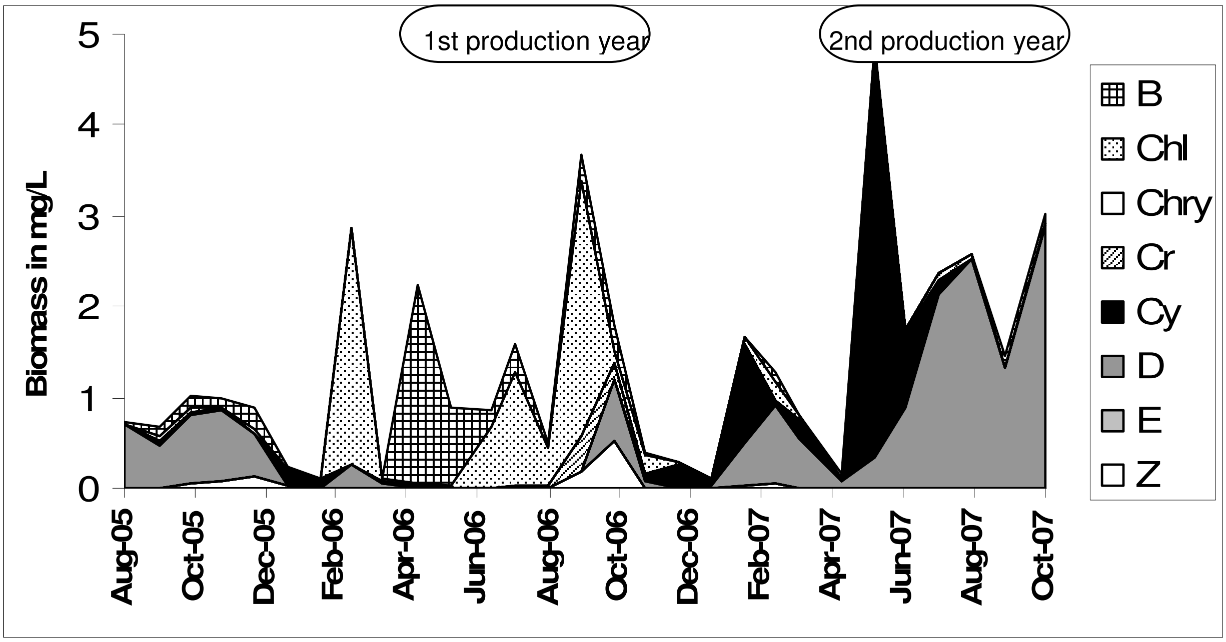

Figure 4.14: The average (surface and 2 m depth) algal class and biomass distribution of phytoplankton in Reservoir 32 from August 2005 to October 2007. B=Bacillariophyceae, Chl=Chlorophyceae, Cr=Cryptophyceae, Cy=Cyanophyceae, D=Dinophyceae, E=Euglenophyceae, Z=Zygnematophyceae.

4.4.6 Low impact sites

According to the above results, some water quality parameters were significantly linked to ecosystem change and could in a broad sense be described as indicators. Good indicators depicting overall ecosystem trophic state and impact of aquaculture, were bottom TP and bottom ammonia concentrations. All sites were subsequently ranked according to an aquaculture impact factor (bottom TP * bottom ammonia concentration). The groups could be divided into two surface area categories. Fifty-eight percent of the sites with the lower ammonia and TP concentrations at the bottom had a surface area of >5 ha (medium impact sites). One exception was a reservoir with a surface area of 0.9 ha, however, the site had a water exchange rate of approximately 50 times per year. The residual sites (42 %) all had surface areas <5 ha with one exception only (high impact sites). The latter reservoir had a surface area of 7.3 ha and a water exchange rate of 4 times per year. Either the water through-flow in the reservoir was suboptimal or there were additional nitrogen sources when compared to other reservoirs (in this case e.g. burning of wood material within the runoff area of the site) since ammonia was proportionally raised and not TP.

All 15 sites with the lower ammonia and TP concentrations were shale/sandstone sites with the exception of two reservoirs of surface areas >10 ha or with water exchange rates of 10 times per year. The residual granite sites were the eight sites with the highest ammonia and TP concentrations, without further exceptions.

The medium impact sites were therefore shale/sandstone based, >5 ha (5.0 to 16.8 ha), with average water exchange rates of 2.3 times per year, while the high impact sites were granite or shale/sandstone based, <5 ha (1.2 to 4.7 ha), with average water exchange rates of 1.1 times per year. The production history indicated that most sites <3 ha had been abandoned after one or two seasons of production due to oxygen problems at the surface. The average bottom TP in the medium impact sites was 156 ± 69 µg/L as P (a 59 % increase from the reference sites), while the high impact sites averaged at 404 ± 163 µg/L as P (312 % increase from the reference sites). Likewise, the mean ammonia concentrations (bottom) of the medium impact group was 254 ± 165 µg/L as N (115 % increase from reference sites), while the high impact group averaged 694 ± 434 µg/L as N (488 % increase from the reference sites).

There were only four sites (16 % of all sites) with a low impact (<15 % increase of bottom TP and ammonia), with the next lowest sites already reaching increases of 42 %. Two sites shared high water exchange rates of 10 and 50 times per year respectively, while the other two had low agricultural pressure and surface areas >10 ha.

4.5 Discussion

The parameters most often applied in studies estimating aquaculture impact are surface total phosphorus concentrations (which enhance algal production), and hypolimnetic hypoxia which induces the conversion of hypolimnetic nitrogen to ammonia and affects the phosphorus binding capacities of sediments (Podemski & Blanchfield 2006, Pillay 2004, Davenport et al. 2003).

The comparison of 16 reservoirs without fish production to 26 reservoirs with fish production showed significant impacts of 5 t production units on an average 6.9 ha reservoir. The differences manifested in increased surface phosphorus levels (68 to 144 µg/L as P; 112 % increase), development of increased pH fluctuations with an alkaline pH in summer and lightly acidic pH in winter (from nearly homogenous pH values throughout the year) and increased maximum pH during the summer period.

In the hypolimnion, changes were most prominent during the summer period of stagnation with increased duration of hypolimnetic hypoxia (2.3 to 4.6 months), increased total phosphorus concentrations (98 to 263 µg/L as P; 168 % increase) and increased ammonia concentrations (118 to 474 µg/L as N; 302 % increase) in near-sediment water below the cages (presumably affecting the majority of the hypolimnion in reservoirs <10 ha).

4.5.1 Effects on physico-chemical parameters

Increases in surface total phosphorus concentrations and total ammonia concentrations in near-sediment water were shown in reservoirs with production (5 t unit of trout) versus reservoirs without production in <10 ha surface area in the Western Cape. Studies by Heath (1990) showed mean surface phosphorus concentrations of 110 to 120 µg/L and maximum concentrations of 420 µg/L at Schoonspruit Dam, with an environmental stocking density of 1600 kg/ha, while De Kroon Dam had average surface TP concentrations of 80 µg/L and maximum TP concentrations of 200 µg/L with an environmental stocking density of 3500 kg/ha. Total ammonia was also measured for both sites and was 4895 µg/L at Schoonspruit and 418 µg/L in the De Kroon reservoir (12 ha). Except for the Schoonspruit ammonia concentrations, these values fall within the range of environmental capacity applied at the Western Cape sites with a similar outcome (especially the De Kroon Dam). In the 2315 ha Lake Wolsey in Canada, the surface phosphorus levels increased by 50 % with an environmental stocking density of 1080 kg/ha (Hamblin & Gale 2002). While a stocking density of 80 kg/ha caused no effects in the 250 ha Lake Menteith (Marsden et al. 1995). Australian research on floating trout cages in small reservoirs concluded that an environmental stocking density of more than 375 kg/ha deteriorated water quality (Gooley & Gavine 2003). Beveridge (1984) states five examples were phosphorus levels were raised after many operational years, with environmental stocking densities between 64 to 380 kg/ha per year (White Oak Lake and Crystal Lake in Arkansas, Lake Glebokie in Poland and Lakes Skarsjon and Byajon in Norway). However, these stocking densities, with effects, have to be seen in the context of the food quality available and food conversion ratios achieved in the respective time periods. Fifteen to 36 kg P loss per ton of produced trout used to be common in the 1980s, 8 to 18 kg P loss in the early 1990s (Gavine et al. 1995), while current operations can lower these values to a 7 to 8 kg P loss (Satoh et al. 2003). There are many studies with regards to the effects of trout production units in the literature, thus supporting the findings of the current study, but additionally in reservoirs of a smaller size (Table 4.4).

Table 4.4: Comparison of available results on effects of trout cage farming in the literature to findings from the current study.

Publication	**Water body**	**Surface area**	**Env. stocking density**	**Effects on surface TP**
own data (Chapter 4)	Western Cape reservoirs	2-10 ha	600-3300 kg/ha	168 % surface TP increase
Hamblin & Gale 2002	Lake Wolsey	2315 ha	1080 kg/ha	50 % surface P increase
Beveridge 1984	Lake Glebokie, Poland White Oak, Arkansas Crystal Lake, Arkansas Byajon, Norway Skarsjon, Norway	47.3 ha 1083 ha 24 ha 140 ha 310 ha	380 kg/ha 138 kg/ha 375 kg/ha 107 kg/ha 43 kg/ha	In all these cases surface P increased after several years of operation
Gooley & Gavine 2003	Australian reservoirs	<10 ha	<375 kg/ha	Presumably no effects
Marsden et al. 1995	Lake Menteith	250 ha	80 kg/ha	no effects

Generally, with increasing eutrophication, the pH fluctuations were enhanced by the naturally low alkalinity and hardness, however seasonal fluctuations increased with a higher eutrophication level (diurnal fluctuation was not measured). Linge and Oldham (2002) found seasonal pH fluctuations of approximately one pH unit in two West Australian urban lakes and they accounted the source of the fluctuation to algal blooms. However, two units difference and more between summer and winter were found in most Western Cape sites, with fluctuations between 6.5 and 8.5. Usually, photosynthetic alkalinisation plays a minor role, however, this process affects water systems in the Western Cape greatly. Algae increase pH by assimilating carbon dioxide which can enhance maximum pH in waters of low alkalinity (Boyd 2000). Aquaculture was shown to enhance eutrophication, and therefore enhances the effects that were described above. When comparing the effects of agriculture (enhancement of surface TP to 1 to 30 %, average of 4 to 8 %) to the effects of aquaculture, aquaculture enhances eutrophication more severly, with effects in the range of >50 % increase per year in 84 % of the reservoirs, which indicates that only some reservoirs have good initial conditions/characteristics for aquaculture production.

Beneath cages, hypolimnetic anoxia is a commonly observed phenomenon and often locally confined (Beveridge 1984). In most studies, the deoxygenating effects of aquaculture waste were minor (Veenstra et al. 2003, Demir et al. 2001, Weston et al. 1996). However, in smaller reservoirs (28 and 56 ha), the whole hypolimnion can be affected or even expanded to the epilimnion (Axler et al. 1996), as was similarly observed in this study. As a consequence, the sediment-water exchange conditions were altered, labile P was released into the water column and nitrate was converted to ammonia. The effect of increased TP and ammonia concentrations was also demonstrated in the current study. Beutel (2001) confirmed a hypolimnetic ammonia increase in relation to hypolimnetic decomposition in Walker Lake in Nevada.

When looking at the hypolimnetic shift in TP concentrations during stagnation (Figure 4.3), there was a clear increase from reference to production sites at granite based sediments and even more distinct increases at shale/sandstone based sediments. However, the absolute bottom TP concentrations in the production sites were the same for both rock types. This might be an indication that an absolute threshold concentration for hypolimnetic TP was reached. In this case, the sediment binding capacity of each site and the absolute sediment TP concentration is essential information to draw conclusions on long-term effects and recovery times from aquaculture pressure. It could also be concluded that once the hypolimnetic water TP threshold limit was exceeded, the impact of future aquaculture will stabilize and the reservoir will stay in the current (however, highly eutrophic) water quality state, until, perhaps, the sediment binding capacities for nutrients reach their upper limit.

4.5.2 Effects on phytoplankton and zooplankton

Elevated surface phosphorus concentrations were found to directly influence algal biomass. Brown et al. (2000) found a sigmoid curve to describe the total phosphorus-algal biomass relationship in temperate and subtropical lakes, with exponential increase with low phosphorus levels (<80 µg/L), a linear mid-section for medium phosphorus concentrations and a top plateau with the highest phosphorus concentrations (>450 µg). The current results confirm the exponential relationship of surface TP concentrations of up to 120 µg/L, as well as a linear relationship with phosphorus levels between 100 and 210 µg/L. Brown et al. (2000) observed a decrease in algal biomass from phosphorus concentrations of approximately 200 to 250 µg/L, which was also found with the four reservoirs exceeding 210 µg/L phosphorus in the current study (230, 300 and 330 µg/L).

Increased algal biomass was accordingly one of the results of additional nutrient input into the reservoirs by aquaculture. Phytoplankton size distribution and community structure were considerably changed with rising eutrophication levels. The number of peaks decreased and the main peak occurred earlier in the season as soon as nutrients became available through lake mixing or low water levels. Reynolds (1984) described the change of phytoplankton structures along a nutrient gradient, where small species were replaced by larger species with increasing eutrophication and certain groups such as chlorophytes, dinophytes and cyanophytes dominated while bacteriophytes and chrysophytes disappeared. According to Reynolds (2006), *C. hirundinella* and *Microcystis* sp. indicated summer epilimnia in eutrophic lakes, usually not at times of mixing. This was not the case in the Hartbeespoort Dam, a hypertrophic reservoir in South Africa, where *C. hirundinella* occurred predominantly at times of mixing, taking over from a *Microcystis* sp. bloom (Van Ginkel et al. 2007, 2001), similar to this study. Pollingher (1986) suggested that subtropical communities are not light and temperature limited, but rather wind (mixing) and rain (runoff and nutrient supply) dependent. In eutrophic Lake Kinneret in Israel, a high standing stock of dinophytes was typical for winter and spring while summer was indicated by low standing stocks of chlorophytes (Pollingher 1986).

The zooplankton structure in the present study showed similarities with findings in subtropical Florida lakes where copepods dominated the biomass (Havens et al. 2007). The overall biomass of the copepods did not change between production and reference sites, however there were more biomass peaks and a shorter presence of zooplankton at each occurrence in the sites with production. The main period of zooplankton presence remained winter and early spring. Havens et al. (1996) postulated that phytoplankton in subtropical lowland sites may be little affected by herbivorous zooplankton species. The current results support these findings in that in reference sites the main phytoplankton growth period was spring and autumn, independent of zooplankton presence and in production sites the main presence of phytoplankton shifted towards the winter period, but did not affect the overall biomass of copepod species or zooplankton in general.

With regards to single species and their indicator value, *Keratella* sp. as one abundant species, are very versatile species that are commonly found at all trophic states. *Testudinella* sp. (4.3) has indeed a lower Trophic Lake Index (TLI) value than *Pompholyx* sp (5.2), an Index used in lake research of New Zealand (Duggan 2007), with *Testudinella sp.* more present in reference sites and *Pompholyx sp.* in production sites.

4.5.3 Indicator parameters (WQ) and influencing factors

Both, agriculture and aquaculture influence the ongoing eutrophication process of the studied Western Cape reservoirs. The extent to which aquaculture contributes to enrichment can be quantified by feasible indicators. The best indicator parameters are obviously the ones undergoing significant changes under aquaculture production and that can be measured over time and cost efficiently (The Commission of the European Communion 2003).

The duration of hypolimnetic oxygen deficiency needs a relatively high frequency of samplings and most reservoirs under production fast consume the available hypolimnetic oxygen reserves. A reliable method for future reference could be the measurement of sediment oxygen demand via respiration chambers (Parsons 2007). An indirect tool for the reducing processes in the sediment, tested in this study, was bottom ammonia.

With pH, the increasing difference between upper and lower boundaries of daily and seasonal fluctuations (ΔpH) could be used as a tool in these reservoirs of low alkalinity, however, there is no generally valid reference. Depending on surrounding land use and geology, the absolute range in which the values fluctuate, differs. Summer maximum pH values >9 are an alarming sign, but maximum pH in general is no absolute measurement of eutrophication level.

The most ready indicator of trophic state in these reservoirs was indeed surface and bottom total phosphorus, in particular when the most influencing factors are considered. Average surface TP values were strongly connected to average algal biomass which can therefore be estimated accordingly (Figure 4.8). There seems to be a tendency towards lower phytoplankton abundance with surface phosphorus concentrations >200 mg/L as P. It would need to be investigated whether this phenomenon is related to greater TSS and lower visibility with higher phosphorus concentrations (insufficient data in the current study). Most reservoirs had a lower algal biomass than indicated by the trend line, primarily due to high TSS loads in August and September, huge abundance of filamentous green algae or other unidentified factors.

The extent to which aquaculture enriches Western Cape reservoirs was detected to be strongly dependent on factors other than nutrient input only. The process of eutrophication depends on the introduced nutrient load per reservoir and the average environmental stocking density, which increased with each reservoir type (Table 4.5), but varied greatly within each group and can not explain the concentration of nutrients available within the reservoir system. Equally, production history (years of production) offered no solution for absolute phosphorus concentrations (within the production sites). Two factors emerged that seem to have a great influence on the nutrients that are available to the lake system, namely: catchment character (dominant rock type) and the water exchange factor (the reservoir surface area * the reservoir water exchange rate). Hayes and Anthony (1958) also found water chemistry to be of better quality in shale/sandstone than granite based sites. Goltermann (2004) described different sediment types and especially rock type (composition of minerals and grain size) to be decisive for sediment phosphorus binding capacity and the consequent phosphorus availability in the water column.

A typology of the reservoirs in this study was attempted and is presented in Table 4.5. It is presented as part of this discussion section as the table is regarded by the author as a basis for the beginning of a discussion.

Table 4.5: Reservoir typology with an arbitrary classification according to bottom TP concentrations (42 stratified reservoirs, most <20 ha, one site 240 ha; n = 6, 20, 10, 6). Relevant parameters and reservoir descriptors were categorized consequently (Bt NH_3 = bottom ammonia between December and February 2005/2006 and 2006/2007; hyp. anoxia = hypolimnetic anoxia).

Type	Average surface TP in µg/L	Average bottom TP in µg/L	Description of reservoirs	Description of reservoir conditions
I	<65	<60	Site above 500 m a.m.s.l. or high water exchange rate (>30) or no agricultural impact on runoff or >200 ha; no production sites, all shale dominated rock type	Hyp. anoxia for 1 to 3 months, bt NH_3 0.01 to 0.15 mg/L, surface pH 4.5 to 7.2, algal biomass peaks <1 mg/L (one exception with 4 mg/L), bacillariophytes and small chlorophytes occur.
II	40-180	60-200	Reservoirs >5 ha, <500 m a.m.s.l. (when >500 with production),	Hyp. anoxia for 2-5 months, bt NH_3 0.05 to 0.3 mg/L, pH 6.5 to 8.5, algal biomass peaks 1-25 mg/L
III	90-160	200-400	Reservoirs >5 ha, <500 m a.m.s.l., all with production or additional nutrient impact (cellar cleaning)	Hyp. anoxia for 5-8 months, bt NH_3 0.1 to 1 mg/L, pH >7.5 (two production sites 6.5), algal biomass peaks 4-40 mg/L, chlorophytes, dinophytes and cyanophytes dominant (blooms of all three classes occurred)
IV	>150	>400	Reservoirs <5 ha, all with production, most sites with granite dominated rock type	Hyp. anoxia full stagnation period (7 or 8 months), bt NH_3 >0.5 mg/L (granite 1 mg/L), max. pH in summer >9 at times, algal biomass peaks >25 mg/L, *Ceratium* blooms, dinophytes and chlorophytes dominant

4.6 Conclusions

An impact of trout cage aquaculture from the first year of trout production was clearly established. Therefore, the natural eutrophication process, already notably enhanced by agriculture in lower regions (1 to 30% addition of surface TP per year), is further increased by trout aquaculture (in 84 % of the reservoirs was >50 % addition per year). Since agricultural practices are already established in the area, the initiation of aquacultural production units in the reservoirs needs to take the existing phosphorus load as well as the loading by agriculture into account, making sure that when adding aquaculture to the systems, they are still able to maintain their ecological equilibrium, and eutrophication progresses sufficiently slowly (<15 % per year). Seeing the severe impact of the trout cage units in most reservoirs, the characteristics of the residual reservoirs suggest however, that production can take place under these specific conditions.

The main consequences of ongoing eutrophication in the studied Western Cape reservoirs, independent of the causal factors (natural or anthropological) and of the time frame in which eutrophication will be established were:

- Earlier and longer deoxygenation of the hypolimnion during stagnation with consequent acidification of the hypolimnion and ammonia accumulation.
- Higher accessibility of nutrients in the surface water during turnover enhances algal peak fluctuations with consequent pH and oxygen fluctuations.
- The water systems attain earlier states where the natural flora and fauna is affected (Specialised towards euryecological species, less dominant species, higher biomass of dominant species).

These changes are expected to take place faster in agriculturally influenced areas and were observed to occur faster, when net cage aquaculture was added to unsuitable sites. In extreme cases, the following effects to the reservoirs uses due to water quality changes were observed and are expected at all sites with a large agricultural and that are unsuitable for trout cage farming in the long run:

- Water extraction and filter systems will be negatively affected (reduced functioning) by higher phytoplankton biomass and larger species.
- Drinking water quality is negatively affected with the emerging presence of *C. hirundinella* or certain cyanophytes.
- Aquaculture itself is affected due to the high oxygen demand of trout and sensitivity to algal taint.

Migrating bird species, however, might profit from the additional structures in the reservoirs and the additional feed input supplying the natural fish stock. The occurrence of any harmful algal bloom development was neither reported nor observed.

In 16% of the reservoirs, favourable conditions that allowr aquaculture production were found. Four out of 26 (16 %) production reservoirs did not show a significant increase in available nutrients (when compared to reference sites) and they were characterised by high water exchange rates (>10 times per year), low agricultural pressure (runoff of nutrients from agricultural land use) and surface areas >10 ha.

Threshold values of suitable parameters (e.g. surface area, maximum reservoir depth, surface phosphorus concentrations, dominant rock type of catchment area and reservoir basin) need to be defined to understand the conditions that provide a minimal impact of net-cage aquaculture. With knowledge of these parameters and an extrapolation of long-term effects, environmentally sound net-cage aquaculture will be possible in small farm reservoirs.

4.7 References

Axler, R. P., Larsen, C., Tikkanen, C., McDonald, M., Yokom, S., and Aas, P. (1996). Water quality issues associated with aquaculture : A case study in mine pit lakes. Water environment research 68(6): 995-1011.

Beutel, M. W. (2001). Oxygen consumption and ammonia accumulation in the hypolimnion of Walker Lake, Nevada. Hydrobiologia 466(1-3): 107-117.

Beveridge, M. C. M. (1984). Cage and pen fish farming. Carrying capacity models and environmental impact. FAO Fisheries Technical Paper (255). Rome, FAO.

Beveridge, M. C. M. (2004). Cage aquaculture. Oxford: Fishing News Books.

Boyd, C. E. (2000). Water Quality: an introduction. Kluwer Academics Publisher.

Boyd, D., Wilson, M. and Howell, T. (2001). Recommendations for Operational Water Quality Monitoring at Cage Culture Aquaculture Operations Environmental Monitoring and Reporting Branch, Ontario Ministry of Environment. Toronto, Canada.

Brown, C. D., Hoyer, M. V., Bachmann, R. W., and Canfield, D. E. (2000). Nutrient-chlorophyll relationships: an evaluation of empirical nutrient-chlorophyll models using Florida and northtemperate lake data. Canadian Journal of Fisheries and Aquatic Sciences 57(8): 1574-1583.

Carlson, R. E. (1977). A trophic state index for lakes. Limnology and Oceanography 22: 361-369.

Davenport, J., Black, K., Burnell, G., Cross, T., Cullory, S., Ekaratne, S., Furness, B., Mulcahy, M., and Thetmeyer, H. (2003). Aquaculture - the ecological issues. Oxford, UK: Blackwell Science Ltd.

Day, J. A., Stewart, B. A., de Moor, I. J., and Louw, A. E. (1999). Crustacea I - Notostraca, Anostraca, Conchostraca and Cladocera. WRC Report No. TT 121/00. Guides to the Freshwater Invertebrates of Southern Africa - Volume 2. Pretoria, Department of Water Affairs and Forestry, South Africa.

Day, J. A., de Moor, I. J., Stewart, B. A., and Louw, A. E. (2001). Crustacea II - Notostraca, Anostraca, Conchostraca and Cladocera. WRC Report No. TT 148/01. Guides to the Freshwater Invertebrates of Southern Africa - Volume 3. Pretoria, Department of Water Affairs and Forestry, South Africa.

Demir, N., Kirkagac, M. U., Pulatsü, S., and Bekcan, S. (2001). Influence of trout cage aqua culture on water quality, plankton and benthos in an Anatolian dam lake. The Israeli Journal of Aquaculture - Bamidgeh 53(3-4): 115-127.

De Villiers, S. (2007). The deteriorating nutrient status of the Berg River, South Africa. Water SA 33(5): 659-664.

Duggan, I. (2007). An assessment of the water quality of ten Waikato lakes based on zooplankton community composition. CBER Contract Report 60. Hamilton, New Zealand, University of Waikato.

FAO (1995). Code of Conduct for Responsible Fisheries. Rome, FAO.

Gale, P. (1999). Addressing Concerns for Water Quality Impacts from Large-Scale Great Lakes Aquaculture. Roundtable Discussion Habitat Advisory Board of the Great Lakes Fishery Commission and Great Lakes Water Quality Board of the International Joint Commission. Toronto, Ontario Ministry of the Environment.

Gavine, F. M., Phillips, M. J., and Murray, A. (1995). Influence of improved feed quality and food conversion ratios on phosphorus loadings from cage culture of rainbow trout, *Oncorhynchus rnykiss* (Walbaum), in freshwater lakes. Aquaculture Research 26: 483-495.

Goltermann, H. L. (2004). The Chemistry of Phosphate and Nitrogen Compounds in Sediments. Dordrecht, Netherlands: Kluwer Academic Publishers (Springer).

Gonzalez, E. J., Matsumura-Tundisi, T., and Tundisi, J. G. (2008). Size and dry weight of main zooplankton species in Bariri reservoir (SP, Brazil). Brazilian Journal of Biology 68(1): 69-75.

Gooley, G. J. and Gavine, F. M. (2003). Integrated Agri-Aquaculture Systems: A Resource Handbook for Australian Industry Development. RIRDC Publication No *03/012*; RIRDC Project No. MFR-2A, 1-189. Rural industries research and development corporation. Kingston, Australia.

Hamblin, P. F. and Gale, P. (2002). Water Quality Modeling of Caged Aquaculture Impacts in Lake Wolsey, North Channel of Lake Huron. Journal of Great Lakes Research 28(1): 32-43.

Havens, K. E., East, T. L., and Beaver, J. R. (1996). Experimental studies of zooplankton-phytoplankton-nutrient interactions in a large subtropical lake (Lake Okeechobee, Florida, U.S.A.) . Freshwater Biology 36(3): 579-597.

Havens, K. E., Beaver, J. R., and East, T. L. (2007). Plankton biomass partitioning in a eutrophic subtropical lake: comparison with results from temperate lake ecosystems. Journal of Plankton Research 29(12): 1087-1097.

Hayes, F. R. and Anthony, E. H. (1958). Lake Water and Sediment I. Characteristics and Water Chemistry of Some Canadian East Coast Lakes. Limnology and Oceanography 3(3): 299-307.
Häusler, J. (1982). Schizomycetes. Stuttgart, New York: Gustav Fischer Verlag.

Heath, R. G. M. (1990). The effects of the cage culture of rainbow trout Oncorhynchus mykiss on dam water quality. In: Aquaculture '90: Proceedings of a joint symposium convened by AASA and the University of Stellenbosch, 11th to 13th July 1990. Pretoria, The Aquaculture Association of South Africa.

Linge, K. L. and Oldham, C. E. (2002). Heavy Metals in the Environment Arsenic Remobilization in a Shallow Lake - The Role of Sediment Resuspension . Journal of environmental quality 31: 822-828.

Marsden, M. W., Fozzard, I. R., Clark, D., McLean, N., and Smith, M. R. (1995). Control of phosphorus inputs to a freshwater lake: a case study. Aquaculture Research 26: 527-538.

Midlen, A. and Redding, T. A. (1998). Environmental Management for Aquaculture. London: Chapman and Hall.

Parsons, M. (2007). Operating Procedure - Sediment oxygen demand. SESDPROC - 507 - R1. Washington D.C., U.S. Environmental Protection Agency, Science and Ecosystem Support Division.

Pillay, T. V. R. (2004). Aquaculture and the environment. Oxford, UK: Blackwell Publishing Ltd.

Podemski, C. L. and Blanchfield, P. J. (2006). Overview of the environmental impacts of Canadian freshwater aquaculture. Canadian Technical Report for Aquatic Sciences. Toronto, Ontario. Fisheries and Oceans Canada (DFO).

Pollingher, U. (1986). Phytoplankton periodicity in an subtropical lake (Lake Kinneret, Israel). Hydrobiologia 138(1): 127-138.

Reynolds, C. S. (1984). Phytoplankton periodicity: the interactions of form, function and environmental variability. Freshwater Biology 14: 111-142.

Reynolds, C. S. (2006). The ecology of phytoplankton. Cambridge: Cambridge University Press.

Rouhani, Q. A. and Britz, P. J. (2004). Contribution of aquaculture to rural livelihoods in South Africa: A baseline study. WRC Report TT 235/04. Department of Water Affairs and Forestry. Pretoria, South Africa.

Satoh, S., Hernandez, A., Tokoro, T., Morishita, Y., Kiron, V., and Watanabe, T. (2003). Comparison of phosphorus retention efficiency between rainbow trout (Oncorhynchus mykiss) fed a commercial diet and a low fish meal based diet. Aquaculture 224: 271-282.

SAWS (2007). Weather data. Cape Town: South African Weather Service.

Sendacz, S., Caleffi, S., and Santos-Soares, J. (2006). Zooplankton biomass of reservoirs in different trophic conditions in the state of Sao Paulo, Brazil. Brazilian Journal of Biology 66(1): 337-350.

Sommer, U. (1996). Plankton ecology: The past two decades of progress . Naturwissenschaften 83(7): 293-301.

Steedman, H. F. (1976). Zooplankton Fixation and Preservation. Paris: UNESCO Press.

Stirling, H. P. and Dey, T. (1990). Impact of intensive cage fish farming on phytoplankton and periphyton of a Scottish Freshwater loch. Hydrobiologia 190: 193-214.

The Commission of the European communities (2003). Commission Recommendation of 10 July 2003 on guidance for the implementation of Regulation (EC) No 761/2001 of the European Parliament and of the Council allowing voluntary participation by organisations in a Community eco-management and audit scheme (EMAS) concerning the selection and use of environmental performance indicators. Document number C2253. Brussels, Commission of the European communities.

Thirion, C. (1999). Introduction to the practice of identifying Zooplankton. Volume 1: Rotifera. IWQS Report number N/0000/00/DEQ0799. Institute for Water Quality Studies, Department of Water Affairs and Forestry. Pretoria, South Africa.

Utermöhl, H. (1958). Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. Mitteilungen der internationalen Vereinigung der theoretischen und angewandten Limnologie 5: 567-596.

van Ginkel, C. E., Hohls, B. C., and Vermaak, E. (2001). A Ceratium hirundinella (O.F. Müller) bloom in Hartbeespoort Dam, South Africa. Water SA 27(2): 269-276.

van Ginkel, C. E., Cao, H., Recknagel, F., and du Plessis, S. (2007). Forecasting of dinoflagellate blooms in warm-monomictic hypertrophic reservoirs in South Africa by means of rule-based agents. Water SA 33(4): 531-538.

Veenstra, J., Nolen, S., Carroll, J., and Ruiz, C. (2003). Impact of net pen aquaculture on lake water quality. Water Science and Technology 47(12): 293-300.

Weston, D. P., Phillips, M. J., and Kelly, L. A. (1996). Environmental impacts of salmonid culture. In Pennell, W. and Barton, B.A.: Principles of Salmonid Culture - Developments in Aquaculture and Fisheries Science. Volume 29. Amsterdam: Elsevier.

Wetzel, R. G. and Likens, G. E. (2000). Limnological Analyses. New York: Springer.

Yunfang, H. M. S. (1995). Atlas of Freshwater biota in China. Beijing: China Ocean Press.

CHAPTER 5 PHOSPHORUS FRACTIONS OF SEDIMENTS IN WESTERN CAPE RESERVOIRS WITH AND WITHOUT RAINBOW TROUT (*ONCORHYNCHUS MYKISS*) AQUACULTURE

Abstract

Regarding phosphorus concentrations in surface waters, internal loading via sediment resources can become the driving component of nutrient excess problems. Aquaculture waste will likely contribute to the total phosphorus present, but could also change processes in the sediment-water interface.

Binding capacities of sediments and possible release under different hypolimnetic conditions was investigated in 10 Western Cape reservoirs and related to phosphorus introduction with net cage aquaculture of rainbow trout (*Oncorhynchus mykiss*). Due to the young geology of the area and also dependent on reservoir construction, the sediment of the reservoirs differed in texture, colour and mineral content, primarily sandstone/shale or granite influenced.

Total phosphorus (TP) was not generally higher in the sediments of production reservoirs (overall range of 59 to 769 mg/kg) or influenced by land use in general. The origin of the reservoirs' geology and the input of Total suspended solids (TSS) has the greatest influence on TP, as well as organic material input (e.g. trees and birds). A comparative study within a site with 11 years of production history showed however, that sediment phosphorus increased by 72 % beneath the cages. The algal available P fraction (NaOH extraction) comprised of 37.9 % (average) of sediment TP, while water soluble P was 2.7 % and readily desorbable P was 0.5 %.

With longer anoxic periods beneath the cages, which was characteristic for eutrophied reservoirs but is further enhanced by aquaculture production, phosphorus will be re-released into the lake water increasing overall phosphorus concentrations. The trend of increasing hypolimnetic phosphorus concentrations during stagnation was very discernible in the production sites. This effect was independent of production history. There was also a higher proportion of organically bound phosphorus found in production site sediments which have different release mechanisms than inorganically bound phosphorus.

A shift in the phosphorus binding and re-releasing processes in sediment areas affected by aquaculture waste is assumed from the data of hypolimnetic water conditions such as extended anoxia and change in organic/inorganic components and therefore processes. However, With a

study of total phosphorus and its fractions and a sampling regime of twice per year, no light was shed on the exact mechanisms in the sediments.

5.1 Introduction

The majority of aquaculture waste products from cage production will be introduced as solids (Phillips et al. 1993) and settle to the lake sediment. Generally, the top layer of sediments are composed of all incoming matter settling to the lake bottom (organic and inorganic) and is therefore affected by surrounding land use, geological background (Olsson et al. 2004) and in-lake activities. Within the sediment structure, the inorganic mineral contents regulate binding capacities, while grain size and shape control adsorption processes (Rabitti et al. 1983). Organic matter is primarily decomposed by microbial activity. All these processes are interdependent and affect each other. Hypolimnetic water conditions will further influence how specifically phosphorus can be immobilised or released from chemical compounds or by bacteria. Depending on the former organic fraction in the sediments, organic waste from aquaculture may change the processes in the sediments (Kelly 1993).

Sediments are composed of minerals, organic and inorganic species and water, and they act as a sink or source for phosphorus (Christophoridis & Fytianos 2006). The main external sources of phosphorus loading are runoff and drainage from intensively cultivated areas, sewage and runoff from livestock farming (Reynolds & Davies 2001) in general, presumably runoff and drainage from cultivated areas and aquaculture for most studied reservoirs. Phosphorus fractions of sediments can be separated by ecological importance and bioavailability (Zhou et al. 2001). To estimate possible environmental impact, several authors use the categories of water soluble, readily desorbable and algal available phosphorus (Branom & Sarkar 2004, Zhou et al. 2001). The water soluble fraction can immediately become available (pore water, diffusion). Readily desorbable and algal available phosphorus, however, are only potentially released fractions. Readily desorbable phosphorus is released by mechanical turbulence and resuspension and corresponds to loosely attached phosphorus adsorbed at the sediment particle surface. Algal available phosphorus can be accessed by change of chemical conditions and binding capacities, such as pH or redox potential (Ting & Appan 1996) and coincides with the physico-chemical fractions of metal-bound and calcium associated P (non-apatite inorganic P). Therefore, periods of resuspension as well as iron content, calcium content and other sediment properties will play a deciding role, as well as ecological processes such as hypolimnetic pH and oxygen content.

The diverse geology of the Western Cape Region and additional sedimentary processes permit different conditions within a small geographical area. The composition of the sediment in a reservoir will depend on the geological background of the catchment (minerals and their grain sizes), the construction of the reservoir (whether artificial support and ground improvement took place, groundwater exchange, runoff accessibility), introduced material (soil erosion with runoff in soluble and particulate form) and the settlement of organic material (biota, phytoplankton, macrophytes, leaves, fish feed and faeces). Depending on oxygen availability and pH, calcium, iron (and other metals) and organic material immobilise or release phosphorus.

Within the Western Cape, small irrigation reservoirs (1-17 ha) were used for rainbow trout rearing in cages. Usually five tons of rainbow trout are reared within four meter deep cage nets (up to 8.1 kg trout/m^3), with a mean environmental stocking density of 900 kg trout/ha (maximum 5000 kg trout/ha) and a feeding efficiency expressed as FCR ranging from 1.1 to 2.0 (average 1.5) (Maleri 2009). Water quality and production problems (low surface oxygen content, algal taint) emerged at some sites while others functioned successfully. Water related mechanisms were described in Chapter 4, but another unknown factor is the sediment layer with sediment-water interactions.

Four phosphorus fractions were measured under anoxic and oxic conditions, with and without aquaculture production in different geological background situations, and compared to hypolimnetic water conditions and other sediment properties, to understand the long-term binding capacities and conditions of phosphorus release in Western Cape reservoir sediments.

5.2 Methods

5.2.1 Sites

Ten sites were chosen to investigate sedimentary qualities and phosphorus fractions. They were composed of six production sites with aquaculture histories of one year, three years (three sites), eight years and eleven years and four reservoirs with no fish production (Appendix 10.12). Samples were collected near the cages or a buoy set in the deepest reservoir area. At the site with 11 years of production history (Reservoir 27), an additional sampling location within the reservoir centre was introduced where the sediment was expected to be uninfluenced by direct waste from the cages.

All sites lie within the south-western area of the Western Cape Province, South Africa. They extended in an area between 33°42’22’’ S to 34°15’48’’ S and 18°51’47’’ E to 19°01’08’’ E, shaped like an upright rectangle (longer north-south axis) of 70 by 20 km. Cool, rainy winters and hot, dry summers characterize the region (Figure 4.1 in Chapter 4).

One of the four sites without fish production was situated in a sandstone and shale dominated geology while the other two were characterized by granite and another reservoir was of shale and alluvial character. Three production sites were granite based while the remaining two as well as the before-and-after site were sandstone and shale dominated.

5.2.2 Sample collection

From the sidewalk of the cages or near the buoy marking the deepest area of the reservoir, a Van Veen grab sampler was deployed, hauling material from three positions at approximately 5 m distance from each other, originating from the upper 10 to 15 cm of the sediment layer. The additional sampling site in Reservoir 27 was 100 m away from the cages and then sampled in a similar way, with a 5 m distance between replicates.

Approximately 2 kg of sediment material was retrieved, thoroughly mixed, and 500 g subsamples taken and stored at 4 °C. After sediment samples were set-aside for the immediate tests which were carried out the following day, approximately 100 g of sediment was weighed, air dried for 90 days, reweighed and the moisture content determined by comparison of initial wet weight with the dry weight.

5.2.3 General properties

The analyses of general sediment properties were outsourced to the BemLab laboratories (BemLab^{Edms} Bpk., De Beers Road, Somerset West, South Africa). They determined available iron content, sulphur, chlorine, pH (KCl), total nitrogen, carbonate, total carbon, Bray II phosphorus and exchangeable cations (K, Na, Ca and Mg) of wet sediment samples.

5.2.4 Sediment phosphorus fractions and moisture content

The three samples from each reservoir were processed for total phosphorus as well as the water soluble, readily desorbable and algal available phosphorus fraction.

For total phosphorus determination, 0.5 g wet sediment was dissolved in 25 ml demineralised water, thereafter 3 ml of 30 % sulphuric acid was added as well as potassium peroxodisulfate (point of a spatula). The solution was heated for 2 hours at 100 °C. After cooling, sodium hydroxide was added (30 %) until a pH 5 was reached. The samples were filtered through 8µm filter paper. The phosphorus content was determined with a diluted sample (1:10) according to the molybdo-vanadate method (APHA et al. 2005). Three test replicates were run for each original sediment sample (nine measurements per site).

For each of the three samples per site, three replicates were treated as follows and phosphorus content determined. For water soluble phosphorus, 1 g of wet sediment was dissolved in 100 ml demineralised water and shaken at 200 rpm (New Brunswick Scientific, NJ, USA) for two hours (Zhou et al. 2001). Equivalently, for the samples tested for readily desorbable phosphorus, 2 g sediment sample was mixed with 50 ml 0.01 M calcium chloride and shaken for one hour (Zhou et al. 2001). For algal available phosphorus, 0.8 g sediment sample was mixed with 0.1 M sodium hydroxide and shaken for four hours (Lathrop et al. 2006, Zhou et al. 2001, Manning et al. 1984). All samples were filtered through 8µm and 0.45µm mesh sizes and the phosphorus concentration determined.

5.2.5 Bottom water parameters

Bottom water samples were collected by means of a 1.5-L water sampler (The Science Source, Waldboro, ME, USA) and oxygen concentration (OxyGuard Polaris) was measured on site. In the laboratory, pH was determined (Hanna pH 211 microprocessor). Ferrous iron was measured with the 1,10-Phenanthroline method (APHA et al. 2005). Total phosphorus samples were acid digested and measured with the molybdo-vanadate method (APHA et al. 2005).

5.3. Results

5.3.1 General sediment properties

Descriptive parameters were determined and metal, mineral and nutrient content of four reference and six production reservoirs was determined. They reservoirs differed in production history (2 to 11 years), geological background (shale/sandstone or granite dominated) and colour of sediments (light to brown/black) (Table 5.1). Inorganically influenced sediments tend to be light in colour (yellow to light grey or brown), while organically enriched sediments are darker (Nichols 1999). With Western Cape reservoirs, the youngest reservoir was the lightest in colour (P23, constructed in 2003). However, reservoirs with substantial influence of sediment runoff can appear lighter than their age and trophic status may suggest.

The pH (KCl) refers to the acidity in the soil solution, but in contrast to water measured pH, includes the reserve acidity in the colloids and is therefore more acidic than water pH (they share the neutral point of 7). Most reservoirs' sediment pH (and hypolimnetic pH) was acidic throughout, with the exception of Reservoir 6 (without fish) where the sediment pH(KCl) was alkaline (Table 5.1).

The minerals and metals in the reservoirs differ greatly, with little specific pattern with the rock type. Available iron levels correlated with the rock types ($p<0.01$), with higher concentrations of iron in granite areas (759-1095 mg/kg) than in shale/alluvial areas (72-654 mg/kg). Therefore, more phosphorus can theoretically be iron-bound in granite dominated areas, but also seems to be more readily released (Chapter 3, higher surface water P in granite dominated areas).

Sulphur concentrations were elevated in Reservoir 27 (with fish) (mean of 335 mg/kg) and in particular in Reservoir 6 (818 mg/kg), without fish. At all other sites, the sulphur concentrations remained <100 mg/kg.

Table 5.1: Production history, geology, pH, metal and mineral content of 10 Western Cape reservoirs (Reservoir 27 had two sampling locations). S-S: Shale/sandstone, G=granite, A/G=alluvial/granite**.**

	Prod	**Rock type**	**Year of constr.**	**Colour**	**pH (KCl)**	**Total iron (Fe)**	**Available S**	**CO_3**	**Cl**	**Nitrogen (TN)**	**Carbon (TC)**
	y					**mg/kg**	**mg/kg**	**mg/kg**	**mg/kg**	**g/kg**	**g/kg**
1	0	S-S	1990s	mocha	4.7	653.8	57.2	2.0	40.1	2.2	22.4
3	0	A/G (S-S)	1973	dark brown	4.5	560.8	16.0	3.0	4.1	1.1	9.0
5	0	G	1965	dark brown	4.6	858.7	44.2	4.6	6.9	3.0	23.4
6	0	S-S	1980s	dark brown	7.3	152.6	817.6	4.2	22.1	8.1	16.6
22	8	A/G (S-S)	1972	dark brown	4.5	466.3	11.9	2.4	2.8	1.1	11.9
23	2	S-S	2003	yellow	4.6	71.6	8.0	1.2	5.5	0.4	1.7
24	2	S-S	1970s	brown	6.0	419.5	47.3	2.1	11.1	1.8	11.8
26	2	G	1990s	brown	5.0	759.3	95.7	1.6	6.9	1.5	14.4
27	11	G	1985	greyish black	5.2	1060.2	326.4	3.4	49.7	3.0	22.9
27 cage	11	G	1985	greyish black	5.5	1095.3	344.6	2.5	23.5	4.0	23.9
32	2	S-S	1980s	dark brown	5.1	232.1	32.5	1.6	13.8	1.4	17.4

Reservoirs 27 (with fish) and 6 (without fish) were additionally influenced by bird presence. Egyptian geese (*Alopochen aegyptiacus*), 200 to 400 birds, visited Reservoir 27 from November to April each year. The bird population at Reservoir 6 was more diverse, however, a total of approximately 100 to 200 birds were present from December to May. At Reservoir 6, primarily Egyptian geese, coots (*Fulica* sp.), cormorants (*Phalacrocorax capensis* and *Phalacrocorax carbo lucidus*) and diverse duck species were sighted. Other factors influencing the input into reservoirs were trees. Reservoirs 24 (with fish) and 5 (without fish) were both surrounded by stands of trees (small forests) stretching along half the lake bank. Reservoirs 27 (with fish), 6 and 1 (without fish) were influenced by trees along about one quarter of the shoreline, while the residual reservoirs had single or no trees in their vicinity.

Moisture/pore water content of the sediments ranged from 32 to 52 % for most shale/sandstone and alluvial sites, and from 54 to 65 % for most granite sites (Figure 5.1). Assuming that the compaction of the upper 10 cm sediment layer in sediments >10 years is similar in the studied reservoirs, it can be postulated that finer material (shale/sandstone grains) has less pore water capacity and a reduced moisture content than granite derived material.

Reservoir 6 was an exception with a shale/sandstone character, but a water content of 68 %. The same observation was made in Chapter 3 (Figure 3.6) where the surface phosphorus content of Reservoir 6 was similar to that of the granite group. Reservoirs 22 (with fish) and 3 (without fish) were granite/alluvial in basic character and have a strong influence of alluvial and shale material introduced by a river, judging from the water content of the upper 10 cm sediment layer (last 10 to 15 years). However, in terms of surface phosphorus measured in Chapter 2, Reservoir 3 (without fish) fits into the granite group.

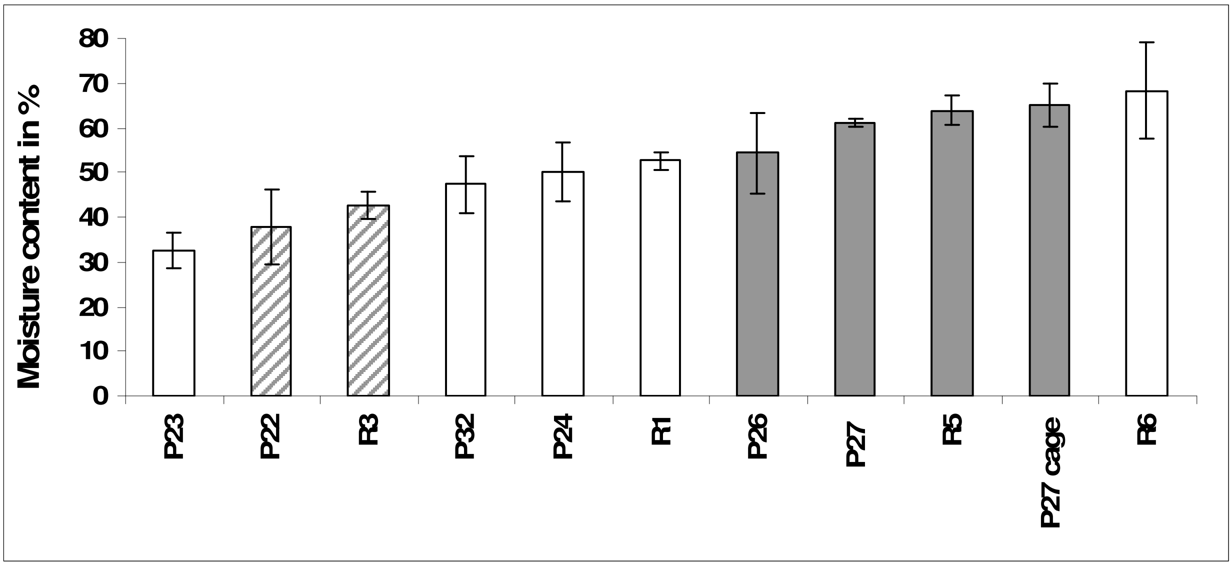

Figure 5.1: Moisture content of the sediment probed reservoirs. In white are the shale/sandstone based reservoirs and in grey the granite based reservoirs. The striped bars represent two granite/alluvial reservoirs.

Table 5.2: Mean exchangeable cations (n=4) in cmol(+)/kg and their base saturation (%) in 10 reservoirs. The T-value represents the cation exchange capacity of the sediments.

	K	K sat	Na	Na sat	Ca	Ca sat	Mg	Mg sat	T-Value
	cmol(+)/kg	%	cmol(+)/kg	%	cmol(+)/kg	%	cmol(+)/kg	%	cmol/kg
1	0.24	4.00	0.44	7.24	2.30	36.56	1.77	29.58	6.07
3	0.44	4.94	0.35	3.94	2.96	33.17	3.85	44.20	8.81
5	0.35	4.83	0.44	6.04	3.49	47.60	2.29	31.15	7.39
6	0.51	1.63	0.54	1.67	28.23	91.38	1.67	5.33	30.94
32	0.32	4.59	0.35	4.91	3.49	49.38	1.88	26.64	7.05
22	0.31	4.42	0.17	2.40	2.26	33.09	2.90	42.62	6.80
23	0.06	2.91	0.04	1.81	1.11	48.38	0.43	18.95	2.25
24	0.34	4.45	0.24	3.27	5.32	68.86	1.35	18.49	7.60
26	0.50	5.98	0.41	4.81	4.21	49.94	2.72	32.00	8.46
27	0.42	4.65	0.49	5.50	4.90	54.78	2.17	24.25	8.95
27 cage	0.38	4.16	0.47	5.10	6.02	64.89	1.68	18.55	9.17

The exchangeable cation contents of the single reservoirs was similar with calcium being the most important cation, followed by magnesium and at much lower concentrations both potassium and sodium (T-values range from 6.1 to 9.2 cmol/kg for most reservoirs) (Table 5.2). Reservoir 23, the youngest site (constructed 3 years prior to sampling), was the exception at the low end, with only approximately one quarter of the cations present in other reservoirs, whereas Reservoir 6 was the exception at the high end with a T-value of 30.9 cmol/kg (four times the concentrations in other reservoirs). Calcium plays a role in phosphorus binding (calcium phosphate salts) and can be replaced by other chelating agents e.g. produced by algae and bacteria.

5.3.2 Total phosphorus and phosphorus fractions

There was no discernible pattern between total phosphorus content and trophic status of the reservoirs, their production history (years that trout were reared within the reservoir), catchment geology, geographic location or surrounding land use. However, surrounding land use combined with the proportion of runoff contributing to the total inflowing water volume correlated with total phosphorus in the sediments (r^2=0.80, $p<0.01$).

The mean concentrations of total phosphorus are illustrated in Figure 5.2 in increasing order. Reservoirs 32 and 1 were both located near Elgin and showed the lowest sediment phosphorus, however, Reservoir 24 also situated near Elgin was one of the high phosphorus sediment sites (very steep shore). The two other high-phosphorus sites were Reservoirs 6 and 27. Beneath the cages in Reservoir 27 the phosphorus content had increased by approximately 60 % ($p<0.01$) in contrast to the other sampling locality (reference) in the reservoir.

The bioavailable portion (non-apatite inorganic P) comprised of 32.7 to 225 mg/kg of sediment and accounted for between 33 and 55 % of total phosphorus for most samples (Table 5.3). Two reservoir sediments had higher and lower bioavailable P fractions. Reservoir 23 (with fish), the youngest reservoir, contained less bioavailable phosphorus (20 %) and Reservoir 26 contained more bioavailable phosphorus (75 %). In 2005, Reservoir 26 (with fish) was impacted by a landslide of agricultural topsoil from the adjacent hill which might have caused the higher inorganic phosphorus proportion.

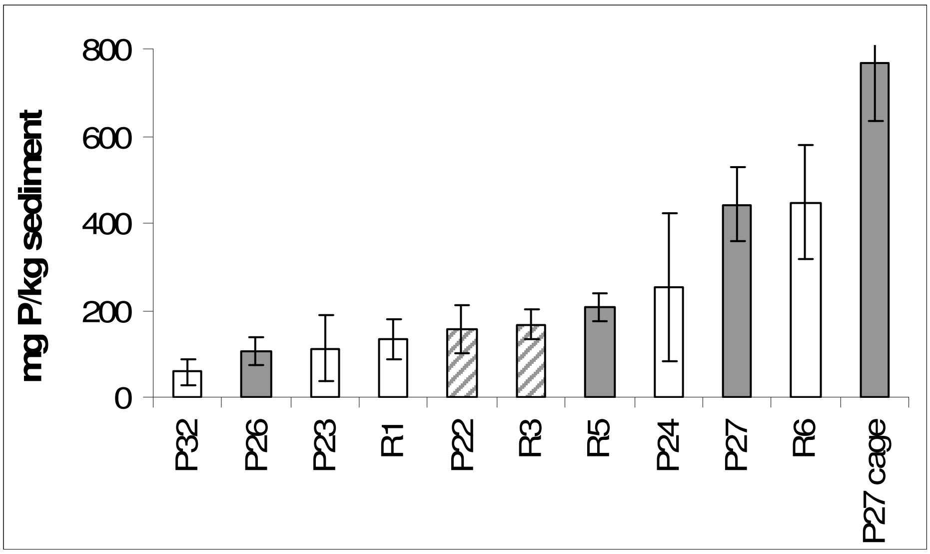

Figure 5.2: Mean total phosphorus (n=4) in the sediments of ten farm reservoirs in the Western Cape in increasing order, with two sampling locations in Reservoir 27. Grey bars = granite, white bars = shale/sandstone, striped bars = alluvial/granite.

The actual flux of bioavailable phosphorus into the sediment was not known, however, the mean phosphorus content of hypolimnetic water was (n=26). When comparing the relative value of bioavailable P and hypolimnetic TP (Hypolimnetic TP/Bioavailable P), in particular Reservoirs 23, 26, 27 and 6 showed higher ratios. These reservoirs therefore have higher release rates than the other reservoirs, relatively speaking, with most favourable conditions for release. Certain conditions seem to release phosphorus from sediments or hinder uptake of phosphorus from the water. Due to the extremely different status in bioavailable P content, different conditions are suspected to cause the respectively higher than expected hypolimnetic P (e.g. geological background, length of hypolimnetic anoxia, production of trout in relation to reservoir size). In contrast, four reservoirs had relatively lower hypolimnetic P concentrations than what was expected from their bioavailable P proportions, namely Reservoirs 1, 5, 6 and 24 (the latter was the deepest reservoir, but production was not in the deepest area). Since total phosphorus content of all these sites differed widely (20 and 75 % of TP (BAP/TP)), total phosphorus content was not a good predictor of actual phosphorus release rates.

Table 5.3: Mean total and bioavailable phosphorus content in sediment and hypolimnetic water, and P-related ratios of ten reservoirs in the Western Cape. BAP = Bioavailable P (sum of AAP = algal available P, WSP = water soluble P and RDP = readily dissolvable P).

	Total P	BAP	BAP/TP	Hypo limn. TP	Hypo limn. TP/ BAP	Bray II P	S:BAP ratio	Anoxic period	Hypolimnetic Fe^{2+}
	mg/kg	mg/kg	%	µg/L	µg/L	mg/kg		months	mg/L
1	133.0	54.8	41.2	55	1.3	18.3	1.32	1.0	0.60
3	166.3	62.1	37.3	160	3.9	5.0	0.28	0.0	0.10
5	207.0	89.7	43.3	95	1.7	25.5	0.41	6.0	0.10
6	448.0	225.9	50.4	260	5.2	11.3	6.19	6.0	0.10
32	58.8	20.7	35.2	141	4.0	42.0	1.05	3.5	0.09
26	105.2	79.1	75.2	613	8.2	22.5	0.99	7.5	3.31
22	155.8	45.4	29.1	131	4.0	13.3	0.28	1.5	0.13
23	111.8	22.1	19.8	183	9.3	22.8	0.27	1.5	0.06
24	253.5	94.2	37.2	123	3.3	136.5	0.45	3.5	0.17
27	443.1	194.1	43.8	624	12.0	58.7	1.79	8.5	1.87
27 cage	769.3	311.2	40.5	624	16.3	519.3	1.59	8.5	1.89

Bray-II-P is a method used to determine plant available phosphorus in soils or sediments. Although primarily applied in soil science, this method has been used in other sediment analyses in South Africa (Grobler & Davies 1981). The Bray-II P concentration was usually lower than the bioavailable phosphorus fraction, with the exception of three reservoirs, where the concentration was slightly higher, viz: Reservoirs 24 and 32 (both with fish) and in Reservoir 27 (with fish) were the concentration almost doubled at the cage locality.

The ratio of sulphur to phosphorus content was interesting in terms of the competition between the two ions for iron. In Reservoir 6, the influence of the elevated sulfide concentration on the binding capacity of ferric or ferrous phosphate was strong under acidic conditions, but had a lower influence under the alkaline sediment conditions in this reservoir.

Table 5.4: Mean values (n=4) of four phosphorus fractions in the sediment of ten reservoirs, as wet weight. (AAP = algal available phosphorus, TP = total phosphorus, WSP = water soluble phosphorus, RDP = readily desorbable phosphorus).

	Additional observations	Total P	Mean AAP	AAP:TP	Mean WSP	WSP:TP	Mean RDP	RDP:TP
		mg/kg	mg/kg	%	mg/kg	%	mg/kg	%
Reservoir/ Range		59-769	20-297	18-70	1.3-15.6	1.1-4.8	0.5-2.4	0.2-1.0
Average		259.3	101.5	37.9	6.4	2.7	1.1	0.5
32 (fish)		58.8	18.6	31.7	1.6	2.8	0.5	0.8
26 (fish)		105.2	74.0	70.3	4.4	4.2	0.6	0.6
23 (fish)		111.8	20.2	18.1	1.3	1.1	0.6	0.5
1		133.0	48.3	36.3	5.3	4.0	1.1	0.8
22 (fish)		155.8	41.3	26.5	3.2	2.1	1.6	1.0
3		166.3	57.0	34.3	4.1	2.5	1.0	0.6
5	trees	207.0	84.3	40.7	4.3	2.1	1.1	0.5
24 (fish)	trees	253.5	81.5	32.1	12.1	4.8	0.6	0.2
27 (fish)	birds	443.1	186.7	42.1	6.4	1.4	1.0	0.2
6	birds	448.0	207.9	46.4	15.6	3.5	2.4	0.5
27 cage (fish)	birds	769.3	296.9	38.6	12.6	1.6	1.8	0.2

Hypolimnetic ferrous iron varied between 0.06 and 0.60 mg/L, with two reservoirs showing much higher ferrous iron contents, namely Reservoir 27 with 1.87 and 1.89 mg/L and Reservoir 26 with 3.31 mg/L (landslide reservoir).

The four evaluated phosphorus fractions in the reservoirs were total phosphorus (TP), algal available phosphorus (AAP), water soluble phosphorus (WSP) and readily desorbable phosphorus (RDP). Their relation to each other is described in Table 5.4. The algal available fraction was highest (18 to 70 % of TP), with WSP the second most important fraction (1.1 to 4.8 % of TP) and RDP accounted for the smallest proportion of bioavailable phosphorus (0.2 to 1.0 % of TP).

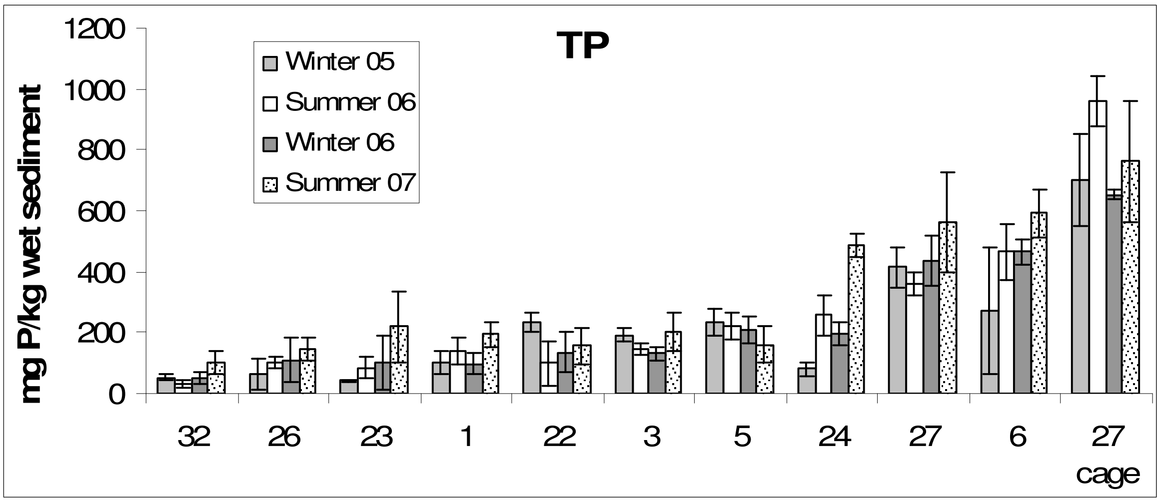

Figure 5.3: Total phosphorus with seasonal differences in ten reservoirs (ranked according to the mean of the four values).

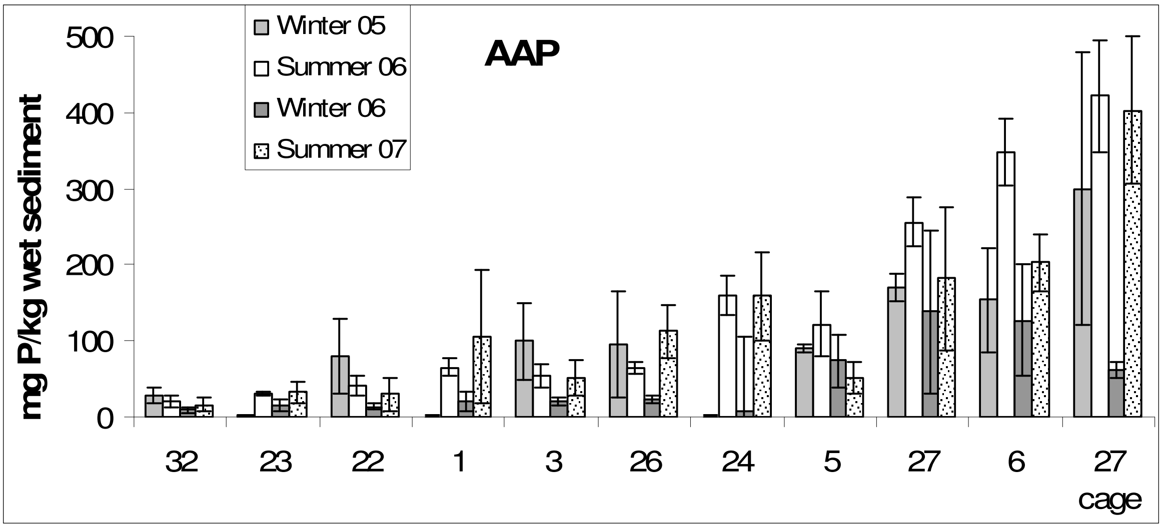

Figure 5.4: Algal available phosphorus with seasonal differences in ten reservoirs (ranked according to the mean of the four values).

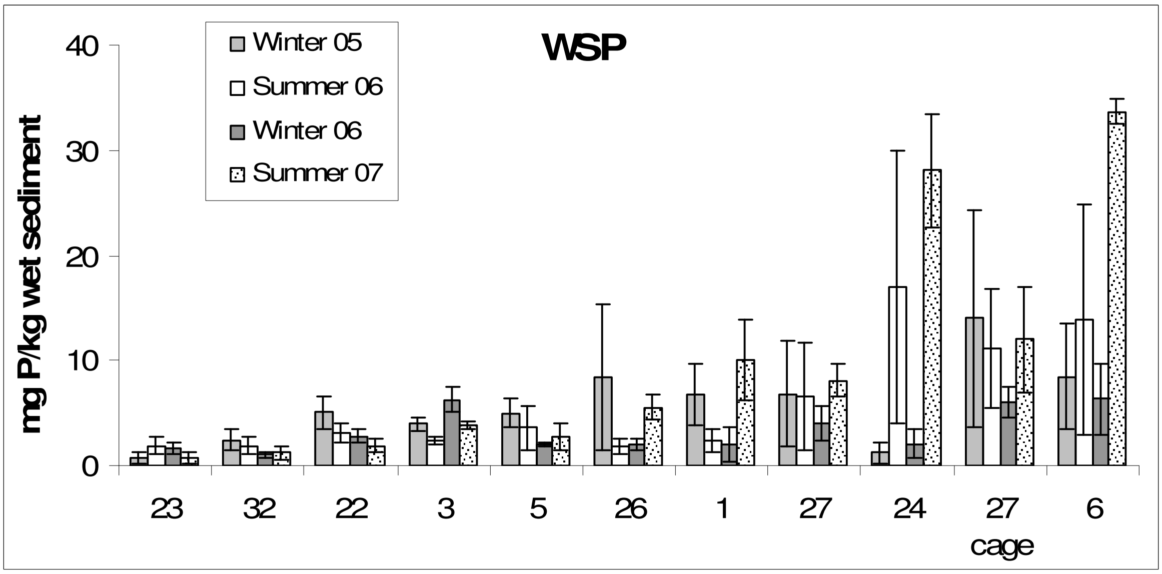

Figure 5.5: Water soluble phosphorus with seasonal differences in ten reservoirs (ranked according to the mean of the four values).

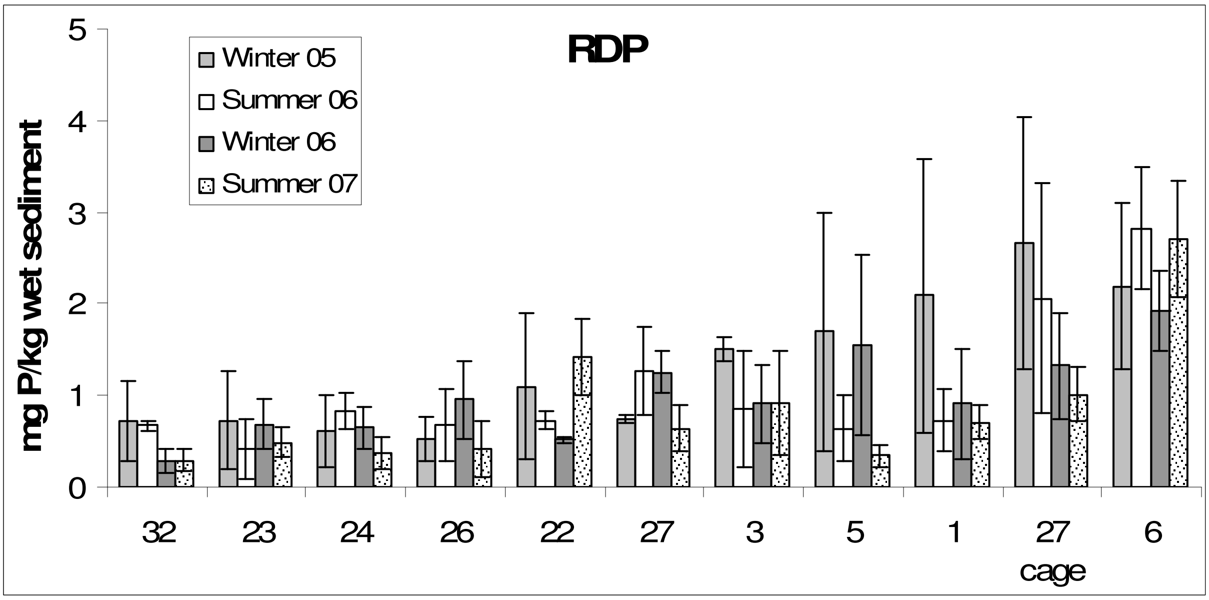

Figure 5.6: Readily desorbable phosphorus with seasonal differences in ten reservoirs (ranked according to the mean of the four values).

The seasonal variability of the fractions varied from reservoir to reservoir. Total phosphorus was evenly distributed throughout the four sampling dates in most reservoirs with similar total phosphorus content in the sediments in late winter (August/September), in a period when lake water was usually evenly mixed, and in late summer (February/March), in a period when most reservoirs were stagnated with an established thermocline (Figure 5.3).

If discernible differences occurred between winter and summer samples (Figure 5.3), the tendency was towards higher TP levels in samples collected in late summer (April/May, congruent with the late stagnation phase). Most production sites showed a tendency towards an increase of released phosphorus, most significant in Reservoirs 23, 24, 27 and 32 (summer 2006 with summer 2007; $p<0.01$).

The trend of higher summer than winter P release was more pronounced with AAP (Figure 5.4). In particular, the AAP values of samples collected under anoxic sediment conditions were higher in reservoirs with the highest TP concentrations. Winter and summer phosphorus values differed significantly ($p<0.01$) in Reservoirs 1, 6, 24, 26, 27, and 27 cage, indicating a phosphorus release from the sediments to the hypolimnion during summer.

The seasonal variation of WSP was primarily even (Figure 5.5). Reservoirs with the highest TP release levels also had the tendency towards elevated concentrations in summer, and RDP concentrations showed no clear tendency (Figure 5.6).

In Reservoir 27, with two separate sampling localities, samples from beneath the cages showed higher TP concentrations and also released more phosphorus into the environment (AAP, WSP and RDP). In summer 2007, the cage was moved to another location and the sampling continued at the old cage location where the three samples showed a consequently higher variation than previously as a consequence of cage dislocation.

5.3.3 Phosphorus cycle and release rates

According to the above results, the processes driving the overall phosphorus sediment release in these reservoirs were primarily the processes driving the release of algal available phosphorus. The equivalent phosphorus fraction consisted of the non-apatite inorganic components of P which were mostly influenced by pH and redox-potential (Fe, Al, Mn and Ca bound). The AAP values and the length of the anoxic period (months of hypolimnetic hypoxia) correlated positively ($p<0.05$), and with increasing TP, the AAP release also increased ($p<0.01$). Higher iron and sulphur concentrations meant higher AAP concentrations ($p<0.05$ in both cases) and the higher the moisture content, the more AAP was released ($p<0.01$). Other factors influencing AAP concentrations were the years of trout production ($p<0.05$) and the presence of birds ($p<0.01$), while better water exchange rates meant lower AAP concentrations ($p<0.05$).

5.4 Discussion

The geology of the south-western African tip is very young and diverse. The sediments of artificially constructed irrigation reservoirs were influenced by the surrounding geology, especially in basins that were naturally flooded with or without compaction of the soil. Only rarely are basins specifically constructed and certain sediment layers introduced. When filled with water, sedimentation of organic material will differ depending on the directly surrounding landscape and the internal ecology. Texture, structure and composition of the reservoir sediment determines binding capacities and release rates of nutrients. Minerals and nutrients introduced with aquaculture wastes (faeces, uneaten feed) will be bound and will also add to the organic component of reservoir sediments. For all but one (very young) reservoir, it was assumed that the upper 10 to 15 cm of sediment material derived from recent (10 to 20 years) organic and inorganic input.

Total phosphorus (TP) in sediments was not generally higher at trout production sites, which indicates that other phosphorus input sources or geology exceeded the effects of aquaculture. Total phosphorus content in sediments could, however, be explained by a combination of runoff volume with the proportion of cultivated land use in the catchment area (and is hence influenced by agriculture), which was also treated as the main phosphorus contributor in the South African reservoir eutrophication model (Grobler 1985).

One to two years of trout production was not sufficient to show differences in sediment TP, however, a significant difference was discernible in the 11 year old production site. The reservoir with 11 years of continuous trout production was 72 % higher in its sediment TP content than the reservoir without fish production within the same reservoir. The site with 7 years of production (1 gap year) was in the mid-section of all studied sites regarding sediment TP, with slightly lower (minus 7 %) concentrations than the reference reservoir (a directly neighbouring reservoir). An explanation for this may be that the cages were moved within the deeper area of the reservoir spreading the aquaculture waste to a larger area (thus a positive effect), and the water exchange rate of the reservoir was higher than in the reference reservoir (0.8 in reservoir without production, 3 in reservoir with production).

The dominating processes driving inorganic phosphorus release from sediments were shown to be related to AAP release (the greatest component of phosphorus release from sediments). Algal available P was determined via 0.1M-NaOH extraction and amounted to between 27 and 46 % (2 exceptions, a newly constructed reservoir with 18 % and a reservoir that experienced a landslide with 70 %). Grobler and Davies (1981) found an average of 32.7 % of TP to be algal available P in five Western Cape reservoirs (ranged from 18 to 50 %). Another author described a 60 % AAP for a highly eutrophic South African reservoir (Twinch 1986). Grobler and Davies (1981) used bioassays for AAP determination (30 d test), however the more rapid NaOH-extraction seems to deliver comparable results (Sharpley et al. 1995).

The AAP fraction consists primarily of metal associated phosphorus, with the acidic and soft water conditions of the hypolimnion and the sediments, probably primarily iron (in addition possibly aluminium and manganese which were not measured). An exception was Reservoir 6 which had a weakly alkaline sediment, low iron content, high sulphur content and a much higher calcium content than other reservoirs. Calcium binding processes are suggested to dominate in this reservoir. Under these conditions, calcium can flocculate phosphorus and lower its availability (Grobbelar & House 1995). The geology of the reservoirs differed and so did the iron and water content of the sediments, however, AAP was more influenced by the total phosphorus content of the reservoirs than the geological background and would therefore also be influenced by the overall phosphorus budget and mass balance. When comparing the AAP value, which is in a sense theoretical and describes the maximum algal available phosphorus release under certain conditions, to the actual total phosphorus concentrations measured in the hypolimnetic water, the reservoirs of highest trophic state and the newly constructed reservoir show the highest release rates relative to their potential. This indicates that hypolimnetic conditions play a decisive role in release from sediments (internal loading) during the summer stagnation, rather than geological background. Overall, surface phosphorus concentration was influenced by the geological background (Chapter 3), combined with the sediment information supposedly by inflowing particles and their binding capacities rather than the character of the sediment.

Organic phosphorus was not determined in the present study, but Grobler & Davies (1981) found AAP to be 70 ± 20 % of the inorganic phosphorus fraction in five Western Cape reservoirs (65 ± 25 % for 28 South African impoundments). Therefore, the organic phosphorus fraction would vary between 40 and 80 % of sediment TP (52 % using all average concentrations). A relationship was found between land use and the AAP:TP ratio (representing the non-apatite organic fraction), suggesting that sites surrounded by natural vegetation accumulate less inorganic P (non-apatite) (27 % of TP) in their sediments than agriculturally influenced sites which accumulate more non-apatite inorganic P (46 % of TP).

There was a significant trend towards a higher AAP fraction of TP in reservoirs without fish production (39.4 ± 4.9 %) than in reservoirs with fish production (29.4 ± 7.6 %). This finding concludes a higher organic phosphorus concentration in production sites than reservoirs without aquaculture. This is additionally supported by a very low inorganic P content in a very young site with aquaculture production and a very high inorganic P fraction in a reservoir with a high sediment load from a landslide (prior to commencement of this study).

Sulphur concentrations were at 335 mg/kg and 818 mg/kg in two of the reservoirs. At all other sites, the sulphur concentrations remained <100 mg/kg. Under anoxic hypolimnetic conditions, sulfates are reduced to sulfide, and FeS bonds will be stronger than iron-phosphate bonds (usually $Fe_3(PO_4)_2$) and releaseing phosphate as an ion.

Total carbon was very low within Reservoir 23, a very young reservoir (with fish), which suggests that little organic biomass was deposited within the short period post construction. The introduction of organic material (TOC) depends on nutrient availability and algal growth, other biota depending on algae and also runoff, lake surroundings (vegetation) and birds. The source can be allochthonous (from outside the lake) or autochthonous (from within the lake).

Organic and inorganic phosphorus seem to be equally important in phosphorus availability in Western Cape reservoirs. Inorganic non-apatite P is controlled by redox processes and oxygen supply, whereas microbial conversion of organic P turns P bioavailable. Not only are biotic processes important in organic phosphorus conversion (Karesalo et al. 1995), but the sediment bacteria themselves bind phosphorus (about the same amount of P that is usually settled with organic debris) (Gächter & Meyer 1983) and release it under anoxic conditions, similar to inorganic processes (e.g. iron reduction). Therefore, winter and turnover periods would more effectively bind phosphorus, while in summer during stagnation, more phosphorus was measurable as AAP (more easily released from metal associations) in the sediments and in hypolimnetic water (Chapter 4). The elevation of hypolimnetic pH (from acidic to a pH>7) was shown to allow better phosphorus mobilisation (Koski-Vähälä et al. 2001). However, Peng et al. (2007) showed the optimal immobilisation of pH by metals to vary between 7 and 8 and Haygarth and Jarvis (1999) suggested that most phosphorus was immobilised between a pH 5 and 6. The sediment pH measured in this study primarily ranked between 4 and 6, and indeed, the reservoirs with the higher surface phosphorus concentrations had pH sediment concentrations of either <5 or >7, whereas a pH between 5 and 6 seems to favour phosphorus release from metal associations. A more detailed knowledge of the actual sediment components (metal complexes) and the microbial activity will be necessary to confirm these findings.

Since effects of aquaculture production were discernible in surface water ecology, and especially in winter algal biomass (Chapter 4, Figures 4.7 and 4.8), this finding is congruent with the release of organic and inorganic phosphorus in anoxic conditions and their availability with the first turnover events (April/May). During the winter period with oxygenated hypolimnetic water, the conversion of organic material by bacteria might be the source of additional nutrients. Furthermore, during the turnover period the hypolimnetic pH of some highly eutrophicated reservoirs neutralises or becomes alkalinic (Chapter 4), which supports the increased phosphorus release.

With hypolimnetic water, there was a clear trend towards elevated total phosphorus concentrations in production sites in contrast to reservoirs without fish production (Chapter 4, Figures 4.3 and 4.4), and these were strongest during the period of stratification (summer). The upper values did differ amongst different sediment types in reservoirs without fish production. However, in the production sites, the upper total phosphorus concentrations in the hypolimnion were similar which might support different mobilisation mechanisms in reference and production sites (higher bacterial mobilisation in production sites).

With the current Fe:P ratio (ranging from 0.4 to 5.5), internal loading plays a great role within the reservoirs (Jensen et al. 1992), even more so with the additional anoxic conditions in the sediments. External loading (runoff, aquaculture, leaf litter, birds) and internal loading (sediment-water processes) with their seasonality and varying quantities, jointly define the phosphorus availability in these reservoirs.

Several studies showed the impact of cage aquaculture on sediment to be localised and affect sediments up to 50 m from the cages in low-current water (Environment Canada 2009, Alpaslan & Pulatsü 2008, Cornel 1993 & Whoriskey, Dobrowolski 1988). This means that accordingly an approximate sediment area of 0.2 to 1.0 ha is affected with the cage operations in the Western Cape, enforcing or leading to hypoxia and increased internal phosphorus release during stagnation, which leads to a high overall phosphorus content in the reservoir water once lake mixing sets in.

5.5 Conclusions

The present phosphorus content of reservoir sediments is primarily influenced by material introduced by runoff (suspended solids) and introduction of organic material (trees and birds). Similarly, in sediments underneath cages, the phosphorus release during stagnation was increased (AAP fraction), which with the relatively small reservoirs (<3 to 5 ha) affected the whole hypolimnion. The increase in release could either have happened by a longer anoxic period, a phenomenon increased by aquaculture production (Chapter 4), or the concurrent effect of iron-bound phosphorus release with elevated microbial activity. The effect of internal re-loading starts from the first year of production, and there was no difference in phosphorus release between 1 or 11 years of production (probably due to fixed locations, high sedimentation rates and sediment changes in the first year, but similar changed sediment surface in the second and consecutive years – once shifted, always shifted). This effect is presumably caused by the relatively high loading from aquaculture in contrast to other external phosphorus sources (34-90 % of P input from aquaculture, Chapter 6). The most aggravated re-loading (highest hypolimnetic water TP concentrations) was observed in production sites with long anoxic periods (>7 months). Generally, the more phosphorus introduced into the reservoirs while production lasts (relative to reservoir size), the longer the sites will continue to be affected by internal loading, especially in the post-production period.

Depending on the reservoir basin shape and the location chosen for the cage, an area of 0.2 to 1.0 ha can be affected by aquaculture waste. It is recommended to keep a minimum area of three-fold the affected area (of the hypolimnetic section – deeper 6 to 7 m at full capacity) unimpacted by waste. Movement of cages seemed to have a good effect on sediment phosphorus content and its even distribution.

The phosphorus concentration beneath the cages in contrast to 50 m away from the cages (Reservoir 27) was found to have increased by 70 % after 11 years of production, which suggests a TP increase of approximately 6 % per year. A positive correlation between TP and AAP was found (with the true phosphorus release rate probably varying between WSP and AAP, approaching AAP with longer anoxic conditions), so that phosphorus release within a reservoir will be proportionally increased from the moment of production commencement, with a hypothesized upper maximum for remobilisation.

To confirm a shift in the binding and remobilisation processes of the sediments with aquaculture waste introduction, a thorough study of sediment chemical components and complexes as well as sediment microbial activity would be necessary.

No differences in sediment TP was found between reservoir and production sites, which leaves the open question as to why the difference is so apparent in the surface TP concentrations and the hypolimnetic water quality parameters. From the current results, the age of the reservoirs and overall introduction of material by runoff was the main driver of TP concentrations in the upper sediment layer. Further studies are necessary to link the findings in surface and bottom waters to mechanisms in the reservoir sediment.

5.6 References

Alpaslan, A. and Pulatsü, S. (2008). The Effect of Rainbow Trout (*Oncorhynchus mykiss* Walbaum, 1792) Cage Culture on Sediment Quality in Kesikköprü Reservoir, Turkey. Turkish Journal of Fisheries and Aquatic Sciences 8: 65-70.

APHA, AWWA and WEF (2005). Standard Methods for the Examination of Water and Wastewater. Standard Methods online. American Public Health Association, American Water Works Association, Water Environment Federation. http://www.standardmethods.org.

Branom, J. R. and Sarkar, D. (2004). Phosphorus bioavailability in sediments of a sludge-disposal lake. Geosciences 11(1): 42-52.

Christophoridis, C. and Fytianos, K. (2006). Conditions Affecting the Release of Phosphorus from Surface Lake Sediments. Journal of Environmental Quality 35(4): 1181-1192.

Cornel, G. E. and Whoriskey, F. G. (1993). The effects of rainbow trout (Oncorhynchus mykiss) cage culture on the water quality, zooplankton, benthos and sediments of Lac du Passage, Quebec. Aquaculture 109: 101-117.

Dobrowolski, Z. (1988). The effect of cage aquaculture of rainbow trout on the distribution and stability of macrobenthos in eutrophic Lake Letowskie. Polish Journal of Ecology 35(3-4): 611-638.

Environment Canada (2009). Organic Waste and Feed Deposits on Bottom Sediments from Aquaculture Operations: Scientific Assessment and Guidance. Ecosystem Health: Science-based Solutions Report No. Gatineau, Quebec, Environment Canada, National Guidelines and Standards Office.

Gächter, R. and Meyer, J. S. (1983). The role of microorganisms in mobilization and fixation of phosphorus in sediments. Hydrobiologia 253(1-3): 103-121.

Grobbelaar, J. U. and House, W. A. (1995). Phosphorus as a limiting resource in inland waters; interactions with nitrogen. In Tiessen, H.: Phosphorus in the global environment: Transfers, cycles and management. Chichester: Wiley.

Grobler, D. C. and Davies, E. (1981). Sediments as a Source of Phosphate: A Study of 38 Impoundments. Water SA 7(1): 54-60.

Grobler, D. C. (1985). Phosphorus budget models for simulating the fate of phosphorus in South African reservoirs. Water SA 11(4): 219-230.

Haygarth, P.M., Jarvis S.C. (1999). Transfer of phosphorus from agricultural soils. Advances in Agronomy 66: 195-249.

Jensen, J. P., Kristensen, P., Jeppesen, E., and Skytthe, A. (1992). Iron-phosphorus ratio in surface sediment as an indicator of phosphate release from aerobic sediments in shallow lakes. Hydrobiologia 235-236: 731-743.

Karesalo, T., Tuominen, L., Hartikainen, H., and Rankinen, K. (1995). The role of bacteria in the nutrient exchange between sediment and water in a flow-through system. Microbial Ecology 29(2): 129-144.

Kelly, L. A. (1993). Release rates and biological availability of phosphorus released from sediments receiving aquaculture wastes. Hydrobiologia 253: 367-372.

Koski-Vähälä, J., Hartikainen, H., and Tallberg, P. (2001). Phosphorus Mobilization from Various Sediment Pools in Response to Increased pH and Silicate Concentration. Journal of environmental quality 30: 546-552.

Lathrop, R. C., Armstrong, D. E., Hoopes, J. A., Karthikeyan, K. G., Mackay, D. S., Nowak, P., Panuska, J. C., Penn, M. R., Potter, K. W., and Wu, C. H. (2006). Measuring and modeling the source, transport and bioavailability of phosphorus in agricultural watersheds. USEPA, Wisconsin Department of Natural Resources, Madison, and UW Extension. USEPA REPORT R830669. Washington D.C., US Environmental Protection Agency.

Maleri, M. (2009). Site selection and production performance of rainbow trout (Oncorhynchus mykiss) cage operations in small farm reservoirs: the Western Cape experience, South Africa. Aquaculture Research 40: 18-25.

Manning, P. G., Birchall, T., and Jones, W. (1984). The partitioning of non-apatite inorganic phosphorus in sediments from lakes Erie and Ontario . The Canadian Mineralogist 22(2): 357-365.

Nichols, G. (1999). Sedimentology and Stratigraphy. Wiley-Blackwell. Malden, MA, USA.

Olsson, S., Regnell, J., Persson, A., and Sandgren, P. (2004). Sediment-chemistry response to land-use change and pollutant loading in a hypertrophic lake, southern Sweden. Journal of Paleolimnology 17(3): 275-294.

Peng, J.-F., Wang, B.-Z., Song, Y.-H., Peng, Y., and Zhenhua, L. (2007). Adsorption and release of phosphorus in the surface sediment of a wastewater stabilization pond. Ecological Engineering 31(2): 92-97.

Phillips, M. J., Clark, R., and Mowat, A. (1993). Phosphorus leaching from Atlantic salmon diets. Aquaculture Engineering 12(1): 47-54.

Rabitti, S., Boldrin, A., and Menegazzo Vitturi, L. (1983). Relationships Between Surface Area and Grain Size in Bottom Sediments. Journal of Sedimentary Research 53(2): 665-667.

Reynolds, C. S. and Davies, P. S. (2001). Sources and bioavailability of phosphorus fractions in freshwater: A British perspective. Biological Reviews of the Cambridge Philosophical Society 76(1): 27-64.

Sharpley, A. N., Hedley, M. J., Sibbesen, E., Hillbricht-Illkowska, A., House, W. A., and Ryszkowski, L. (1995). Phosphorus Transfers From Terrestrial To Aquatic Ecosystems. In H. Tiessen (ed.), Phosphorus in the global environment: Transfers, cycles and management. Chichester: Wiley.

Ting, D. S. and Appan, A. (1996). General characteristics and fractions of phosphorus in aquatic sediments of two tropical reservoirs. Water Science and Technology 34(7-8): 53-59.

Twinch, A. J. (1986). The phosphorus status of sediments in a hypertrophic impoundment (Hartbeespoort Dam): implications for eutrophication management. Hydrobiologia 135(1-2): 23-34.

Zhou, Q., Gibson, C. E., and Zhu, Y (2001). Evaluation of phosphorus bioavailability in sediments of three contrasting lakes in China and the UK. Chemosphere 42: 221-225.

CHAPTER 6 WARM MONOMICTIC RESERVOIRS AND RAINBOW TROUT (ONCORHYNCHUS MYKISS) CAGE PRODUCTION: PHOSPHORUS BUDGETS AND PRODUCTION CAPACITIES

Abstract

Inland trout cage farming introduces additional nutrients into lake systems and effects their ecology. Aquaculture and the introduction of fish feed changes the mass balance of phosphorus input. An annual phosphorus budget from data from a production period from August 2005 to October 2006 was determined for 10 reservoirs in the Western Cape. The reservoirs varied between 100,000 to 450,000 m^3 full capacity load (1.6 to 11.0 ha surface area). Six of the reservoirs had additional phosphorus input from 5 t net cage trout production systems.

The major phosphorus input into the reservoirs derived from inflowing water via rain-fed rivers and diffuse runoff, or in water from other reservoirs (11 to 425 kg P, depending on throughflow), while the main output of phosphorus occurred via overflowing structures and with water extraction for irrigation (16 to 495 kg P, depending on throughflow). Amongst 10 reservoirs, the overall phosphorus balance suggested a phosphorus negative or neutral balance for four reservoirs, a moderate gain in four reservoirs and a high phosphorus gain in two reservoirs.

The net P introduction by aquaculture ranged from 36.2 to 103.7 kg P, mostly depending on the food conversion ratio (FCR). When testing the additional waste load by aquaculture the six production sites can carry, only two reservoirs were ready to harbour 5 t of trout without phosphorus increase in the system. Only under ideal conditions (FCR 1.1, trout feed 1 % P content and an allowance of a 10 % P increase in the system), would a third reservoir have been found suitable. The carrying capacity calculation according to Beveridge (1984) suggested primarily 12 to 20 % higher production limits, in one case a lower production limit. Equivalently, only two sites were found suitable for sustainable trout production, both are characterised by water exchange rates of six to nine times per year.

6.1 Introduction

Aquaculture nutrient input into natural systems or within a temporary pond system needs to be known in order to estimate which load the respective system can carry without risking environmental degradation or harm to fish health. Historically, when aquaculture activities increased and spread, there was already an awareness of the process of eutrophication and various phosphorus mass balance models had been developed (Vollenweider 1968, 1975, Dillon & Rigler 1974, 1975, OECD 1982). The pressure on aquaculture to develop strategies of low environmental impact has also been increasing steadily (FAO 1995).

To understand the phosphorus balance in lakes and water bodies, nutrient budgets are useful tools (Schussler et al. 2007, Mueller 1982). In the process of developing a budget, both external and internal sources and sinks are identified and quantified. Additionally, hydrological and nutrient data on all major lake components, inflows and outflows are necessary (Boyd & Tucker 1998). In the end, long-term trends of phosphorus accumulation or reduction become apparent. When estimating the long term effect of aquaculture, each water body must be regarded individually with the relative impact of aquaculture within a certain system.

A different approach is mass balance modelling that includes aquaculture input. The most prominent models were developed by Beveridge (1984) and Persson (1991). The models worked well in some cases (Hamblin & Gale 2002), but failed in others (Hakanson et al. 1998). For example: Johansson and Nordvarg (2002) and Nordvarg (2001) reported that when using the existing classic mass balance models (Vollenweider 1968, 1975, Dillon & Rigler 1974, 1975, OECD 1982), the TP which was attributed to cage fish farms in lakes in Sweden, Scotland and Poland was overestimated by a factor of two because of an underestimation of the net phosphorus sedimentation. However, the Beveridge model allows a change of the sedimentation rate when local data are available and works with a mean value determined in about 200 temperate lake sites otherwise (Beveridge 1984). Another criticism with the model is the assumption regarding the steady state of nutrients with the principle conservation of mass. The assumptions include: (i) complete mixture of all substances in the lake at all times, (ii) phosphorus is defined as the limiting nutrient (TN:TP ratio), (iii) that phosphate is entering or leaving the water mass in equal proportions and (iv) that the proportion of phosphorus retained in the sediment is seasonally equal. Maleri et al. (2008) found the model suitable for small South African reservoirs due to the coincidence of mixing period (winter), water mass exchange and aquaculture production.

The severity of the impact of trout cage aquaculture in small South African reservoirs was initially underestimated and showed itself via emerging water quality problems. Without developing a full dynamic model such as e.g. PROTECH (Reynolds et al. 2001, Lewis et al. 2002, Moreno-Ostos et al. 2007) or DEPOMOD (Cromey et al. 2002) and adapting them to the additional input of aquaculture production, a nutrient budget and an adapted mass balance model from Beveridge (1984) were chosen as time-efficient, suitable tools. With the support of these two approaches, the overall system impact of aquaculture was quantified and the sustainable carrying capacity for trout cage production, with a 10 % maximum increase of average surface P concentrations, was determined.

6.2 Methods

Hydrological data and data on phosphorus content of different reservoir components was identified and quantified. The core study period was August 2005 to October 2006, with additional data collected from October 2006 to October 2007.

Most private irrigation reservoirs in the Western Cape are <15 ha in size with an oval to rectangular shape, with few exceptions where natural basins were flooded. Most in- and outgoing water masses and substances were obvious and were measured or estimated by simple means. An overview of all major components influencing the phosphorus budget is shown with Figure 8.1.

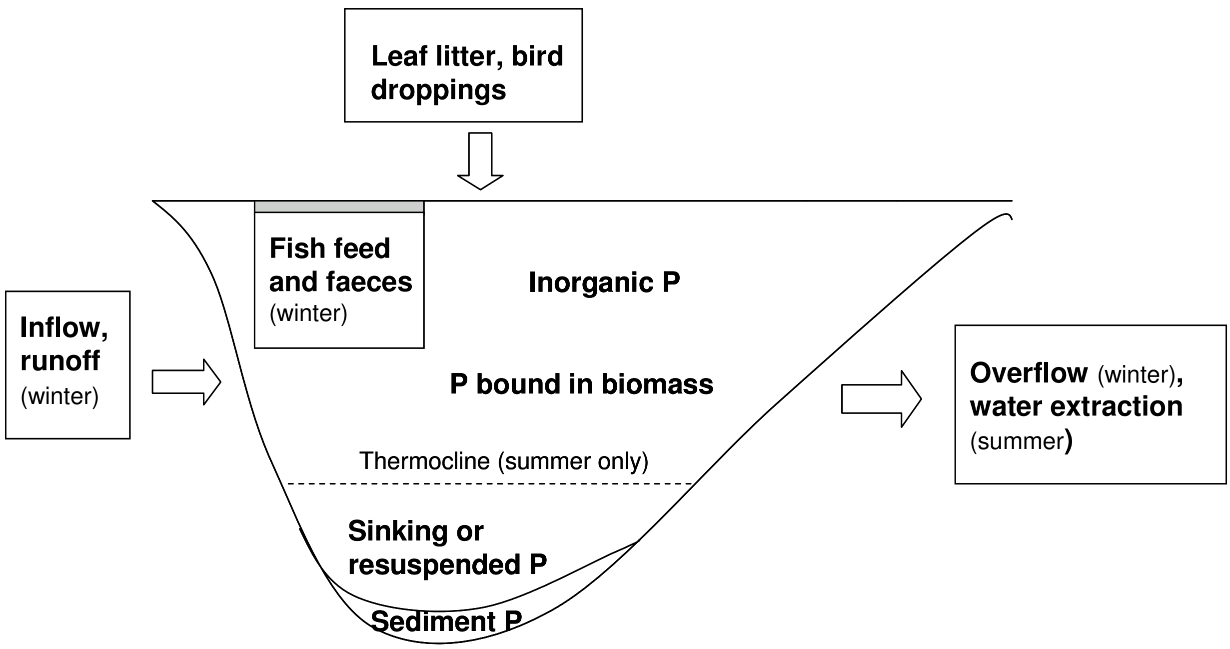

Figure 6.1: Generalised phosphorus budget model of Western Cape irrigation reservoirs. Inflow and runoff consisted of water from non-perennial rivers, influenced by land use in the watershed (specifically agriculture).

6.2.1 Sites

Ten sites were considered in this study, six with net cage production of rainbow trout (Reservoirs 22, 23, 24, 26, 27 and 32) and four reservoirs without fish production (Reservoirs 1, 3, 5 and 6). Within each reservoir, 5 t of rainbow trout were reared within four meter deep cage nets (stocking density up to 8 kg trout per m^3), with an average environmental stocking density of 900 fish per ha.

All sites were within the south-western area of the Western Cape Province, South Africa. They occurred in an area between 33°42'22'' S to 34°15'48'' S and 18°51'47'' E to 19°01'08'' E (Figure 3.1 and Figure 4.1), shaped like an upright rectangle (longer north-south axis) of 70 by 20 km. Cool, rainy winters and hot, dry summers characterize the region.

One of the four reservoirs without fish production was situated in sandstone and shale dominated geological substrate while the other two were characterized by granite and another one by alluvial material. Three of the production sites were granite based while the remaining three were sandstone and shale dominated. The average surface area of reference and production sites was similar at 5.5 and 5.6 ha respectively. The maximum depth and consequently volume, was approximately 30 % more at the production sites. Further information about the sites can be found in Chapters 3 and 4.

6.2.2 Hydrological data

The basin volume of the reservoirs was measured using the surface area and basin intersections. The length and width of each reservoir at full capacity was measured using Google Earth software (http://earth.google.com) and the surface area was estimated via the nearest geometrical shape (usually oval or rectangular). The basin shape was measured with transect measurements of the longest horizontal and vertical lines, most were commonly the central lines (Wetzel & Likens 2000), with depth measurements every 10 m in small and every 20 m in larger reservoirs. From the transect data, the volume was determined in stepwise 1 m vertical layers. The highest and lowest water volume was calculated using level measurements at metal droppers inserted at a suitable reservoir bank (water level, rope and meter) for the duration of the study period. Additionally, the depth records at the sampling stations served as level indicators and control.

The average inflowing water volume (cubic meter per hour) was estimated via flow velocity (float and stopwatch measurements) * flow area (measured at available structures such as concrete basins, channels or pipes) and these results were adjusted and complemented by information from farm managers. Outflowing water volume was measured in the same way at overflowing structures.

Yearly average rainfall for the study period and area was derived from data provided by the South African Weather Service (SAWS 2007). Evaporation was estimated using the average evaporation measurements by Vardavas and Fountoulakis (1996) for a comparative climatic region in Australia (3.7 mm/day).

Water extraction for irrigation was defined as the volume difference between highest supply volume (early spring) and lowest supply volume (summer) minus the loss due to evaporation. These data were adjusted and verified with records from some farm managers.
Introduction of water via runoff was the most difficult parameter to measure and could only be estimated as the difference between the sum of the outflowing water volume minus the sum of the inflowing water volume. Therefore, this value could also include an eventual gain of water by groundwater through-flow. Groundwater through-flow was assumed to be a steady-state, which would be most imprecise with reservoirs located in a strongly inclined landscape with different altitudes of terrain above and beneath the site.

6.2.3 Major phosphorus components

The major sources and sinks of phosphorus were shown graphically in Figure 6.1. They comprised of the inflowing water masses (rain-fed runoff water, river water or water from another reservoir, rainfall, leaves, birds) and their phosphorus content (Table 6.1). Groundwater influence on the P budget had to be neglected due to time and monetary constraints, but was regarded as negligible.

The phosphorus content of the standing water body, divided into the epi- and hypolimnion during stagnation, was determined. Organically bound phosphorus as phytoplankton, zooplankton, macrophytes and macroalgae could be estimated from the biomass of the respective organisms or plants. The surrounding landscape (land uses, trees within 10 m of the reservoir shoreline) and bird presence was recorded.

Total phosphorus (TP) of water was determined via acid-digestions of unfiltered reservoir water (and determination of P via the colorimetric method using the Molybdo-vanadate reaction)(Chapter 3 and 4). In order to determine the total dissolved fraction, filtered (0.45 µm mesh size) water was acid digested and to determine the total inorganic fraction, the phosphorus content of untreated, unfiltered water was measured (total reactive P, primarily inorganic). The total organic fraction was calculated as the difference between TP and total inorganic P, while total particulate P was defined as the difference of TP and total dissolved P (Toor et al. 2004). With additional determination of inorganic dissolved P (soluble, reactive P), a total of 9 different P species were calculated from four measurements (Total P, dissolved inorganic P, particulate inorganic P, dissolved organic P, particulate organic P, total inorganic P, total organic P, total dissolved P, total particulate P).

Table 6.1: Important lake component influencing the phosphorus budget and the methods of P content determination.

	Component	Method	P content
INPUT	Inflowing water	Water samples	Molybdo-vanadate method (TP)
	Runoff water	Indirect (balance of inflowing and outflowing water mass)	Mean P content in- and outflowing water contained in the respective water mass
	Rainfall (+ dust)	SAWS data, literature	Miranda & Matvienko (2003)
	Leaf litter	One-time tree count	Mean tree litter per tree (Palma et al. 1998)
	Bird droppings	Monthly bird counts	Average P content per average dropping amount of ducks, geese or cormorants (Marion et al. 1994)
	Aquaculture	Inserted and harvested fish counts, FCR	Calculation of P introduced with AquaNutro Fish feed (AquaNutro 2005, Beveridge 2004)
LAKE CONTENT	Epilimnion	Water samples	Total P and 4 subfractions (inorganic, organic particulate, dissolved)
	Hypolimnion	Water samples	Total P and 4 subfractions (inorganic, organic particulate, dissolved)
	Phytoplankton	Biomass determination (chapter 4)	P content calculated from biomass (Redfield ratio; Redfield 1963)
	Zooplankton	Biomass determination (chapter 4)	P content calculated from biomass (Uye & Matsuda 1988)
	Macrophytes	Presence and extension monitored	Average 556 g/m2 wet biomass (Rooney 2002), P content approx. 1 % of C content (Redfield ratio), WW:DW set as 3:1
	Macroalgae	Presence and extension monitored	Average 182 g/m2 wet biomass (Piper 2006), P content approx. 1 % of C content (Redfield ratio), WW:DW set as 3:1
SEDIMENT	Sediment	Sediment samples	Total P measurement and resuspendable fractions (chapter 6)
OUTPUT	Overflowing water	Epilimnetic water samples	Molybdo-vanadate method
	Irrigation water	Epilimnetic or hypolimnetic water samples	Molybdo-vanadate method
	Evaporation	literature	(negligible)
	Bird plant removal	Monthly bird count	Russell et al. (2009)

The phosphorus content of the sediment was also quantified (Chapter 5). The phosphorus content of water extracted for irrigation purposes during summer was derived from epi- or hypolimnetic water quality depending on the water extraction type (surface water or bottom water extraction).

The gross gain of P per reservoir will sink to the reservoir bottom eventually (e.g. after inorganic P has been bound as particulate organic biomass), while the water soluble fraction (released fraction) and potentially released fraction (AAP) help to estimate the retention of incoming P in the sediment as minimum and maximum retention capacity (100 %-(AAP/TP in %)) and (100 %-(WSP/TP in %)).

6.2.4 Phosphorus introduction by aquaculture

The impact of aquaculture on phosphorus concentrations by aquaculture was estimated using the number and average mass (g) of the introduced juvenile fish, as well as the number and average mass (g) of the fish at harvest. The percentage of P contained in whole fish was estimated as 0.48 % (Beveridge 2004). The difference between harvested and stocked fish allowed the calculation of phosphorus that was removed from the reservoir system. The FCR value, times the total tonnage of average fish weight in the reservoir, allowed the calculation of the total amount of feed introduced into the reservoirs. With a P content of 1.2 % of the used aquafeed (AquaNutro 2005) the final amount of phosphorus introduced by aquaculture was calculated via the difference between introduced P by feed, minus the P accumulated in fish biomass during the reservoir stage. All information regarding initial fish weight, harvested fish and FCR were kindly provided by the Hands-On Fish Farmer's Co-operative Limited management.

6.2.5 Models

As a first approach to estimate the impact of aquaculture (in addition to influences of agriculture, birds, trees etc.), an annual phosphorus budget (mass balance model) of the reservoirs was chosen where the phosphorus content of all relevant components (Figure 6.1 and Table 6.1) was transformed into average yearly phosphorus mass (using respective water volumes). Also, the model allows the indication of the main sinks of phosphorus and the proportional distribution thereof. The model uses a summated annual phosphorus budget.

In a second approach, the Beveridge model permits the calculation of the maximum carrying capacity of the currently used reservoirs. An allowance of a 10 % increase of the average surface phosphorus concentration was set. Reservoirs with opportunities to increase stocking intensity and reservoirs were overstocking takes place were identified. The model requires data on the average volume and surface area of the water body, the water exchange rate (calculated as total inflowing water divided by the average total volume of the water body) and a value called R_{fish} expressing the ability of the water body and the sediment to absorb P (Beveridge

2004). The FCR, the average P content of a whole fish (set as 0.48 % according to Beveridge (2004)) and the phosphorus content of the fish feed (1.4 was used as a seasonal average value of different trout feeds), and a theoretical Retention coefficient (average of more than 200 European Lakes) were also necessary to calculate R_{fish}.

The basic assumptions of the model were best served when using average winter and early spring data for the calculation, since complete water mixis can be postulated during that period. The Beveridge model estimates the optimal production capacity of a reservoir without changing the original average phosphorus concentration of the reservoir water by a given percentage (10 % was chosen).

6.3. Results

6.3.1 Hydrological information

The hydrological conditions encountered in 10 Western Cape reservoirs were summarised in Table 8.2. The surface areas varied between 1.7 and 11.0 ha, with maximum depths between 8.6 and 16.5 m and minimum depths between 2.5 and 11.0 m. The water exchange rates varied between 0.6 and 9.2, with two reservoirs with a water exchange less than the maximum supply volume and six reservoirs which exchange the water volume twice. The full supply volume ranged from 117 210 and 460 313 m^3, and differed with surface area, maximum basin depth and basin shape (conic or cuboid).

In five reservoirs the inflowing water mass (majority) was derived from non-perennial rivers (directed into the reservoir via channel or pipe), and from other reservoirs for three reservoirs (via pipe or overflow) (Table 6.7). In two reservoirs, the main input was in the form of runoff collected in spontaneous rivulets. These sources (summarised in Table 6.2 as inflowing water) amounted to 48 to 97 % of all incoming water and calculated diffuse runoff was 1 to 18 %. Rainfall accounted for 3 to 27 % (one exception of 49 %) of all incoming water. Groundwater flow was assumed to be proportional (yearly average), with similar in- and outflow volumes.

The outflowing mechanisms varied more. Four reservoirs had no overflowing structures, while the remaining six lost 23 to 90 % via overflow structures. Only two reservoirs were known to use water for cleaning and rinsing of cellars. The bulk of water was lost to irrigation (8 to 76 %). Evaporation losses relative to the outflowing water mass, accounted for 2 to 43 %. Relative to the reservoir full supply volume, evaporation losses accounted for 14 to 36 %, a severe loss, but unavoidable with surface water storage systems.

Table 6.2: The major hydrological parameters influencing the water exchange and flow dynamics in 10 Western Cape reservoirs from August 2005 to October 2006, given as a yearly mean value. Reservoir 1 to 6 were the reservoirs without fish production and 22 to 32 were the production sites.

Reservoirs	1	3	5	6	22	23	24	26	27	32
Surface area (ha)	1.7	7.4	1.8	11.0	6.5	7.3	7.2	1.6	4.2	6.9
Maximum depth (m)	12.0	9.8	11.0	9.1	15.1	12.1	16.5	8.6	11.7	16.1
Minimum depth (m)	4.6	7.5	3.8	2.5	8.1	7.1	11.0	5.2	8.9	7.1
Water exchange rate (max. volume exchange/y)	2.5	0.7	2.7	2.0	9.2	6.7	1.6	0.6	1.2	2.1
Minimum water volume (m^3)	37422	220000	36000	23000	127440	94920	243966	84851	146300	97344
Full supply volume (m^3)	117210	350000	122945	417280	424800	349440	460313	154275	209000	324480
Total volume water in (= total volume water out)	297147	229937	370454	842835	3887143	2353107	713583	91032	254721	693569
Water volume inflow (% of total water in)	85	48	89	73	97	93	70	55	63	82
Rainfall (%)	9	49	7	20	3	5	15	27	25	15
Runoff (m^3) (%)	6	3	3	7	1	2	14	18	12	3
Volume overflow (% of total water out)	65	0	23	0	90	85	56	0	0	54
Cellar usage (%)	0	0	0	36	0	0	0	0	39	0
Evaporation (%)	8	43	7	18	2	4	14	24	22	13
Irrigation (%)	27	57	70	47	8	11	30	76	38	33

Table 6.3: August 2005 to October 2006 differences between the hydrology of reference and production sites. Means of Table 8.2, separated into reference (mean ref) and production sites (mean prod).

	Mean	Mean ref	Mean prod
Surface area (ha)	5.6	5.5	5.6
Maximum depth (m)	11.8	10.5	13.3
Minimum depth (m)	7.0	4.6	7.9
Water exchange rate (max. volume exchange/y)	3.0	1.8	3.8
Average minimum water volume (m^3)	111124	79106	132470
Full supply volume (m^3)	292974	251859	320385
Total volume water in (= total volume water out)	988553	435093	1332193
Inflowing water volume (% of full supply volume)	77	74	77
Rainfall (surface and area dependent) (%)	17	21	15
Runoff/groundwater (%)	7	5	8
Overflowing water volume (%)	39	27	47
Extraction for cellar usage (%)	6	9	7
Evaporation (surface dependent) (%)	15	19	13
Irrigation (%)	40	45	33

Table 6.3 states the differences between the hydrology of reference and production sites. The production sites had higher water exchange rates than the reservoirs without fish production, and accordingly higher volumes of incoming water and a higher percentage of water lost to overflowing structures. The absolute mean gain and loss by rainfall and evaporation did not differ greatly between the two groups (similar mean surface areas), but the proportional influence of these two factors differed (the ratio of volume:surface area differed) and had a lower impact in the mean production reservoir. The production history of the six reservoirs was 2, 7 and 11 years, respectively.

6.3.2 Aquaculture input

The total phosphorus load in kg introduced into the respective reservoirs is shown in Table 6.4. FCR (amount of dry feed introduced relative to the wet weight gain of fish) and the total tonnage of fish are the two main determinants of the absolute amount of phosphorus input. Three reservoirs had low FCR values resulting in lower phosphorus input (< 100 µg/L) while another three reservoirs had outcomes with considerable addition of phosphorus.

Table 6.4: Production data for the respective reservoirs in the production season of winter 2006 and the resulting gross phosphorus load that the respective reservoirs need to absorb, expressed as dilution into the reservoir volume (in µg/L).

Reservoir	**32**	**23**	**24**	**22**	**26**	**27**	**AVERAGE**
FCR achieved in 2006	1.3	1.2	1.4	2.0	2.9	2.2	1.8
Harvested fish load (in kg)	6166	5772	6462	4843	4128	4821	5366
Total P introduced by aquaculture (in kg) = Net feed (P in feed + P in introduced trout – P in harvested trout)	39.6	36.2	47.3	66.9	103.7	76.2	61.7
P dilution into full supply reservoir volume	93	80	80	129	563	194	190

6.3.3 Phosphorus budget

Input, output, reservoir water and sediment phosphorus content are listed in Table 6.5. Most components could be included, except for bacterial biomass (and phosphorus content) which was only indirectly included as part of the dissolved organic phosphorus fraction (most bacteria measure 0.2 to 0.3 µm in diameter which is below the filter mesh size of 0.45 µm).

Table 6.5: Yearly average phosphorus input, distribution and output (in kg) of ten Western Cape reservoirs. Data were collected between August 2005 and October 2006. *) The total phosphorus content of the reservoir water included the different phosphorus species (in brackets). Bacteria formed part of the dissolved organic P fraction, and phytoplankton and zooplankton P (quantified) formed part of the particulate organic P fraction. Average input/output = reservoir turnover.

		Reservoirs without trout				Reservoirs with fish production					
P BUDGET	**Reservoir**	**1**	**3**	**5**	**6**	**22**	**23**	**24**	**26**	**27**	**32**
Reservoir descriptors	**Surface area (ha)**	1.7	7.4	1.8	11	6.5	7.3	7.2	1.6	4.2	6.9
	Max. volume (m^3)	117210	350000	122945	417280	424800	349440	460313	154275	209000	324480
INPUT	**Inflow**	20.9	11.2	38.0	160.6	416.3	345.9	50.0	5.0	35.0	32.3
(P content in kg)	**Rain**	0.7	3.1	0.7	4.5	2.7	3.0	3.0	0.7	1.7	2.9
	Runoff	3.6	1.2	2.6	11.9	7.2	10.4	20.7	3.3	2.9	4.3
	Birds	0.3	0.3	0.6	1.5	0.2	0.0	0.0	0.6	6.1	4.5
	Trees	0.6	0.9	2.3	4.2	1.1	0.0	18.4	1.5	0.5	4.9
	Introduced trout					5.0	4.9	4.1	6.8	5.2	3.7
	Feed					73.0	50.6	63.6	100.0	80.7	56.1
	Sum input	26.2	16.6	44.2	182.7	505.5	414.8	159.8	118.0	132.2	108.7
OUTPUT	**Overflow**	12.8	0.0	10.8	0.0	465.8	366.0	49.2	0.0	0.0	32.5
(P content in kg)	**Irrigation**	5.3	15.6	32.3	48.5	30.6	46.6	25.3	6.6	11.3	26.4
	Bird plant removal	0.6	0.5	1.2	2.9	0.3	0.0	0.1	1.2	5.7	8.8
	Cellar usage	0.0	0.0	0.0	88.2	0.0	0.0	0.0	0.0	11.5	0.0
	Harvested trout					23.3	27.7	31.0	19.8	23.1	29.6
	Sum output	18.6	16.1	44.3	139.6	520.0	440.3	105.6	27.6	51.6	97.2
BALANCE	**Gross P gain/y**	7.6	0.5	-0.1	43.1	-14.5	-25.5	54.2	90.6	80.6	11.5
RESERVOIR	**Total P**	7.4	41.7	15.9	62.2	48.5	41.2	30.8	28.2	27.4	26.9
(P content in kg)	**(Dissolved inorganic P)**	6.45	25.55	6.27	32.13	29.74	15.03	21.17	5.71	11.08	9.41
	(Dissolved organic P)	0.10	5.15	4.18	7.51	11.86	5.27	4.43	15.79	6.48	5.19
	(Particulate inorganic P)	0.02	5.45	2.70	1.27	2.12	6.97	0.02	3.24	6.73	7.30
	(Particulate organic P)	0.92	5.60	2.71	21.63	4.70	14.03	5.10	3.50	3.39	5.50
	(Phytoplankton)	0.2	0.8	0.7	7.4	0.8	1.0	1.5	2.1	1.6	1.3
	(Zooplankton)	0.9	1.5	1.1	4.6	3.8	3.3	3.6	1.3	1.3	4.3
	Macrophytes	0.2	1.5	0.7	0.7	0.7	0.7	3.7	1.1	4.5	2.2
	Macroalgae	0.0	0.0	0.2	9.0	0.0	0.0	1.2	3.0	0.2	0.0
	Fish	0.0	0.1	0.2	2.4	0.5	0.0	1.4	0.5	0.5	0.6
	Sum reservoir*	7.6	43.3	17.1	74.3	49.7	41.9	37.1	32.8	32.6	29.8
ANALYSES	**Turnover relative to sum reservoir P**	2.9	0.4	2.6	2.2	10.4	10.3	3.7	2.5	3.0	3.6
	Surplus particulate org. P	-0.2	3.3	0.9	9.7	0.1	9.7	0.0	0.1	0.5	0.0
	% P gain relative to sum reservoir	49.8	1.1	-0.6	36.7	-22.6	-37.9	59.4	73.4	71.2	27.8
	% org. P (of TP)	20.6	32.8	43.5	46.3	34.2	46.6	40.3	68.3	31.3	15.7
	% dissolved P	87.4	92.4	65.9	63.7	96.3	49.3	99.4	85.2	64.1	54.2
SEDIMENT	**TP**	27.7	799.9	2421.9	3520.9	658.3	530.5	1581.8	109.4	2305.7	351.6
(P content in kg)	**Release (WSP)**	1.1	19.7	5.0	111.5	13.5	6.2	75.5	4.6	39.3	9.6
	Pot. Release (AAP)	10.1	66.7	27.4	64.9	166.4	21.0	455.6	8.7	339.7	36.3
ANALYSES	**% WSP release relative to stock**	14	46	30	150	27	15	204	14	121	32
	% min. retention by sediment	59.7	89.2	98.7	95.0	72.7	94.9	66.4	87.8	83.6	87.0

Total input P varied between 16.6 and 505.5 kg P per year (Table 6.5). With reservoirs without fish production, the main input (67.5 to 87.9 %) of phosphorus entered the reservoirs via inflowing water (Table 6.6).

Table 6.6: Relative importance (%) of single components pertaining to phosphorus input, output, reservoir water and sediment. Percentages are related to the sum of input, the sum of output, the sum of the reservoir and sediment TP respectively. The phosphorus components were related to reservoir TP. Phytoplankton and zooplankton were independently related to TP, but actually form part of particulate organic P.

		Reservoirs without fish				Reservoirs with trout production					
P BUDGET	**Reservoir**	**1**	**3**	**5**	**6**	**22**	**23**	**24**	**26**	**27**	**32**
INPUT	Inflow	79.8	67.5	86.0	87.9	86.3	89.4	38.8	5.1	32.1	40.8
As % of sum input	Rain	2.7	18.5	1.6	2.5	0.6	0.8	2.3	0.7	1.6	3.6
	Runoff	13.7	7.2	5.9	6.5	1.5	2.7	16.1	3.4	2.7	5.5
	Birds	1.5	1.5	1.4	0.8	0.0	0.0	0.0	0.6	5.6	6.2
	Trees	2.2	5.3	5.2	2.3	0.2	0.0	14.3	1.6	0.5	6.2
	Net feed					11.4	7.2	28.5	88.6	57.6	38.2
	Sum input	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**
OUTPUT	Overflow	68.5	0.0	25.2	0.0	93.8	88.7	66.0	0.0	0.0	48.1
As % of sum output	Irrigation	28.3	97.0	72.1	34.7	6.2	11.3	33.9	84.6	39.7	39.0
	Bird plant removal	3.2	3.0	2.7	2.1	0.1	0.0	0.1	15.4	19.9	12.9
	Cellar usage	0.0	0.0	0.0	63.2	0.0	0.0	0.0	0.0	40.4	0.0
	Sum output	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**
RESERVOIR	Total P (as % of sum reservoir)	96.6	96.2	93.0	83.7	97.5	98.3	83.0	86.0	84.0	90.5
As % of TP	(Diss. inorganic P)	84.0	61.3	39.5	51.7	61.3	36.5	68.7	20.2	40.5	34.9
	(Dissolved organic P)	1.3	12.4	26.4	12.1	24.5	12.8	14.4	55.9	23.7	19.3
	(Part. inorganic P)	0.3	13.1	17.1	2.0	4.4	16.9	0.1	11.5	24.6	27.1
	(Particulate organic P)	14.3	13.4	17.1	34.8	9.7	34.0	16.6	12.4	12.4	20.4
	(Phytoplankton & zoo) (independent)	15.1	5.4	11.5	19.2	9.6	10.4	16.6	12.1	10.4	20.6
As % of sum reservoir	Macrophytes	2.9	3.5	4.3	1.0	1.5	1.7	10.0	3.4	13.8	7.4
	Macroalgae	0.0	0.0	1.4	12.1	0.0	0.0	3.2	9.1	0.6	0.0
	Fish	0.5	0.3	1.3	3.2	1.0	0.0	3.8	1.5	1.5	2.1
	SUM Reservoir	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**
SEDIMENT	**Release (WSP)**	4.0	2.5	0.2	3.2	2.1	1.2	4.8	4.2	1.7	2.7
As % of TP	**Pot. Release (AAP)**	36.3	8.3	1.1	1.8	25.3	4.0	28.8	8.0	14.7	10.3
	TP	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**

Other sources with a proportionally great influence were runoff >5 % in six reservoirs. In one reservoir tree litter contributed 14.3 % to total P input, while in another three reservoirs P input by leaves exceeded 5 %. In one reservoir with very low P input via inflowing water and a low water exchange rate, rainfall supplied 18.5 % of the total input P.

With four of the six reservoirs with fish production, the phosphorus input via feed varied between 28.5 and 88.6 % of all incoming P (for absolute numbers refer to Table 6.5) and therefore feed contributed to the majority of phosphorus introduction. With two other production reservoirs, fish feed input contributed 9.2 and 13.5 % to the yearly input of phosphorus, respectively. These two reservoirs had water exchange rates of >5 times per year, an indication of the extremely important role of water exchange rates.

In reservoirs without aquaculture, inflowing water accounted for the majority of P input (67.5 to 87.9 % of all input). In Table 6.7, the origin of the inflowing water is indicated as well as the phosphorus concentration of the main source. The annual sum of all input related to the surface area is often an applied value, and for the study reservoirs varied from 0.2 to 8.4 g/m^{2}y.

Table 6.7 Parameters to estimate the quality of inflow water, including: source, phosphorus concentration and the overall sum of P input into the reservoirs per m^2 surface area.

INFLOW LOAD	**1**	**3**	**5**	**6**	**22**	**23**	**24**	**26**	**27**	**32**
Origin of inflow water	Palmiet River	Kromme River	Theewaters-kloof Dam, farm dam	Farm dam, rivulet	Kromme River	Theewaters-kloof Dam, Du Toits River	Farm Dam	Farm Dam	Cellar, rivulet	Swart River
Inflowing phosphorus concentration (µg/L)	83	101	115	311	111	158	113	201	219	57
Sum input/surface area (g P/m2/y)	1.5	0.2	2.5	1.7	8.0	5.8	2.4	8.4	3.5	1.7

The loss of P occurred via overflowing structures (25.2 to 93.8 % when applicable, average 65.0 %) and irrigation water (6.2 to 97.0 %, average 47.7 %) or cellar water (40.4 and 63.2 % when applicable). In three reservoirs, phosphorus loss by birds (feeding on submerged macrophytes and macroalgae) was 12.9 to 19.9 % (Table 6.6).

The sum of phosphorus present in reservoir water (including semi-permanently bound macrophytes, macroalgae and fish, while macrozoobenthos was not measured), was primarily made from total phosphorus retrieved with the unfiltered mid-lake water samples. Within this group, the dominant fraction was dissolved inorganic P (39.5 to 87.4 %) with the exception of Reservoir 26 where dissolved organic phosphorus formed the main fraction (55.9 %) (Table

6.6). Another two reservoirs had a relative high proportion of dissolved organic P (Reservoirs 5 and 22) as the second largest P fraction. In the residual reservoirs, the second largest proportion after dissolved inorganic P was primarily particulate organic P (especially Reservoirs 1, 6, 23 and 24), while some reservoirs had almost equal fractions of dissolved organic, particulate inorganic and particulate organic P (Reservoirs 3, 27 and 32). The particulate organic fraction just covers the independently determined (via cell counts and biomass estimations) phyto- and zooplankton biomass per reservoir, while in three reservoirs, there were 3.3 to 9.7 additional kg P as particulate organic matter (Reservoirs 3, 6 and 23) (Table 6.5). The phytoplankton contribution to the measured TP value varied between 1.9 and 11.9 % and zooplankton contributed from 3.5 to 15.9 %. Macroalgae impact (swimming mats of filamentous green algae) on the P budget was primarily insignificant (0.0 to 3.2 %), while in Reservoirs 6 and 26, they contained 12.1 and 9.1% of total reservoir P, respectively. Similar results were obtained for macrophytes which contributed 7.4, 10.0 and 13.8 % to total reservoir P in Reservoirs 32, 24 and 27. However, in the other reservoirs macrophytes contributed >5 % to total reservoir P. The fish present other than stocked trout (less than 200 kg) contributed less than 3 % to the phosphorus reservoir budget, with Reservoirs 6 and 24 reaching 3.2 and 3.8 % respectively. Common fish species found in the reservoirs were Bluegill sunfish (*Lepomis macrochirus*), Largemouth bass (*Micropterus salmoides*), grass carp (*Ctenopharyngodon idella*), carp (*Cyprinus carpio*) and rainbow trout (*Oncorhynchus mykiss*).

Sediment TP can be an indication of the long-term history of the reservoirs, but is also influenced by reservoir age. The adapted TP content (equalised to 100.000 m^3) varied between 23.6 and 1969.9 kg and reflected a vast difference between reservoir conditions. Reservoirs with the highest sediment TP content were Reservoirs 6, 27 and 5, which can be caused by the long history of introduction of hypolimnetic water from other reservoirs (Reservoirs 5 and 6) and the long history of trout production and recirculation of reservoir water and its usage for cellar cleaning in Reservoir 27. The potential P release relative to the reservoir standing stock was severe in Reservoirs 27, 6 and 24 with 133, 150 and 204 % respectively. The sediment minimum retention varied from 59.7 to 98.7 %, with maximum retention from 95.2 to 99.8 % (Table 6.5). This was assuming that incoming P (gross gain) would eventually sink to the reservoir bottom (e.g. by being assimilated into organic biomass which then sinks to the reservoir bottom).

The phosphorus balance of the ten reservoirs varied from a phosphorus loss of 25.5 kg per year to a phosphorus gain of 90.6 kg per year (Table 6.5), with no direct linkage to the production status. To compare reservoirs of different reservoir volume, the volume (equalised to 100.000 m^3) and the adapted parameters are shown in Table 6.8. The equalised gross gain varied between 7.3 kg P loss and 58.6 kg P per year, where two reservoirs stand out with higher P gains (Reservoir 26 and 27). The net P gain per µg/L reservoir water is linear to the net gain per year (equalised water volume). This is however, a relative value (and therefore also the net gain per µg/L with respect to the actual reservoir volume).

Table 6.8: Phosphorus input, reservoir steady-state and output (in kg) of ten Western Cape reservoirs as a yearly average with equalised reservoir volumes (with a reference volume of 100.000 m^3). The reservoirs were sorted from a negative phosphorus balance to a positive phosphorus balance, with an increasing phosphorus balance from left to right.

P BUDGET	**Reservoir**	**23***	**22***	**5***	**3***	**32***	**1***	**6***	**24***	**27***	**26***
Reservoir descriptors	Surface area (ha)	2.1	1.5	1.5	2.1	2.1	1.5	2.6	1.6	2.0	1.0
	Carlson TSI (TP) (Chapters 3 and 4)	73	72	74	73	67	64	77	65	74	81
	Water exchange rate	6.7	9.2	3.0	0.7	2.1	2.5	2.0	1.6	1.2	0.6
INPUT (kg)	Sum input	118.7	119.0	35.9	4.7	33.5	22.4	43.8	34.7	63.3	76.4
OUTPUT (kg)	Sum output	126.0	122.4	36.0	4.6	29.9	15.9	33.5	22.9	24.7	17.9
BALANCE (kg)	**Gross P gain/year**	**-7.3**	**-3.4**	**-0.1**	**0.1**	**3.5**	**6.5**	**10.3**	**11.8**	**38.6**	**58.6**
RESERVOIR (kg)	**SUM Reservoir**	**12.0**	**11.7**	**13.9**	**12.4**	**9.2**	**6.5**	**17.8**	**8.1**	**15.6**	**21.3**

* volume dependent variables adapted to a theoretical volume of 100.000 m^3

The reservoir turnover (mean of P input and output) was compared to the mean reservoir P standing stock (inorganic P, bacteria, phytoplankton, zooplankton) with most ratios varying between 2.2 and 3.7. Two reservoirs (22 and 23) had a turnover rate of more than 10 times the standing stock (except sediment). Reservoir 3 had a P turnover of less than half the reservoir steady-state, with a neutral yearly P balance and generally low water P content.

The dissolved organic P fraction was significantly lower in reservoirs without fish production (mean of 1.70 kg) than in production sites (3.36 kg) and could represent the pelagic bacterial P content (and therefore biomass) of the reservoirs. Dissolved organic P in kg was highest in Reservoir 26 and 27, the reservoirs of highest eutrophication status (Chapter 4) and largest net gain per year where the net feed inputs were highest.

The environmental carrying capacity of the reservoirs according to an estimation based on the phosphorus budget results, showed that three of the six studied production reservoirs were not suitable for the production of 5 t of trout without adding to the phosphorus balance (Table 6.9).

Table 6.9: Estimated carrying capacity of the reservoirs according to the net gain per reservoir derived from the annual phosphorus budget. The additional P input with a 10 % increase of the steady state was calculated as 10 % of the standing stock times the retention coefficient (= turnover relative to sum reservoir in Table 6.5).

CARRYING CAPACITY (P budget)	**Reservoir**	**23**	**22**	**32**	**24**	**27**	**26**
	Annual P balance "as was in 2006"	-25.5	-14.1	11.5	54.2	80.6	90.6
	Load of 5 t trout unit advisable	yes	yes	no	no	no	no
Recommended load (no P increase in reservoir)	**Annual input by aquaculture (kg)**	36.2	66.9	39.5	47.3	76.3	103.7
	Theoretical annual P balance without aquaculture	-61.7	-81.4	-28.0	6.9	4.3	-13.1
	Allowable waste introduction to keep neutral balance	61.7	81.4	28.0	0.0	0.0	13.1
	Waste introduction per t of fish with FCR 1.1 and feed P 1.4 % (according to Beveridge 2004)	8.2	8.2	8.2	8.2	8.2	8.2
	Waste introduction per t of fish with FCR 1.1 and feed P 1.0 % (according to Beveridge 2004)	6.2	6.2	6.2	6.2	6.2	6.2
	Maximum carrying capacity without P gain (FCR 1.1, Fish feed P 1.4 %)	7.5	9.9	3.4	0.0	0.0	1.6
	Maximum carrying capacity without P gain (FCR 1.1, Fish feed P 1.0 %)	10.0	13.1	4.5	0.0	0.0	2.1
Maximum load (10 % P increase)	**Additional P input in kg with consequenting 10 % steady-state increase**	43.3	52.0	10.8	13.7	9.9	8.3
	Maximum carrying capacity with 10 % P gain (FCR 1.1, Fish feed P 1.4 %)	12.8	16.3	4.7	0.0	0.0	0.6
	Maximum carrying capacity with 10 % P gain (FCR 1.1, Fish feed P 1.0 %)	16.9	21.5	6.3	0.0	0.0	0.8

Four reservoirs underwent a phosphorus gain even without any aquaculture impact. According to the annual nutrient budget model, Reservoirs 22, 23 and 33 can support 9.9, 7.5 and 4.7 t of trout, respectively, without increasing the phosphorus load of the reservoir (sustainable production)(Table 6.10).

6.3.3 Carrying capacity according to the Beveridge model

The trout carrying capacity of the six production reservoirs was estimated with support of the Beveridge model (Beveridge 1984). The final outcome indicated that two reservoirs with relatively high water exchange volumes per year, can support trout production in the range of 5 to 6 t per winter season (Table 6.10). With a production restricted to 5.5 t, the phosphorus concentration of Reservoir 22 would presumably increase by 10 % after one year, while in Reservoir 23, the production load would achieve a 10 % increase after 1 to 2 years (6.7 t production and 9.4 t production respectively). The other reservoirs were only able to support production loads between 300 kg and 2.1 t according to the model.

It was also calculated how production staying within the limits of a 10 % P increase could be prolonged with improved FCR and feed formulation managing on 1 % P content without reducing its efficiency. Four reservoirs were still not found suitable to support the production of five tons of trout (not even for one season), while the other two reservoirs could produce sustainably.

Table 6.10: Outcome of the Beveridge model sorted according to carrying capacity load.

CARRYING CAPACITY (Beveridge)	**Reservoir**	**23**	**22**	**32**	**24**	**27**	**26**
	Average yearly TP (mg/L)	118	114	83	67	131	183
	Targeted P concentration (max. 10 % increase)	130	125	91	74	144	201
	Surface area (ha)	7.3	6.5	6.9	7.2	4.2	1.6
	Water exchange rate (times/y)	6.7	9.2	2.1	1.6	1.9	0.6
	FCR in 2006	1.2	2.0	1.3	1.4	2.2	2.9
	Carrying capacity with given WQ conditions, achieved FCR and feed with 1.4 % P (in t)	9.5	6.7	2.1	1.9	1.2	0.3
	Optimal production capacity (max. 10 % increase) with given WQ conditions, FCR 1.1 and feed with 1.4 % P (in t)	10.8	15.5	2.7	2.8	2.6	1.2
	Optimal production capacity (max. 10 % P increase) with given WQ conditions, FCR 1.1 and feed with 1.0 % P (in t)	14.6	21.0	3.7	3.8	3.6	1.7

6.4 Discussion

Generally, the main period of water input in reservoirs under Western Cape climatic conditions were the winter months. Seventy to 90 % of annual rainfall in the area fell between April and October (SAWS 2007). Rainwater collected and fed the reservoirs directly, as small runoff streams or via the primarily non-perennial rivers. Inflowing structures varied from spontaneous rivulets via concrete or natural surface inflows to pipe systems. Most water was derived from rainwater, that was to various degrees influenced by the landscape it travelled through, while some reservoirs were filled with water from other reservoirs.

Surplus inflow water left the reservoir during the winter season via a constructed or natural overflow or a submersed pipe. In- and outflow were primarily located at opposite ends, with one exception where the outflowing water quality was more similar to the inflowing water quality, than the reservoir average. Evaporation rates were expected to be higher during the summer months and the residual water that left the reservoirs was also extracted during the summer months and used for irrigation or wine cellar activities (e.g. cleaning, rinsing). Due to stagnation of the water body with the resultant accumulation of phosphorus towards the reservoir bottom, the depth of the outflowing structure (epi- or hypolimnion) plays an important role in the overall phosphorus budget.

While the annual budget model used the sum of internal and external flows (input-output) to estimate the net balance with negligence of internal processes (black box), the Beveridge model represented an input-output model using a steady state approach by respecting only additional input (aquaculture) as well as internal and external sinks.

The annual budget model is not dependent on any assumptions or simplifications, but the seasonal division of major flows was advantageous. Incoming and through-flowing water prevails in winter, while outgoing nutrients leave in summer when epilimnetic and hypolimnetic nutrient concentrations can be determined. The advantage with the Beveridge model is that less data input is necessary. The basic assumptions of the Beveridge model (Vollenweider 1968, Dillon & Rigler 1975) are primarily fulfilled with a warm monomictic lake during the winter period, which is when more than 90 % of phosphorus enters the reservoirs and when cage production takes place. In more detail, the complete mixture of reservoir water occurred during production and water through-flow was concurrent with the main production, while the exact proportion lost to sedimentation is still not known. The sink for aquaculture waste in the current example were overflows and the sediment, with a proportion also adding to the phytoplankton and zooplankton biomass during winter.

The actual sediment release rate was not quantified but should be somewhere between the release rate of water soluble phosphorus and the potential release rate of so-called algal available phosphorus (dependent on the sediment chemistry as well as hypolimnetic water quality conditions, such as pH and oxygen concentration). Macrozoobenthos biomass (and phosphorus content) could also not be determined which leads to an underestimation of the semi-permanently bound P biomass (reservoir standing stock).

Hakanson et al. (1998) reported overestimations of the Beveridge model primarily due to the negligence of invertebrate and wild fish feeding on particulate waste from farm reservoirs. In the current study, macroinvertebrate and wild fish populations were considered to have an insignificant impact relative to other sources. Johansson and Nordvarg (2002) and Nordvarg (2001) claimed that the sedimentation rate in the model was underestimated and so suggested loading capacities by a factor of two.

The outcome of the two models with different approaches was reasonably similar (10 to 20 % divergence). In two reservoirs, 5 t production units of trout (7.5 and 9.9 t) are environmentally sustainable with the commonly used fish feed, and practically FCR values of 1.2 and 2.0 were achieved. A third reservoir, 32, seems reasonably suitable for trout production with an estimated 4.7 t carrying capacity. The remaining three production sites are not suitable for sustainable trout production because the reservoir systems as managed cannot assimilate the additional phosphorus load, even if the model overestimated by a factor of 2 (Nordvarg 2001).

There are, however, options to improve the phosphorus budgets of Western Cape reservoirs with and without aquaculture production. The input load of phosphorus via inflowing water can be positively influenced. Water flow through reservoirs is advantageous for water quality in general and should be considered with reservoir construction (e.g. natural structures that reduce input of solids). In some cases, the phosphorus content of the reservoirs can be influenced by water extraction management (e.g. extraction from hypolimnetic water, however while ecologically useful especially with increased eutrophication status, he enriched bottom water seemed to block irrigation water filters).

The main P input into the reservoirs enters via inflowing water from rivers, spontaneous runoff rivulets, farm reservoirs or large reservoirs (67.5 to 87.9 %, aquaculture as influence excluded). This was similar to results by Miranda and Matvienko (2003) who state that 82 % of the phosphorus input in Brazilian reservoirs is derived from rivers.

Runoff-fed river water and diffuse runoff are influenced by land uses in the catchment. In the Western Cape region this includes natural fynbos vegetation, agriculture, forestry and human settlements. The measured river phosphorus concentrations ranged from 57 (Swart River) to 158 (Du Toits) µg/L (mg/m^3) which defines them as eutrophic (47-130 µg/L annual mean TP) and hypertrophic (>130 µg/L) sources according to Volume 7 of the South African Water Quality

Guidelines (DWAF 1996). Using the same classification, De Villiers (2007) found eutrophic conditions with episodic hypertrophic conditions in the middle reaches of the Berg River, one of the two major tributaries in the study area. When water from other farm reservoirs was the dominating source, the phosphorus input increased from 115 to 311 µg/L inflowing water (due to the accumulation of nutrients in standing water bodies). The phosphorus loads entering the water bodies in this study ranged between 0.2 and 8.4 g $P/m^2/y$. Typical loads received by Hartbeespoort Dam in the North-West Province amounted to 15 to 25 g $P/m^2/y$, were enriched by wastewater treatment works, and described as high phosphorus loads (Thornton 1989). Therefore, on a general note, more strict regulations on phosphorus input minimisation from diverse sources would greatly support the water quality of rivers and consequently lowland water bodies and their trophic status.

The gross input of aquaculture waste ranges from an additional 80 to 563 µg/L per year (Table 6.4) and exceeds agricultural input which was estimated as net input of 2 to 10 µg/L per year (Chapter 4). Assuming that 95 % of the gross phosphorus input will be assimilated by the systems, the input by net-cage aquaculture in small farm reservoirs ranges from a net input of 5 to 30 µg/L per year, equalling and exceeding agricultural input in the same systems.

The input of aquaculture waste can be reduced via good feed management and feed quality. The achievement of FCRs of <1.2 and trout feeds with phosphorus content <1.1 % have become regulatory standards in Norway, the USA and Canada (Bergheim & Cripps 1998, MacMillan et al. 2003). The average FCR achieved was 1.8 (6 production sites) which can be improved upon. Research with feeding optimisation is already a component of the "Hands-On" project, and deserves further close attention. The net waste load by aquaculture in relation to expected total phosphorus input was calculated and was <13 % at the two suitable sites, but >34 %, 45, 62 and 90 % at the other sites.

Since the net waste per unit produced trout (e.g. one ton) is relatively stable with good management practices, this shows the advantage of larger reservoirs with sufficient volume in their capacity to dilute the absolute amount of phosphorus introduced. Hence, reservoirs of sufficient total volume need to be selected for production.

Another important factor for successful trout production is sufficient water exchange as various examples show (Edwards 1987, Heath 1992, Sedgwick 1995, Beveridge & Steward 1997, Ingram et al. 2000, Truscott 2008). With trout production good initial water quality and high water replacement were seen to be the most important criteria with cage production in the UK (North et al. 2006) while stocking densities did not have a great impact and varied between 20 and 80 kg/m^3 (max. 8 kg/m^3 in the Western Cape). Since higher stocking densities than practised in the Western Cape do not seem to be disadvantageous to fish health, more confined fish in less space could however be advantageous in terms of feeding efficiency and FCR.

The location of the irrigation water outlet (epi- or hypolimnion) determines if less or more phosphorus than the reservoir average will be extracted during the summer period (reservoir stagnation). Most reservoirs with water of a lower water quality need to employ surface water extraction, while reservoirs with good general water quality practise hypolimnetic water extraction for irrigation purposes with the most severe impact being filter blocking (personal observation). With the commencement of cage aquaculture, two reservoirs with previous hypolimnetic water withdrawal had to shift to surface water extraction. However, hypolimnetic water extraction can decrease the phosphorus load of a reservoir and influence its yearly phosphorus budget quite significantly (Cooke 1993, Nürnberg 1997) and would therefore be a suitable method to reduce the phosphorus concentration in the water bodies at most current production sites. When hypolimnetic withdrawal is accessible and sufficient inflowing water available, additional hypolimnetic withdrawal will be an effective option for nutrient removal (but should not cause an impact in receiving waters).

6.5 Conclusions

The aim of this chapter was to verify whether sustainable trout production with units of 5 t of trout (smallest economically feasible production unit, Pers. Com., Stander, H. 2007) change the phosphorus budget of small water bodies in the Western Cape. The results show that two to three of the six studied reservoirs are able to support the 5 t unit where two reservoirs share the feature of a water exchange rate of >5 times per year and have a negative phosphorus balance (more phosphorus leaves than enters the reservoir), while a third reservoir has a total depth of 20 m and a water exchange rate of at least 2 times per year.

In many reservoirs, 10 to 15 % additional introduction of phosphorus per year is already exceeded. Aquaculture matches and exceeds the already incoming phosphorus load in half the production sites (especially in reservoirs with low water exchange rates and low volumes). Hence, many reservoirs in the Western Cape are not suited for net-cage production under the current conditions when considering that agriculture is already pressuring the systems and additional load in a similar quantity can not be supported by the reservoirs without severe impact on the eutrophication status.

In examples with a negative phosphorus balance and in reservoirs with neutral to slightly positive balance, the systems can support net-cage production. Improvements on waste-catching/management techniques, feeding management and measures to support the hydrodynamics of the reservoirs would increase the number of suitable reservoirs.

Generally, data on inflowing water quantities and quality showed that eutrophic conditions of the reservoirs are the inevitable consequence. The winter turnover rates of phosphorus through the water bodies, exceeded the phosphorus mass in the water by a factor of two or more in all but one case (only reservoir with a water exchange rate <1 and a 50 % rainwater input volume). Nutrient turnover was high and influenced by different processes depending on the respective reservoir. Some were bound to internal loading (ratio of the water soluble phosphorus mass in the sediment to the phosphorus mass is >1.2 (Reservoirs 6, 27 and 24), not <0.3 as in most reservoirs). Some reservoirs profited from hypolimnetic water extraction for irrigation purposes (Reservoir 22), others had very high water exchange rates (Reservoirs 22 and 23) and some were incapable of assimilating the additional input by aquaculture, despite a naturally neutral balance (Reservoir 26 and 32). In conclusion, the input by inflowing water is extremely high, especially if less enriched water is extracted than flows in (e.g. with surface water extraction for irrigation purposes as only outflowing source). However, it has been shown that the input by aquaculture equals in 50 % of reservoirs with aquaculture all other incoming sources or exceeds them. In these reservoirs, aquaculture added 100 % or more P in addition to the P already introduced by runoff and inflowing rivers (which includes the influence of agriculture).

The budget approach compares in- and output information and shows the difference between both in the nutrient balance (fed by external or internal loading). The Beveridge model (Beveridge 1984) assumes a steady state of phosphorus and uses the water exchange rate and a net sedimentation rate estimate (adapted by current data in this study) to calculate the removal and sediment immobilisation of additionally introduced P. It is suggested that the Beveridge model can be successfully applied in Western Cape reservoirs with primary water and nutrient exchange during the winter period, which also coincides with the production period. The stagnation during the summer period when different phosphorus levels emerge in different water layers does not affect the outcome of the capacity results, especially when the model is fed with winter average phosphorus concentrations. The concentration at turnover in autumn would be the determinant for an allowable load. In contrast to the budget model, less data input was necessary to feed the model. However, in contrast to the budget model, an increase in average surface phosphorus was part of the model's assumptions and an upper limit needs to be set (e.g. 10 % increase).

The parameters that influence the outcome the strongest were water exchange rate, the FCR and the phosphorus content of the feed. Water exchange rates >3 times per year, ideally >5 times per year (phosphorus turnover was >10 at the two suitable sites), greatly increase the carrying capacity. With optimum conditions, the carrying capacity can be further increased, however it is still recommended that four of six reservoirs maintain a <5 ton production (which could be economically unfeasible).

As an option to decrease the phosphorus load, water can be extracted from the hypolimnion in summer and autumn and used for irrigation or disposed of. However, in eutrophied reservoirs it can often not be used for drip irrigation due to emerging filter problems when using eutrophied hypolimnetic water usually enriched with organic debris (e.g. phytoplankton).

6.6 References

AquaNutro (Pty) Ltd (2005). Trout grower - nutritional information AN-TGRO 02 H. AquaNutro (Pty) Ltd. Malmesbury, South Africa.

Bergheim, A. and Cripps, J. S. (1998). Effluent management: overview of the European experience. Rogaland Research Publication No. 1998/083, 233-238. Norway.

Beveridge, M. C. M. (1984). Cage and pen fish farming Carrying capacity models and environmental impact. FAO Fisheries Technical Paper (255). Rome, FAO.

Beveridge, M. C. M. and Steward, J. A. (1997). Cage Culture: Limitations in Lakes and Reservoirs. FAO Fisheries Technial Paper 374. 1997. Dhaka, Bangladesh, FAO.

Beveridge, M. C. M. (2004). Cage aquaculture. Oxford: Fishing News Books.

Boyd, C. E. and Tucker (1998). Pond aquaculture and water quality management. New York: Springer US.

Cooke, D. G. (1993). Restoration and Management of Lakes and Reservoirs. Chelsea, Michigan: Lewis Publishers.

Cromey, C. J., Nickell, T. D., and Black, K. D. (2002). DEPOMOD - modelling the deposition and biological effects of waste solids from marine cage farms. Aquaculture 214(211): 239.

De Villiers, S. (2007). The deteriorating nutrient status of the Berg River, South Africa. Water SA 33(5): 659-664.

Dillon, P. J. and Rigler, F. H. (1974). A Test of a Simple Nutrient Budget Model Predicting the Phosphorus Concentration in Lake Water. Journal of the Fisheries Research Board of Canada 31: 1771-1778.

Dillon, P. J. and Rigler, F. H. (1975). A Simple Method for Predicting the Capacity of a Lake for Development Based on Lake Trophic Status. Journal of the Fisheries Research Board of Canada 32(9): 1519-1531.

DWAF (1996). South African Water Quality Guidelines. Volume 7: Aquatic Ecosystems. Pretoria.

Edwards, D. (1987). Freshwater Fish Culture in Greece. FI:DP/GRE/85/002. Rome, FAO.

FAO (1995). Code of Conduct for Responsible Fisheries. FAO Conference. Rome, FAO.

Hakanson, L., Carlsson, L., and Johansson, T. (1998). A new approach to calculate the phosphorus load to lakes from fish farm emissions. Aquaculture Engineering 17(149): 166.

Hamblin, P. F. and Gale, P. (2002). Water Quality Modeling of Caged Aquaculture Impacts in Lake Wolsey, North Channel of Lake Huron. Journal of Great Lakes Research 28(1): 32-43.

Heath, R. G. M. (1992). The effects of cage culture of rainbow trout in South Africa on the water quality of three shallow impoundments with long retention times. In: Gall, G. A. E. The rainbow trout: Proceedings of the 1st Aquaculture Symposium held at the Institute of Aquaculture. Stirling, University of Stirling, Institute of Aquaculture.

Ingram, B. A., Gooley, G. J., McKinnon, L. J., and de Silva, S. S. (2000). Aquaculture-agriculture systems integration: an Australian prospective. Fisheries Management and Ecology 7: 33-43.

Johansson, T. and Nordvarg, L. (2002). Empirical mass balance models calibrated for freshwater fish farm emissions. Aquaculture 212(1-4): 191-211.

Lewis, D. M., Elliott, J. A., Lambert, M. F., and Reynolds, C. S. (2002). The simulation of an Australian reservoir using a phytoplankton community model: PROTECH. Ecological Modelling 150: 107-116.

MacMillan, J. R., Huddleston, T., Woolley, M., and Fothergill, K. (2003). Best management practice development to minimize environmental impact from large flow-through trout farms. Aquaculture 226: 91-99.

Maleri, M., Du Plessis, D., and Salie, K. (2008). Assessment of the interaction between cage aquaculture and water quality in irrigation storage dams and canal systems. WRC Report 1461/1/08. Department of Water Affairs and Forestry. Pretoria, South Africa.

Marion, L., Clergeau, P., Brient, L., and Bertru, G. (1994). The importance of avian-contributed nitrogen (N) and phosphorus (P) to Lake Grand-Lieu, France. Hydrobiologia 279-280(1): 133-147.

Miranda, S. A. and Matvienko, B. (2003). Rain and groundwater as phosphorus sources of a small reservoir. Lakes & Reservoirs: Research & Management 8(1): 27-30.

Moreno-Ostos, E., Elliott, J. A., Cruz-Pizarro, L., Escot, C., and Basanta, A. (2007). Using a numerical model (PROTECH) to examine the impact of water transfers on phytoplankton dynamics in a Mediterranean reservoir. Limnetica 26(1): 1-11.

Mueller, D. K. (1982). Mass balance model estimation of phosphorus concentrations in reservoirs. Journal of the American Water Resources Association 18(3): 377-382.

Nordvarg, L. C. Y. (2001). Predictive Models and Eutrophication Effects of Fish Farms. Uppsala, Uppsala University, Department of Earth Sciences.

North, B. P., Ellis, T., Turnbull, J. F., Davis, J., and Bromage, N. R. (2006). Stocking density practices of commercial UK rainbow trout farms. Aquaculture 259: 260-267.

Nürnberg, G. K. (1997). Coping with water quality problems due to hypolimnetic anoxia in Central Ontaria Lakes. Water Quality Research Journal of Canada 32: 391-405.

OECD (1982). Eutrophication of Waters. Monitoring, Assessment and Control. Paris, Organisation for economic co-operation and development.

Palma, R. M., Prause, J., Fontanive, A. V., and Jimenez, M.P. (1998). Litter fall and litter decomposition in a forest of the Parque Chaqueño Argentino. Forest Ecology and Management 106(2-3): 205-210.

Persson, G. (1991). Eutrophication resulting from salmonid fish culture in fresh and salt water: Scandinavian experiences. In: Cowey, C. B. and Cho, C. Y. Proceeding of 1st international symposium on nutritional strategies in the management of aquaculture wastes: Nutritional Strategies and Aquaculture Waste. Guelph, Ontario, University of Guelph.

Piper, G. (2006). Measuring Water Quality. Living Lake Macquarie - a community newsletter October 2006 (Issue 11). Lake Macquarie, Australia, The office of the Lake Macquarie and catchment coordinator.

Redfield, A. C., Ketchum, B. H., and Richard, F. A. (1963). The influence of organisms on the composition of sea water. In: Hill, M.N. The sea. Wiley, New York.

Reynolds, C. S., Irish, A. E., and Elliott, J. A. (2001). The ecological basis for simulation phytoplankton responses to environmental change (PROTECH). Ecological Modelling 140: 271-291.

Rooney, N. (2002). Scale of analysis and the influence of submerged macrophytes on lake processes. Montreal, Quebec, McGill University.

Russell, I. A., Randall, R. M., Randall, B. M., and Hanekom, N. (2009). Relationships between the biomass of waterfowl and submerged macrophytes in a South African estuarine lake system. Ostrich - Journal of African Ornithology 80(1): 35-41.

SAWS (2007). Weather data. Cape Town, South African Weather Service.

Schussler, J., Baker, L. A., and Chester-Jones, H. (2007). Whole-system phosphorus balances as a practical tool for lake management. Ecological Engineering 29(3): 294-304.

Sedgwick, S. D. (1995). Trout farming handbook. Oxford, Fishing News Books (Blackwell Science Ltd).

Thornton, J. A. (1989). Aspects of the phosphorus cycle in Hartebeespoort Dam (South Africa). Hydrobiologia 183: 87-95.

Toor, G. S., Condron, L. M., Di, H. J., and Cameron, K. C. (2004). Seasonal Fluctuations in Phosphorus Loss by Leaching from a Grassland Soil. Soil Science Society of America Journal 68: 1429-1436.

Truscott, T. (2008). Department of Primary Industries of the State of Victoria - Fisheries Regulations. Melbourne, Australia, Department of Primary Industries.

Uye, S.-I. and Matsuda, O. (1988). Phosphorus Content of Zooplankton from the Inland Sea of Japan. Journal of the Oceanographical Society of Japan 44: 280-286.

Vardavas, I. M. and Fountoulakis, A. (1996). Estimation of lake evaporation from standard meterological measurements: application to four Australian lakes in different climatic regions. Ecological Modelling 84(1-3): 139-150.

Vollenweider, R. A. (1968). Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. OECD Technical Report DA5/SU/68-27. Paris, Organisation for economic co-ordination and development.

Vollenweider, R. A. (1975). Input-Output Models. With Special Reference to the Phosphorus Loading Concept in Limnology. Schweizerische Zeitschrift für Hydrologie 37: 53-84.

Wetzel, R. G. and Likens, G. E. (2000). Limnological Analyses. New York: Springer.

CHAPTER 7 CONDITIONS CAUSING TAINTED RAINBOW TROUT (*ONCORHYNCHUS MYKISS*) FLESH IN WARM MONOMICTIC RESERVOIRS

Abstract

The off-odour and muddy taste in fish flesh (also called taint) is a worldwide problem which reduces the market value of the product, and this also holds true for net cage production undertaken in South African farm reservoirs (Western Cape). With a total of seventeen small reservoirs (one to eight ha) that held net cages for trout production (5 t units), fish flesh of six reservoirs tested positive for algal taint (when using organoleptic methods).

The most commonly known substances released by cyanobacteria and linked to taint in rainbow trout flesh are 2-methylisoborneol (MIB) and geosmin (short for 1,2,7,7-tetramethyl-2-norborneol). The presence of the two compounds was not verified, but cyanobacteria species known to release MIB and geosmin were found when harvested fish tested positive for off-odour (*Microcystis robusta*, *Aphanothece* sp. and *Chroococcus* sp.).

Cyanobacteria of sufficient biomass to cause algal taint occurred in each reservoir, however species and seasonal distribution differed. Correlation analyses of factors known to influence cyanobacterial abundance compared to characteristics of reservoirs with and without taint respectively, allowed conclusions on risk minimising water quality criteria. These factors included physico-chemical water quality parameters, nutrient concentrations (nitrogen and phosphorus), morphometric parameters (surface area, rock type of catchment) and production parameters (production intensity, the date of introduction and harvest of fish), and cyanophyte presense and biomass.

After analysing reservoirs with algal taint and without, including their characteristics (geomorphology, water quality, production history), the following criteria were suggested to minimise the risk of algal taint occurrence: deep reservoirs with a longer period of 15 to 17 °C in the hypolimnion in spring, granite dominated catchment areas and basins, sites of low total phosphorus concentrations in spring (<60 µg/L), TN:TP ratio >7, no history of algal taint in produced fish, a diverse cyanophyte community (5 to 9 species with biomasses >0.1 mg/L within a year) and dinophyte dominance.

7.1 Introduction

Cage aquaculture of rainbow trout in Western Cape irrigation reservoirs (farm dams) has the potential to add to the usage of these water resources and to raise additional incomes for farmers and farm workers (Rouhani & Britz 2004). The quality of fish produced determines the market value and eventual financial success. Water quality factors observed to affect fish quality in the Western Cape are conditions such as low oxygen (reduced growth, increased mortality) and the presence of ammonia and off-flavours (Maleri 2009). While unfavourable water conditions affect the risk for diseases and increase mortality rates, off-flavours will not affect fish health, but rather palatability and marketability.

Two compounds known to cause a characteristic muddy-earth smell and taste are 2-methylisoborneol (MIB) and geosmin (1,2,7,7-tetramethyl-2-norborneol) (Selli et al. 2006, Robertson et al. 2006). Other than an undesired taste in drinking water and water farmed products, these substances cause no ill effects (Robertson et al. 2006). Cyanophytes are the main algal group associated with an earthy-musty taint. Certain species release geosmin (e.g. *Anabaena* sp.) while others release MIB (e.g. *Oscillatoria* sp.). Another systematic group that has been shown to produce MIB and geosmin are actinobacteria (Zaitlin & Watson 2006). Actinobacteria have a lower abundance in winter, but occur throughout the year either in sediments or floating (Klausen et al. 2005). This chapter concentrates on cyanophytes and whether they could be responsible for algal taint in Western Cape irrigation reservoirs.

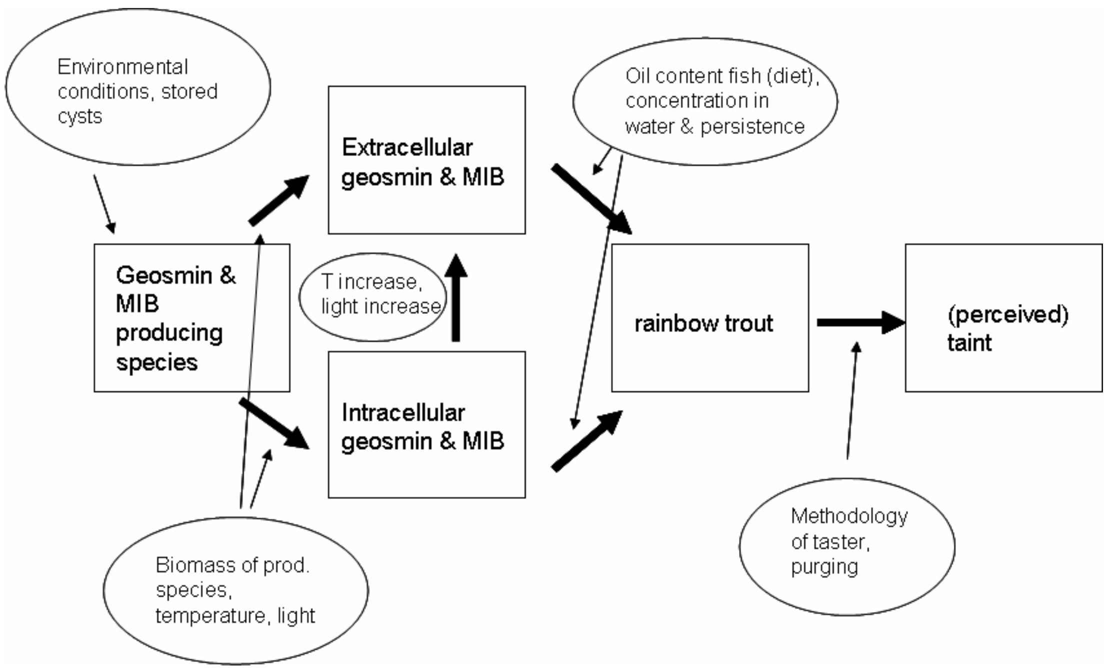

Figure 7.1: Pathway of accumulation of algal taint in rainbow trout and its detection (rectangles) with the major influencing factors (ovals).

The pathway of mechanisms and factors deciding the actual concentration of taint is shown in Figure 7.1. Blue-green algae prefer total surface phosphorus concentrations >25 µg/L and warm water conditions of 15 to 25 ℃ (Morscheid et al. 2006, Warrington 2001, Branco et al. 2001, Robarts & Zohary 1987). The life cycle of most cyanobacteria species goes through a sedimentation stage in autumn, a dormant or overwintering stage in the sediments at temperatures <15 ℃ (Fay 1988), with recruitment in spring. With cyanophyte recruitment, the driving factors are temperature, resuspension and bioturbation, with other factors such as light, nutrients and anoxic conditions also playing a role (Tan et al. 2008). Temperatures >30 ℃ and intensive UV radiation showed negative effects on cyanophyte biomass (Zhang et al. 2009). Tung (2006) found that 300 ng MIB-2 were produced per µg chlorophyll a in an *Oscillatoria* species in laboratory experiments.

The actual MIB concentration pertained (intracellular concentration) and released (extracellular concentration) by cyanophytes depends on temperature and light as well, where with increasing temperatures and light conditions the extracellular proportion rises (Zhang 2009). When cells die, all their intracellular geosmin and MIB will be released. Therefore, the concentration of geosmin and biomass in water will be slightly delayed when compared to the actual peak biomass of the geosmin or MIB producing species (Figure 7.2). It takes one month to eradicate approximately 30 ng MIB per litre source water (Westerhoff et al. 2005). Dependent upon the maximum MIB and geosmin concentration, it can take several months to have a water body without tainting substances.

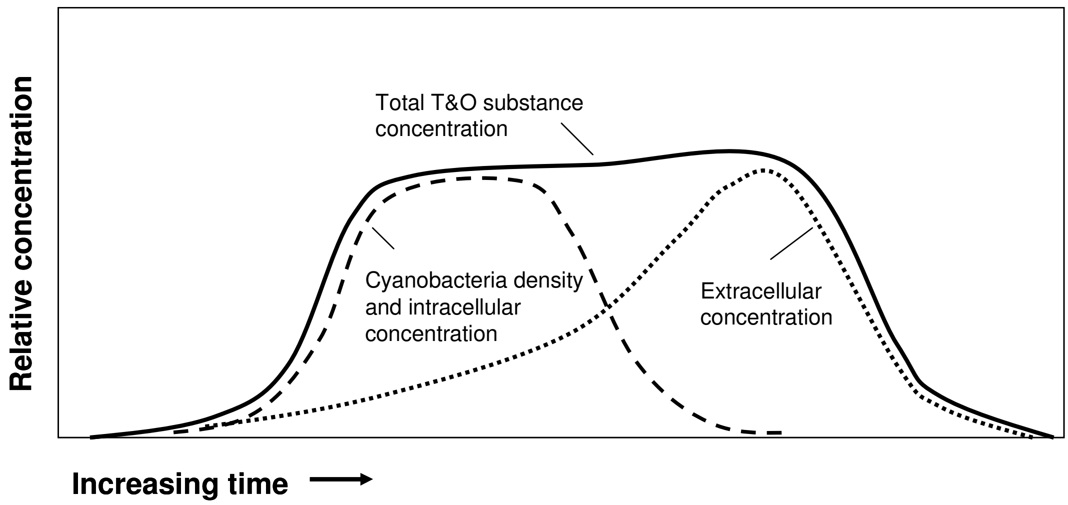

Figure 7.2: Theoretical temporal distribution of total, intracellular (particulate) and extracellular (dissolved) cyanobacterial geosmin and MIB concentrations dependent on cyanobacterial biomass (modified from Graham et al. 2008).

The concentration of geosmin and MIB accumulated in fish flesh varies with the percentage of oil (fatty tissue) in the whole body biomass (Howgate 2004). The standard trout feed in South

Africa contains 16 to 18 % crude fat (AquaNutro 2005). The accumulated whole body fat was shown to be diet independent (approximately 15 %) (Jobling et al. 1998), however, fat in flesh varies between 8.3 and 11 % (Regost et al. 2001), remaining below 10 % with low fat feed (<20 %). Purging can clean fish from algal taint within 48 hours with levels of 3 µg geosmin/kg flesh and lower, when transferred to taint-free water, however this is not always practical. When remaining in the same water with reduced to eventually no taint producers (presumably cyanophytes), the purging process will be much slower and will depend on geosmin and MIB degradation.

Organoleptic assessments by trained taste panels provide the most ready method to detect off-odour in fish flesh. Human noses can detect the characteristic musty-earthy odour of geosmin and MIB from 15 or 35 ng/L in water respectively (Howgate 2004), while gas chromatography can detect geosmin concentrations from 1 ng/L (Durrer et al. 1999), but this methodology is very costly. In this study, fish flesh was tested and not the concentration in the actual water body. The human detection limit would be approximately 0.2 µg geosmin/kg trout flesh (Robin et al. 2006) and is perceived as a very light taint while from 3 µg geosmin/kg flesh taint is noticeably perceived. With MIB, 0.5 µg/kg flesh forms the absolute human detection limit (corresponds to approximately 35 ng/L in water) (Robertson et al. 2005). According to Howgate (2004), however, the expected detection threshold concentration in flesh would vary between 4.5 and 5.6 µg/kg with geosmin and 5.7 and 7.2 µg/kg with MIB.

Tainted flesh was found in trout reared in Western Cape irrigation reservoirs. A combination of estimated purging period and the estimated quantity of geosmin and MIB production by cyanophytes helped to assess the role of cyanophytes in taste and odour problems in Western Cape production sites (in the absence of geosmin and MIB analyses). Within Western Cape reservoirs where fish are harvested from mid October to mid November, cyanophytes have to be present at sufficient concentrations to accumulate and cause algal taint around harvest time or in higher concentrations prior of harvest (dependent on the maximum biomass of the last cyanophyte occurrence).

This chapter investigated whether cyanophytes were responsible for taint problems and described water conditions and reservoir characteristics that could potentially favour cyanobacterial presence and seasonality in these reservoirs. Suggestions on risk assessment for algal taint likeliness are provided.

7.2 Methods

Seventeen reservoirs with 5 t units of rainbow trout were monitored from August 2005 to October 2006. Due to trout fingerling supply problems only three sites were stocked with trout in the 2007 production reservoirs. Monthly phytoplankton data were available for 11 sites throughout the study period and another six sites were sampled during winter only (April to November 2006 and 2007).

7.2.1 Sites

All 17 sites lie within the south-western area of the Western Cape Province, South Africa. The sites extend in the area of 32°59'59''S to 34°17'35''S and 18°43'22''E to 19°39'46''E and cover a rectangle of 150 by 100 km. The sites range from 1.2 to 7.8 ha in surface area and occur at altitudes from 132 to 530 m above sea level. The water exchange rate of most reservoirs was less than once per year.

Cool, rainy winters and hot, dry summers characterise the region, with the temperature conditions and changes between seasons becoming more extreme further from the coast. Total yearly rainfall also declines the further inland one goes.

7.2.2 Phytoplankton samples

Unfiltered water samples were collected monthly at depths of 0 m, 2 m, 6 m and at the bottom at 11 sites. In the remaining six sites samples were collected at a depth of 2 m and at the bottom. Lugol's iodine solution was immediately added as a preservation method. Species identification and enumeration was possible using Utermöhl counting chambers (Utermöhl 1958) and various literature sources such as Entwisle (1997), van den Hoek et al. (1995), Yunfang (1995), Häusler (1982), Prescott (1978) and Huber-Pestalozzi (1938). The biovolume of each specimen was taken from the literature or calculated via the nearest geometrical shape. Biomass was calculated from the volume data using factors of 1.02 to 1.30 kg/m^3 (Sommer 1996). Wet weight values are presented throughout the paper.

7.2.3 Algal taint monitoring

For most small trout producers in the Western Cape, the processing industry serves as a quality controlling organ concerning off-flavour findings. A taste panel using organoleptic methods differentiated their different taste perceptions into three categories (no, light or strong taint).

A conversion of the panel assessment into a minimum cyanobacterial biomass that would have been necessary to cause the taint, is presented in Tables 7.1 and 7.2. The calculated biomass of geosmin or MIB producing algae is rather an underestimation of minimum occurrence, since geosmin is distributed into extracellular and intracellular components and not all geosmin contained by the algae is immediately released. The geosmin production rates of Jüttner and Watson (2007) were utilised (140-500 ng/mg DW). The dry weight (dw) : wet weight (ww) conversion ratio of 1:5 was applied (Strickland 2009). For MIB, production rates of 3000 ng/mg ww (300 ng/µg chl a) were reported by Tung (2006). A conversion of mg chl a:mg ww of 1:335 was applied (Clesceri et al. 1999).

Table 7.1: Estimated conversion of musty-earthy taint perceived by the taste panel into minimum algal biomass (MIB producing species) needed to release tainting substances. (1) Robin et al. 2006, (2) Howgate 2004, (3) Robertson et al. 2005.

Taste panel	MIB in flesh	MIB in water	Minimum biomass of MIB producing algae
			at harvest
no taint	< 0.5 µg/kg[1]	<35 ng/L[2]	<0.04 mg/L
light taint	>0.5 µg/kg	50 ng/L[3]	>0.04-0.10 mg/L
taint	5.7-7.2 µg/kg[2]	70 ng/L[3]	
strong taint	>7.5 µg/kg	>80 ng/L[3]	

Table 7.2: Estimated conversion of musty-earthy taint perceived by the taste panel into minimum algal biomass (geosmin producing species) needed to release tainting substances. (1) Robin et al. 2006, (2) Howgate 2004, (3) Jüttner & Watson 2007, (4) Robertson et al. 2005.

Taste panel	Geosmin in flesh	Geosmin in water	Biomass of geosmin producing algae
			at harvest
no taint	< 0.2 µg/kg[1]	<15 ng/L[2]	<0.15 – 0.5 mg (WW)[3]
light taint	>0.2 µg/kg[1]	25 ng/L[4]	>0.5 mg (WW)
taint	4.5–5.6 µg/kg[2]	50 ng/L[4]	
strong taint	>6 µg/kg[3]	>60 ng/L[4]	

Consequently, all algal occurrences >0.05 or 0.10 mg wet weight respectively, were compared to the occurrence of algal taint in fish at harvest time. However, algal occurrences of >0.4-1.5 mg/L one month prior to harvest or algal occurrences of >0.7-2.5 mg/L two months prior to harvest could also have produced sufficient geosmin levels to cause algal taint in fish, or even lower biomasses in case of MIB producing species.

7.2.4 Factors influencing cyanophyte presence and biomass

Water quality parameters (physico-chemical conditions, nutrient concentrations), morphometric parameters (surface area and maximum depth, rock type of catchment) and production parameters (production intensity, date of harvest) were correlated with cyanophyte presence and biomass (Spearman ranking correlation).

7.2.5 Statistical analyses

Statistical analyses were conducted using the Statistica 7.0 program (StatSoft, Inc.). Correlations were analysed using the Spearman's rank correlation coefficient. Coefficient values were considered statistically significant with $p<0.05$ and highly significant with $p<0.01$.

7.3. Results

7.3.1 Odour and taint occurrences in the Western Cape

Within the Hands-On Fish Farmer's Co-operative Limited, trout were reared in 21 irrigation reservoirs in 2006 (not all were monitored), while in 2008, 23 irrigation reservoirs were stocked with rainbow trout (winter-grow-out). In 2006, fish flesh from six reservoirs (29 %) was discovered to be tainted by off-flavours and in 2008 an additional six reservoirs (26 %) were discovered to contain tainting components, one of them with a strong taint. Of the six reservoirs in 2006, two projects were abandoned in 2008 and four again had tainted fish at harvest time in 2008 (Table 7.3). Two of the projects with tainted fish in 2008 were newly stocked reservoirs. In 2007, only five reservoirs could be stocked with fish in total and three of them developed taint problems. Other trout producers in the Western Cape also regularly reported cases of tainted reservoirs and fish flesh (Stubbs 2007). The data in Table 7.3 allow the conclusion that reservoirs with algal taint in one year, are more likely to develop algal taint in another harvest season as well (4 of 5 reservoirs), while reservoirs without algal taint problems stayed free of fish flesh tainting compounds in other seasons (8 of 8 reservoirs). There is, however, no guarantee that algal taint could develop in future production years.

Table 7.3: Taint occurrence of fish at harvest time (Early October to mid November) ("-"=no taint, "+"=light taint, "++"= strong taint, ns=not stocked). Reservoirs sorted according to taint occurrence patterns.

Reservoir	Taint in 2004	Taint in 2005	Taint in 2006	Taint in 2007	Taint in 2008
23	ns	-	-	ns	-
24	ns	-	-	ns	-
25	ns	-	-	ns	-
26	ns	-	-	ns	-
27	-	-	-	ns	-
28	ns	ns	-	ns	-
31	-	-	-	ns	-
32	ns	ns	-	+	-
29	ns	+	+	ns	+
30	ns	+	+	ns	+
39	ns	ns	+	ns	+
42	ns	ns	+	ns	+
21	ns	+	++	ns	ns
22	+	ns	+	ns	ns
40	ns	ns	ns	-	++
46	ns	ns	-	+	ns
15	ns	ns	ns	ns	+

In addition to musty-earthy (probably geosmin and MIB related) taint, other taint and odour problems arose at both drinking water reservoirs included in the study. In one case, there was a fishy taint, while the other experienced a foul egg smell. The foul egg smell was detected in October 2006 and derived from elevated hydrogen sulfide concentrations at the reservoir bottom. The drinking water was extracted from the hypolimnion and oxygenation of sulfide to the odourless sulfate was not achieved from water source to the consumer. Early stratification with oxygen depletion caused the problem (0.8 mg/L oxygen end of October 2006). Aquaculture production might have aggravated the early deoxygenation in the reservoir, while later in the season the reservoir level was usually low enough to allow intermediate mixing, therefore destratification and oxygenation of the water. The source of the fishy smell was not clearly established (could have been the occurrence of *Oscillatoria limnetica* in the hypolimnion or intermediate mixing that caused aquaculture waste to enter the bottom pipe).

7.3.2 Cyanophyte presence

Overall 50 cyanophyte species were found in 17 reservoirs, while a total of 25 species was the highest cyanobacterial species diversity shown at any one site. There was no pattern indicating that any species would only occur at the sites where trout with tainted flesh were identified. However, 11 species were omnipresent and occurred in more than half of the reservoirs. These species were *Aphanothece* sp., *Anabaena circinalis*, *Anabaena* sp., *Dactylococcopsis* sp., *Merismopedia glauca*, *Microcystis robusta*, *M. minutissima*, *Oscillatoria splendida*, *O. lacustris*, *O. tenuis*. With the exception of *Merismopedia glauca*, all mentioned species produce tainting compounds (Jüttner & Watson 2007, Izaguirre & Taylor 2004). A literature review revealed that two thirds of the 50 species present have been found to release MIB and geosmin, causing algal taint (Jüttner & Watson 2007, Izaguirre & Taylor 2004). The reservoirs where algal taint occurred had only one or two species of cyanophytes >0.04 mg/L throughout the year, while all reservoirs without taint had five to nine species exceeding 0.04 mg/L within the course of a year (0.04 mg/L as the postulated minimum concentration of taint producers – MIB – necessary to cause taint).

Most reservoirs with algal tainted fish in October 2006 had elevated cyanobacterial biomasses around harvest time. In fact, four sites with algal taint at harvest in 2006, had a cyanophyte biomass <0.1 mg/L until September 2006, but biomasses >0.8 mg/L at sampling on 24th October 2006. The sites without tainted fish in 2006 had low cyanophyte abundances (<0.1 mg/L) in October and peaked in late November or as late as January 2007. A cyanophyte biomass increase from 0.01 to 1.86 mg/L (from 1 to 178 cells/ml of *Microcystis robusta*), within an 8-day period in September 2005 was observed in Reservoir 22 and this finding suggests that rapid biomass changes can occur and are difficult to predict. Single species dominated in the reservoirs with taint around harvest time and comprised of *Aphanothece* sp. at three sites, *Microcystis robusta* at two sites and *Chroococcus* sp. at one site.

The seasonal biomass distribution between the average reservoir with tainted fish and the average reservoir with untainted fish differed, probably reflecting different dominating species with their different life cycles and different favourable conditions (Figure 5.3). For production year 2006, the reservoirs with tainted fish developed the cyanophyte peak earlier in spring (average peak was 4.5 mg/L), on 25th October 2006, than in any other reservoir. Reservoirs where fish could be harvested without taint had an average biomass of 0.5 mg/L on 22nd November 2006, with an increasing trend only towards December.

Table 7.4: All species with biomasses of >0.1 mg/L at any given time during the study period from August 2005 to October 2007. Highlighted in dark grey are the sites with algal taint at harvest time in October 2006; in light grey, the three species that occurred in September or October at biomasses >0.1 mg/L in these six reservoirs.

Species / Reservoir	21	22	23	24	25	26	27	28	29	30	31	32	39	
Anabaena circinalis			+	+					+		+			4
Anabaena sp.		+	+	+							+	+	+	6
Aphanothece nidulans			+				+					+		3
Aphanothece sp.		+	+	+			+		+	+	+	+	+	9
Chroococcus sp.			+			+	+		+		+			5
Gloeocapsa punctata		+		+		+								3
Gloeothece linearis												+		1
Merismopedia glauca			+	+			+		+			+		5
Microcystis aeruginosa			+	+		+	+							4
Microcystis minutissima			+	+	+	+	+	+	+		+	+		9
Microcystis robusta	+		+	+	+	+	+	+	+	+	+	+		11
Microcystis sp.		+	+				+							3
Oscillatoria irrigua			+				+							2
Oscillatoria lacustris			+			+	+					+		4
Oscillatoria limnetica			+	+			+							3
Oscillatoria tenuis				+		+	+					+		4
Pleurocapsa sp.						+								1
Pyramidopsis sp.		+												1
Spirulina sp.				+										1

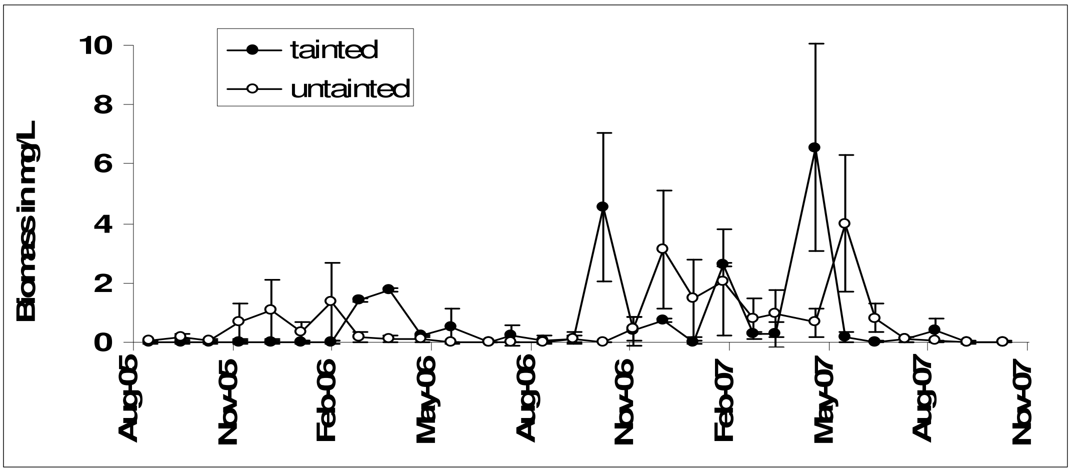

Figure 7.3: Seasonal distribution of cyanophyte biomass (in mg/L) in reservoirs with and without algal taint in fish at harvest (n=3 tainted fish; n=8 untainted fish).

The general cyanophyte biomass distribution of reservoirs known for tainted fish problems in comparison to “clear” sites showed no evident patterns (Figure 7.3). Figure 7.3 strengthens the finding that temperature plays a role in recruitment, with cyanophyte presence from November to April 2006 and October to June 2007. In 2007 an exceptionally warm autumn was experienced with hypolimnetic water temperatures of approximately 18 ℃ until May (SAWS 2007).

Although *Microcystis robusta* and *Aphanothece* sp. occurred at many sites, the species only caused tainting in some reservoirs. The seasonal occurrence of both species (Figure 7.4 and Figure 7.5) shows that their temperature preferences seem to differ. *Microcystis robusta* restricted its presence to the summer of 2006, with the hypolimnion temperature <17 ℃, while in 2007, the species occurred in winter as well (however, the water temperature in June 2007 was still at 18 ℃ in some reservoirs). *Aphanothece* sp. preferred spring and autumn conditions in 2007 (March and July), while in 2006 its presence stretched from February to July with an additional November peak. The hypolimnetic temperature exceeded 17 ℃ before the peaks. Due to the *Aphanothece* sp. biomass in spring and autumn, turnover initiated sediment re-suspension may play a stronger role with *Aphanothece* sp. than with *Microcystis* sp.

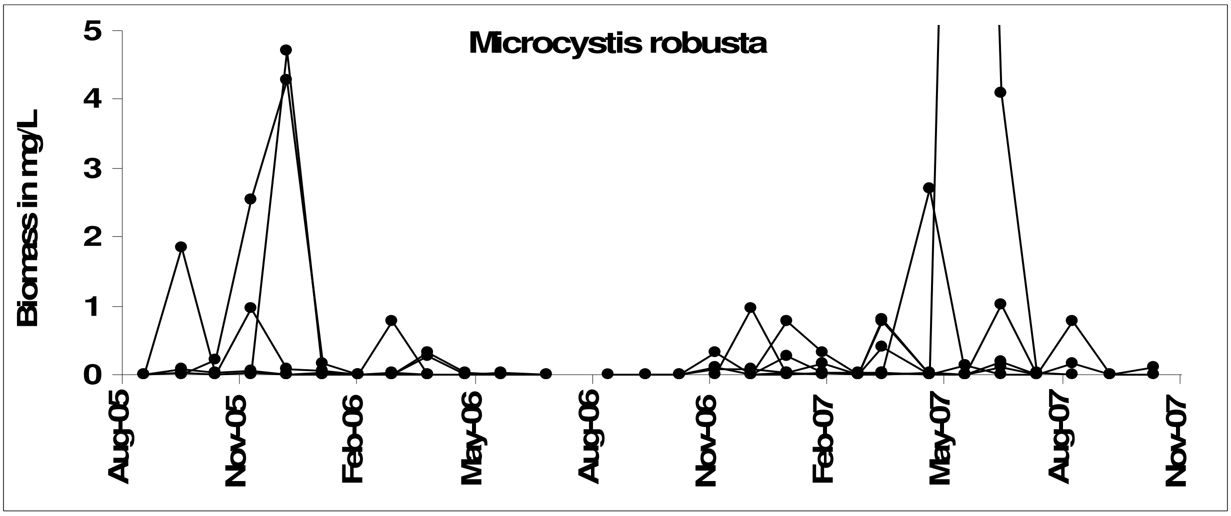

Figure 7.4: Seasonal distribution pattern of *Microcystis robusta* which occurred in 11 reservoirs. The June 2007 peak at one site was at approximately 20 mg/L (cut off).

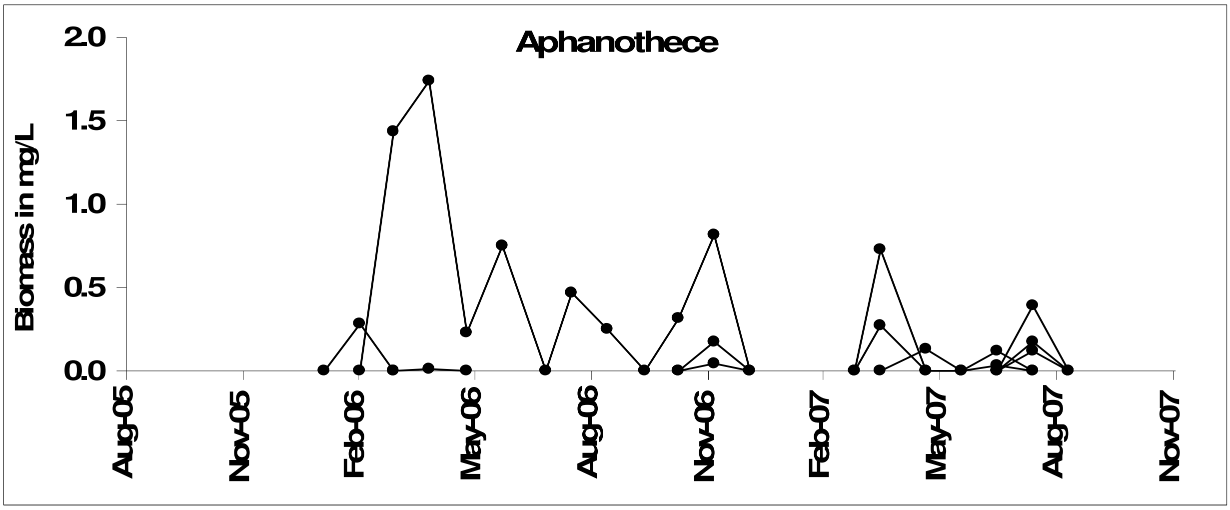

Figure 7.5: Seasonal distribution pattern of *Aphanothece* sp. which occurred in 9 reservoirs.

7.3.3 Factors controlling cyanophyte presence

Planktonic cyanophytes originate from cysts overwintering in the sediments and their recruitment is triggered by different factors, one of them temperature. Water temperatures recorded near the reservoir sediments in 11 reservoirs in October 2006 were much higher than in October 2005 (2005: 14.9 ± 1.0 °C; 2006: 17.0 ± 1.6 °C). In 2007 they decreased slightly to 16.6 ± 1.2 °C.

System descriptors that correlated significantly with algal taint occurrences were the number of cyanophyte species in the course of the year that yielded biomasses of 0.1 mg/L (maximum of 2 mg/L was found in the study area) ($p<0.05$). With all sites that had no cyanophyte peaks in October, five to nine species were found. In the production sites with tainted fish, only one or two cyanophyte species achieved biomasses >0.1 mg/L at any time of the year. There was one exception, a reservoir that had algal taint in 2006 with ten cyanophyte species of higher biomass (>0.1 mg/L). In 2008 the reservoir had no algal taint.

No site with a granite based geology developed algal taint problems (the cysts might prefer finer sediment material).

A parameter that correlated significantly with the presence of algal taint was the actual harvest date ($p<0.01$). All reservoirs that were harvested later than the 10th of November had algal taint, independent of any other condition. When comparing this date to the surface temperature of the water, a 19 °C/20 °C threshold for higher risk for cyanophytes can be set.

In the reservoirs with algal taint, the TN:TP ratio was always <7 (the TN:TP ratio in the reservoirs ranged from 2.4 to 12.7 as a yearly average).

There were no significant correlations between surface area, pH, total suspended solids (TSS), phosphorus content, duration of hypoxic hypolimnetic conditions, bottom ammonia content or general algal biomass and the occurrence of tainted fish around harvest time.

7.4 Discussion

Davenport et al. (2003) reported that cyanophyte blooms (including harmful algal blooms) were increasing globally, but were not easily, causally linked with pollution. There is agreement that reservoirs with oligotrophic phosphorus conditions (TP concentrations <25 µg/L) rarely show significant cyanophyte presence (>0.1 mg/L) and therefore no tainting (Dodds 2002), but low phosphorus conditions were not recorded in any of the studied reservoirs in the Western Cape. All sites harbour the three species that caused algal taint in 2006,,with sufficient biomass to cause taint in most sites, but only one quarter of the production reservoirs showed tainting.

The group that were most probably responsible for algal taint in the studied Western Cape reservoirs were indeed cyanobacteria. The life cycle of cyanobacteria includes sedimentation, overwintering in the sediment and recruitment (usually in spring), with biomass increase and migration to the surface (Whitton & Potts 2000). The critical stages are recruitment and consequent biomass increase, with the equivalent supporting factors. The main drivers of recruitment are temperature, re-suspension, bioturbation, light, nutrients and anoxia (Tan et al. 2008).

Temperature was identified as an important factor affecting abundance and distribution of cyanophytes in the reservoirs. The observation was that no algal abundances >0.5 mg/L (considerable taint) were found unless the temperatures in the hypolimnion were >17 °C and >19 °C in surface water. Fay (2009) found that the preferred temperature range of cyanophytes was 15 to 30 °C, while a study in China suggested 18 to 20 °C as the optimum temperature for *Microcystis* sp. recruitment (Tao et al. 2005). Westerhoff et al. (2005) showed that surface water temperatures correlated positively with MIB concentrations in water supply reservoirs in Arizona. MIB was only present at surface temperatures of 20 °C and higher.

Re-suspension as a means of recruitment can be another important factor (Verspagen et al. 2005, Rengefors et al. 2004, Stahl-Debanco & Hansson 2002). In general, the recruitment and life cycle of *Aphanothece* sp. and *Chroococcus* sp. were less intensively studied in the literature than *Microcystis* sp.. They are both smaller in size than *Microcystis* sp. and their appearance showed no correlation with temperature distributions. Chroococcal cyanobacteria (*Aphanothece* sp. and *Chroococcus* sp.) did occur at lower temperatures so other factors may play a more important role. The TSS measurements in the reservoir hypolimnion and epilimnion were significantly elevated at times, both before or at chroococcal peaks, so re-suspension may be an important factor for this group. Most cyanophytes are actually attached to particles and only one third are planktonic (Verspagen et al. 2005). Bioturbation is another factor that can enhance sediment re-suspension and algal recruitment, but would be regarded as negligible under the primarily anoxic conditions in the hypolimnia where benthic life would be expected to be scarce. The fact that resuspension plays an important role in Western Cape cyanobacterial recruitment was confirmed by the finding that no algal taint was found in granite dominated catchments. In these reservoirs, the sediment consisted of larger particles which are not as easily re-suspended.

Not only does recruitment from the sediment play a role, but the number of cysts that accumulated in the sediment (every time after cyanophytes had a planktonic stage) and cyst survival after settlement are also important factors (Brunberg & Blomqvist 2002). Cysts prefer darkness, coupled with low temperatures and hypoxia (Reynolds et al. 1981) which is why long anoxic conditions enhance the chance of cyst recruitment. Cysts can survive for several years. Light is not perceived as an important factor in Western Cape reservoirs due to the relatively low Secchi depths (never more then 2.5 m) and the depth of >6 m of all reservoirs.

Nitrogen to phosphorus ratios have often been used to explain cyanophyte presence or dominance (Havens et al. 2003, Barbiero et al. 1996). With low nitrogen availability in the water, species with the ability to fixate air-bound nitrogen have an advantage (many cyanophytes). Although not an exclusive criterion (because ratios were also lower at other sites), the TN:TP mass ratio in the reservoirs that developed tainted fish was <7 in all cases.

Stahl-Debanco et al. (2003) found that intermediate phosphorus contents (134 µg/L) and low TN:TP ratios (<8) favoured maximum cyanophyte recruitment in laboratory studies. In fact, the sites with the highest cyanophyte peaks overall fell within a similar hypolimnetic total phosphorus range (131 to 183 µg/L). However, not all of these sites had algal taint.

With phosphorus concentrations >600 mg/L in the hypolimnion, dinophyte cysts showed an advantage and cyanophytes disappeared altogether which was also found by Jeppesen et al. (2005) in Swedish lakes. The absence of the cyanophyte risk in hypertrophic reservoirs is a trade-off, since dinophytes will not cause algal taint in trout, but they can enhance blockage of sensitive irrigation systems (filters) and create unpleasant odours in drinking water (Van Ginkel et al. 2007).

Reservoirs that developed algal taint in previous years tended to develop it again in following years. This is definitely a sign of mass cyst occurrences in the sediment with concurrent favourable recruitment and developmental conditions. Eutrophication with consequent elevated surface phosphorus concentrations and longer hypolimnetic anoxia can enhance the longevity of cyanobacterial cysts (Chapter 3). Reservoirs with many competing cyanophyte species were less likely to develop cyanobacterial biomass around the time of trout harvest.

7.5 Conclusions

Many factors play a role in controlling cyanophyte presence and abundance in reservoirs, of which the most important are general water quality conditions, means of inoculation, competition with other algal groups and zooplankton grazing, consequent settlement of cysts in the sediment, sediment conditions and cyst survival, recruitment conditions and factors controlling the buoyancy and migration of species towards the surface. Once cyanophyte presence was established in higher biomasses in a lake, the repeated occurrence of certain patterns continues until the general water quality conditions change for better or worse (100 to 500 mg/L hypolimnetic total phosphorus concentration as suggested, the initial factor of cyanophyte presence in biomasses >0.5 mg/L). Other factors will thereafter control the actual seasonal distribution and biomass of cyanophytes. Generally, cyanophytes rarely dominated the phytoplankton structure of the studied reservoirs (Chapter 2), which decreases the risk of bloom formation and the development of health hazards (harmful algal blooms). However, taint in irrigation and drinking water and taint in water bound products (e.g. fish) can be caused by very low cyanophyte biomasses (0.1 mg/L) and their release of respective substances.

Factors that could be linked to low cyanophyte abundance in Western Cape reservoirs were summarized as characteristics of sites with a lower chance of algal taint around the harvesting period (spring). None of the sites included in this study were devoid of cyanophyte species. While the main period for cyanophyte peaks (>0.5 mg/L) was September to March, there were additional peaks in May or June at some sites. Also, the biomass of peaks ranged from 0.3 to 19.0 mg/L with no correlation to nutrient contents or any other physico-chemical water parameters.

Sites with a lower risk of developing tainting cyanobacterial concentrations (especially in October or early November) were in summary, reservoirs with most of the following characteristics:

- Deeper reservoirs (>9 m) and reservoirs of low hypolimnetic temperatures (<15-17 °C), especially until harvest time

- Granite dominated basin and catchment area
- Sites of low average inorganic phosphorus concentrations (<90 µg/L) or sufficient water exchange
- TN:TP ratio >7 (by mass: mg/L N:mg/L P)
- No history of algal taint at certain periods of time
- No history of certain species (e.g. *Aphanothece* sp., *Chroococcus* sp., *Microcystis robusta*)
- Diverse cyanophyte community (5 to 9 species with biomasses >0.1 mg/L during the course of the year)
- Dinophyte dominance (trade-off due to possible filter blockage or taint in drinking water).

7.6 References

Barbiero, R. P., Speziale, B. J., and Ashby, S. L. (1996). Phytoplankton community succession in a lake subjected to artificial circulation. Hydrobiologia 331(1-3): 109-120.

Branco, L. H. Z., Necchi, O., and Branco, C. C. Z. (2001). Ecological distribution of Cyanophyceae in lotic ecosystems of São Paulo State. Revista Brasileira de Botânica 24(1): 99-108.

Brunberg, A.-K. and Blomqvist, P. (2002). Benthic overwintering of *Microcystis* colonies under different environmental conditions. Journal of Plankton Research 24(11): 1247-1252.

Clesceri, L. S., Greenberg, A. E., and Eaton, A. D. (1999). Standard Methods for Examination of Water and Wastewater . Washington D.C., American Public Health Association.

Davenport, J., Black, K., Burnell, G., Cross, T., Cullory, S., Ekaratne, S., Furness, B., Mulcahy, M., and Thetmeyer, H. (2003). Aquaculture - the ecological issues. Oxford, UK: Blackwell Science Ltd.

Dodds, W. K. (2002). Freshwater Ecology. Academic Press.

Durrer, M., Zimmermann, U., and Jüttner, F. (1999). Dissolved and particle-bound geosmin in a mesotrophic lake (lake Zürich): spatial and seasonal distribution and the effect of grazers. Water Research 33(17): 3628-3636.

Entwisle, T. J., Sonneman, J. A., and Lewis, S. H. (1997). Freshwater Algae in Australia - a guide to conspicuous genera. Potts Point, Australia: Sainty and Associates Pty Ltd.

Fay, P. (2009). Viability of akinetes of the planktonic cyanobacterium *Anabaena circinalis*. Proceedings of the Royal Society B 234: 283-301.

Havens, K. E., James, R. T., East, T. L., and Smith, V. H. (2003). N:P ratios, light limitation, and cyanobacterial dominance in a subtropical lake impacted by non-point source nutrient pollution. Environmental Pollution 122: 379-390.

Häusler, J. (1982). Schizomycetes. Stuttgart, New York: Gustav Fischer Verlag.

Howgate, P. (2004). Tainting of farmed fish by geosmin and 2-methyl-iso-borneol: a review of sensory aspects and of uptake/depuration. Aquaculture 234(1-4): 155-181.

Huber-Pestalozzi, G. (1938). Das Phytoplankton des Süßwassers - 1. Teil Allgemeiner Teil, Blaualgen, Bakterien, Pilze. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Izaguirre, G. and Taylor W.D. (2004). A guide to geosmin- and MIB-producing cyanobacteria in the United States. Water Science and Technology 49(9): 19-24.

Jeppesen, E., Soendergaard, M., Jensen, J. P., Havens, K. E., Anneville, O., Carvalho, L., Coveney, M. F., Deneke, R., Dokulil, M. T., Foy, B., Gerdeaux, D., Hampton, S. E., Hilt, S., Kangur, K., Köhler, J., Lammens, E. H. H. R., Lauridsen, T. L., Manca, M., Miracle, M. R., Moss, B., Noges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Schelske, C. L., Straile, D., Tatrai, I., Willen, E., and Winder, M. (2005). Lake responses to reduced nutrient loading - an analysis of contemporary long-term data from 35 case studies. Freshwater Biology 50: 1747-1771.

Jobling, Koskela, and Sabolainen (1998). Influence of dietary fat level and increased adiposity on growth and fat deposition in rainbow trout, *Oncorhynchus mykiss* (Walbaum) . Aquaculture Research 29(8): 601-607.

Jüttner, F. and Watson, S. B. (2007). Biochemical and Ecological Control of Geosmin and 2-Methylisoborneol in Source-Waters. Applied and environmental microbiology 73(14): 4395-4406.

Klausen, C., Nicolaisen, M. H., Strobel, B. W., Warnecke, F., Nielsen, J. L., Jørgensen, and N.O.G. (2005). Abundance of actinobacteria and production of geosmin and 2-methylisoborneol in Danish streams and fish ponds. FEMS Microbiology Ecology 52(2): 265-278.

Maleri, M. (2009). Site selection and production performance of rainbow trout (Oncorhynchus mykiss) cage operations in small farm reservoirs: the Western Cape experience, South Africa. Aquaculture Research 40: 18-25.

Morscheid, H., Fromme, H., Krause, D., Kurmayer, R., Morscheid, H., and Teubner, K. (2006). Toxinbildende Cyanobakterien (Blaualgen) in bayerischen Gewässern - Massenentwicklungen, Gefährdungspotential und wasserwirtschaftlicher Bezug. Materialienband Nr. 125. Munic, Germany: Bayerisches Landesamt für Umwelt.

Prescott, G. W. (1978). How to know the freshwater algae. Dubuque, Iowa, USA: Wm. C. Brown Company Publishers.

Regost, C., Arzel, J., Cardinal, M., Laroche, M., and Kaushik, S. J. (2001). Fat deposition and flesh quality in seawater reared, triploid brown trout (*Salmo trutta*) as affected by dietary fat levels and starvation. Aquaculture 193(3-4):325-345.

Rengefors, K., Gustafsson, S., and Stahl-Delbanco, A. (2004). Factors regulating the recruitment of cyanobacterial and eukaryotic phytoplankton from littoral and profundal sediments. Aquatic Microbial Ecology Journal 36: 213-226.

Reynolds, C. S., Jaworski, G. H. M., Cmiech, H. A., and Leedale, G. F. (1981). On the annual cycle of the blue-green alga *M. aeruginosa* Kütz. Philosophical Transactions of the Royal Society B: Biological Sciences 293: 419-477.

Robarts, R. D. and Zohary, T. (1987). Temperature effects on photosynthetic capacity, respiration, and growth rates of bloom-forming cyanobacteria. New Zealand Journal of Marine and Freshwater Research 21: 391-399.

Robertson, R. F., Jauncey, K., Beveridge, M. C. M., and Lawton, L. A. (2005). Depuration rates and the sensory threshold concentration of geosmin responsible for earthy-musty taint in rainbow trout, *Onchorhynchus mykiss*. Aquaculture 245(1-4): 89-99.

Robertson, R. F., Hammond, A., Jauncey, K., Beveridge, M. C. M., and Lawton, L. A. (2006). An investigation into the occurrence of geosmin responsible for earthy–musty taints in UK farmed rainbow trout, *Onchorhynchus mykiss*. Aquaculture 259(1-4): 153-163.

Robin, J., Cravedi, J. P., Hillenweck, A., Deshayes, C., and Vallod, D. (2006). Off flavor characterization and origin in French trout farming. Aquaculture 260: 128-138.

Rouhani, Q. A. and Britz, P. J. (2004). Contribution of aquaculture to rural livelihoods in South Africa: A baseline study. WRC Report No: TT 235/04. Pretoria, Department of Water Affairs and Forestry.

SAWS (2007). Weather data. Cape Town: South African Weather Service.

Selli, S., Rannou, C., Prost, C., Robin, J., and Serot, T. (2006). Characterization of Aroma-Active Compounds in Rainbow Trout (*Oncorhynchus mykiss*) Eliciting an Off-Odor. Journal of Agricultural and Food Chemistry 54(25): 9496-9502.

Sommer, U. (1996). Plankton ecology: The past two decades of progress . Naturwissenschaften 83(7): 293-301.

Stahl-Delbanco, A. and Hansson, L.-A. (2002). Effects of bioturbation on recruitment of algal cells from the "seed bank" of lake sediments. Limnology and Oceanography 47: 1836-1843.

Stahl-Delbanco, A., Hansson, L.-A., and Gyllström, M. (2003). Recruitment of resting stages may induce blooms of Microcystis at low N:P ratios. Journal of Plankton Research 25(9): 1099-1106.

Strickland, J. D. H. (2009). Measuring the production of marine phytoplankton. Bulletin No 122. Ottawa, Fisheries Research Board of Canada.

Stubbs, G. (2007). Western Cape Trout Association - Annual Meeting. Stellenbosch, South Africa.

Tan, X., Kong, F.-X., Cao, H.-S., Yu, Y., and Zhang, M. (2008). Recruitment of bloom-forming cyanobacteria and its driving factors. African Journal of Biotechnology 7(25): 4726-4731.

Tao, Y., Kong, F. X., Cao, H. S., and Zhang, X. F. (2005). Laboratory investigations on recruitment of Microcystis in sediment of Taihu Lake. Journal of Lake Science 17: 231-236.

Tung, S.-C. (2006). Identification and Oxidation of 2-MIB and Geosmin in Source Water. Tainan City, Taiwan, Cheng Kung University.

Utermöhl, H. (1958). Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. Mitteilungen der internationalen Vereinigung der theoretischen und angewandten Limnologie 5: 567-596.

van den Hoek, C., Mann, D. G., and Jahns, H. M. (1995). Algae - an introduction to phycology. Cambridge: Cambridge University Press.

van Ginkel, C. E., Cao, H., Recknagel, F., and du Plessis, S. (2007). Forecasting of dinoflagellate blooms in warm-monomictic hypertrophic reservoirs in South Africa by means of rule-based agents. Water SA 33(4): 531-538.

Verspagen, J. M. H., Snelder, E. O. F. M., Visser, P. M., Jöhnk, K. D., Ibelings, B. W., Mur, L. R., and Huisman, J. (2005). Benthic-pelagic coupling in the population dynamics of the harmful cyanobacterium Microcystis. Freshwater Biology 50: 854-867.

Warrington, P. (2001). Aquatic Pathogens – Cyanophytes. Victoria BC, Canada, Environmental Protection Division.

Westerhoff, P., Rodriguez-Hernandez, M., Baker, L., and Sommerfeld, M. (2005). Seasonal occurrence and degradation of 2-methylisoborneol in water supply reservoirs. Water Research 39(20): 4899-4912.

Whitton, B. A. and Potts, M. (2000). The ecology of cyanobacteria: Their Diversity in Time and Space. Dordrecht, Netherlands: Kluwer Academic Publishers.

Yunfang, H. M. S. (1995). Atlas of Freshwater biota in China. Beijing: China Ocean Press.

Zaitlin, B. and Watson, S. B. (2006). Actinomycetes in relation to taste and odour in drinking water : Myths, tenets and truths. Water Research 40(9): 1741-1753.

Zhang, T., Li, L., Song, L., and Chen, W. (2009). Effects of temperature and light on the growth and geosmin production of *Lyngbya kuetzingii* (Cyanophyta). Journal of Applied Phycology 21(3): 279-285.

CHAPTER 8 SITE SELECTION AND PRODUCTION PERFORMANCE OF RAINBOW TROUT (ONCORHYNCHUS MYKISS) CAGE OPERATIONS IN SMALL FARM RESERVOIRS: THE WESTERN CAPE EXPERIENCE

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Abstract

Cage farming of rainbow trout (*Oncorhynchus mykiss*) has expanded rapidly into private farm reservoirs of the Western Cape. During the grow-out season 2006, production problems such as increased mortality rates and off-flavour of the final product caused economic losses.

To set criteria for improved site selection, water quality information of surface and near-bottom water was correlated with morphometric reservoir data, fish production data as well as the observed production difficulties.

The data revealed good linkage between reservoir information and fish production data as well as production problems. The production problems correlated ($p<0.01$) with single water quality parameters so that suggestions for risk management indicators can be made.

By ranking the data, threshold values for better production practice were determined. Thereafter, reservoirs of a minimum surface area of at least three ha would be recommended as well as reservoirs with good water exchange rates (>1 per year) to avoid oxygen problems. Reservoirs of larger surface area (>5 ha) produced significantly larger fish that grew faster and showed lower mortality rates.

8.1 Introduction

South Africa's potential in aquaculture has yet to be reached. On a worldwide scale, the African continent is among the minor role players and within Africa, South Africa's contribution to aquaculture is below 1 % (FAO 2007). One of the major target species for freshwater aquaculture in South Africa to date is rainbow trout, *Oncorhynchus mykiss* (Walbaum 1792) (Hoffman et al. 2000, AASA conference 2007). The processing industry of South Africa has to import the majority of this species to satisfy the market demand of delicacy products and gourmet fare in the tourist industry (Western Cape Trout Association meeting June 2007) and has a high interest in local production rather than import. In 2006, the South African production of *O. mykiss* totalled at about 900 t. The production of rainbow trout in the Western Cape amounted to 320 t in 2006 with large fluctuations in production numbers in the past, however an average growth rate of 3 % since 1993 according to information exchanged at the Western Cape Trout Association meeting in June 2007.

To support historically disadvantaged members of the communities financially and provide trout as local resource to the fish industry, the Hands-On Fish Farmer's Co-operative Limited was founded. The basic idea was to make use of the manifold existing water structures in the Western Cape Region of South Africa, small irrigation reservoirs (1.5 to 20 ha), and grow trout in cages during the winter months when water temperatures allow production in the Mediterranean climatic conditions.

Albeit the advantages of cage aquaculture such as low capital requirements and no water consumption, minor to severe water quality related problems emerged in more than half of the small scale farmers' trout projects in the Western Cape of South Africa. These problems were recognized to be related to nutrient loading (eutrophication) and co-dependent excessive algal biomass and altered succession. There was no correlation between the number of years the trout operations were already in place and the likelihood of production setbacks, so that the production problems of 2006 have been attributed to the existing ecological condition of the reservoir and not the changed conditions after aquaculture input. Therefore, unsuitable selection of production sites was postulated.

Because of the dry summer conditions and seasonal rainfall patterns (during winter) in South Africa, most surface water is stored in reservoirs to provide water during the summer production season. Next to large regional drinking water reservoirs, these small reservoirs which are primarily privately owned, offer an opportunity for non-water-consumptive aquaculture such as net-cage farming. Currently, the Western Cape region has more than four thousand registered farm reservoirs (dam wall higher than 5 m) with a total storage volume of 100 million cubic meters (Berg et al. 1994). Only the winter in the study area is of suitable character for open water trout farming. Therefore, most producers buy in juvenile fishes in late autumn (April and May) and grow them from sizes between 100 and 250 g to the market size of 900 to 1300 g.

The individual stocking density within the cage varied hereby from an average of 1.2 kg trout/m^3 at start to a maximum of 8.1 kg trout/m^3 at harvest time. The current environmental stocking density varied from about 380 fish to about 3300 fish per ha of total reservoir surface. Because of the relatively low individual stocking rate, a 5 t production per farm reservoir was originally perceived to be of low environmental impact. Experiences with trout production in small standing water bodies in South Africa were limited. On a world-wide scale, most production of rainbow trout takes place in large reservoirs. When small reservoirs are utilised, they have a high water exchange rate by being through-flown by rivers. In Australia, experiments with small farm reservoirs for small-scale trout cage production is a parallel development and 375 fish ha^{-1} were suggested as environmental stocking limit (Gooley & Gavine 2003).

Water quality plays a crucial role for aquaculture operations. To limit risks and make informed decisions, the water quality of the water bodies before operations start should guarantee that production is viable. Furthermore, the water bodies have to maintain the qualitative level required for concurrent water uses such as irrigation, drinking water or water for agriculture related activities. Ultimately, it is recommended that the self-cleaning ability of the reservoirs is maintained to save great remedial costs for all users.

The main problem observed in the process of trout production in Western Cape farm reservoirs was oxygen depletion in the surface water causing elevated mortality levels and decelerated growth. Another problem lay in the tainting of fish caused by substances released by specific microscopic algae (phytoplankton), which lead to a loss in flesh quality and reduced market prices.

Production performance diverged extremely in the past and improvements had to be made on intuitive basis rather than with an analysis of the main factors driving success. Water quality changes were especially perceived as difficult to predict and tools for risk minimisation were missing.

Farm reservoir information, fish-production related information collected during the production process, and water quality information were compared for this paper. Insights on optimal reservoir selection and avoidance strategies of production problems were the main aim of the study.

The results of this study can be used to set criteria for future decision making, and to avoid preventable economic risks and losses, as well as distress for the people from an underprivileged background that were given the opportunity to supplement their income and to gain experiences with business operations.

8.2 Methods

The Hands-On Fish Farmer's Co-operative Limited established the infrastructure for observations and research on net-cage operations of *O. mykiss* in the Western Cape. The project has operated since 1996 and the similarity and synchronization of the operations enables direct comparison of fish data with water quality data from up to 21 different water bodies. All cage systems were designed similarly, the fish farmers were trained in Stellenbosch University organised courses, and all water quality information was collected and analysed uniformly.

Water quality information was sampled monthly for a minimised selection of reservoirs between August 2005 and April 2007. Water samplings for oxygen, temperature, total phosphorus and ammonia at the surface and near the reservoir sediment led to information on seasonal cycles and nutrient distribution in small farm reservoirs of the Western Cape Province (Chapter 3 and Chapter 4). The end of October 2006 was accordingly chosen for the water quality sampling of 21 production reservoirs for direct comparison with the 2006 production statistics. The detailed overview of all parameters included in that comparison can be found in Table 8.1.

Table 8.1: Categories and parameters as used to feed the correlation analysis.

Reservoir data (morphometric data)	Surface area
	Maximum depth
	Volume at full supply
Production data	FCR (food conversion ratio)
	Maximum fish size
	Growth rate
	Mortality
	Duration of grow-out in cage
	Date of harvest
Water quality	Surface and bottom oxygen
	Secchi depth
	Conductivity
	Total suspended solids
	Surface and bottom pH
	Surface and bottom TP
	Surface and bottom NH_3
	Alkalinity
	Ceratium spp. abundance
	Microcystis spp. presence
	Number of present algae species
	Total phytoplankton biomass
Production performance	Presence of oxygen problems
	Occurrence of algal taint
	Pigmentation
	Problems with primary reservoir uses

8.2.1 Morphometric information

A total of 21 reservoirs were investigated for this study, all of them within the south-western area of the Western Cape Province of South Africa. The farm furthest in the north was located at 33° 4' 52'' S and 19° 18' 38'' E and the furthest reservoir in the south at 34° 17' 35'' S and 19° 9' 4'' E.

The length and width of each reservoir were measured using Google Earth software (http://earth.google.com) and the surface area in hectares was estimated via the nearest geometrical shape. The area was calculated for the case of full supply water volume of the reservoir. The reservoir depth at the cage location was measured with help of the water sampler's rope where the sampler itself acted as weight to indicate the ground. In all but one case, the depth at cage location represented the maximum depth of the reservoirs. The exception was the deepest reservoir where the production site was at a depth of two thirds of the maximum depth.

8.2.2 Fish production data

The fish produced in the farm reservoirs were weighed and counted at the time of introduction into the reservoir and again at harvest time. Approximately 20 samples of 10 fish were weighed. Mortality was calculated from the difference between the number of fish stocked into the reservoir and the number of individuals harvested. The mortality rate did not differentiate between fishes lost to natural mortality and escapes during production or lost by predators or theft. The growth of the fish was recorded as specific growth rate, where the difference between mean initial weight and weight at harvest was divided by the growth period in days and presented as a percentage.

The food conversion ratio (FCR) to assess feeding management on the farms was determined. The total amount of dry feed used at one site per season was divided by the total wet weight of fish harvested. Fishes lost due to mortality, escapes, theft or predation were therefore incorporated into the FCR value as was any uneaten or wasted feed.

Contact with the farmers was made and information on occurrences during production exchanged as well as problems monitored.

8.2.3 Water quality

The turbidity was recorded by means of a Secchi disc. Water was collected from a 2-m depth and from the bottom area (30 to 50 cm above the sediment surface) by means of a 1.5 L water sampler with a single line trigger mechanism (The Science Source, Waldoboro, USA). Oxygen and temperature were measured (OxyGuard Polaris, OxyGuard International A/S, Birkerød, Denmark) on site. The water samples were cooled and stored, and analysed the next day for conductivity (Hach CO 150, Hach Company, Loveland, USA), alkalinity (buret titration with 0.02 N sulphuric acid), pH (Hanna pH 211 microprocessor, Hanna Instruments, Woonsocket, USA), total suspended solids, ammonia and total phosphate. The latter three parameters were analysed via colorimetric methods (Hach DR/890 procedures manual 2004, Hach Company, Loveland, USA). Total suspended solids were determined photometrically by measuring the thoroughly mixed, but otherwise unprepared samples against demineralised water. Ammonia was determined via the salicylate method. The samples for total phosphate analysis were acid-digested and subsequently analysed for soluble reactive phosphate by means of the molybdo-vanadate method.

Unfiltered water from a 2-m depth was preserved with Lugol’s solution to determine dominant phytoplankton species and phytoplankton quantity (Utermöhl 1958).

The presence or absence of algal taint was determined by the processor purchasing all the fish produced by the twenty-one reservoirs. Quality control was undertaken by a taste panel using organoleptic methods to verify the optimal quality of fish products. The differentiation made was between no, slight, or strong taint which was included via a simple scale.

8.2.4 Statistical analyses

Linkages between water quality, reservoir and fish production information were evaluated by correlation matrices and Spearman ranking (Statistica 7.0, StatSoft Inc., Tulsa, USA). Differences were considered significant when $p<0.05$ and highly significant when $p<0.01$.
Box and Whisker plots visualise the observations made during the monthly sampling events.

8.3 Results

8.3.1 Reservoir turnover and nutrient distribution

During the stagnation phase, the bottom water of 14 of 16 observed farm reservoirs became anoxic to a level of less than 1.0 mg/L oxygen. The core of the anoxic phase lasted from November 2005 to February 2006 and the same months in the summer of 2006/2007. The hypolimnion of 40 % of the reservoirs established oxygen depletion in the course of September 2005 as well as of 30 % of the reservoirs in September 2006. The residual reservoirs' hypolimnia became oxygen depleted during October and November depending on the mesoclimatic conditions. In autumn, most reservoirs started mixing in March. During the mixing period, the surface and bottom levels of oxygen approximate gradually to similar levels. The core period where most reservoirs were supplied with near-surface level oxygen at the bottom (fully mixed) was May to August 2006.

During the turnover phase, the ammonia levels at the bottom varied little between 0.02 and 0.14 mg/L ammonia-nitrogen. At the surface, the ammonia levels varied between 0.03 and 0.12 mg/L ammonia nitrogen.

In contrast, among reservoirs during the stagnation phase when the ammonia concentration decreased in the surface layer and accumulated in the near-bottom water, the ammonia distribution varied greatly. The ammonia level development in the bottom water samples between August 2005 and April 2007 (Figure 8.1). Five reservoirs distinguished themselves strongly from the others (Figure 8.2). In summer, one type of reservoir accumulated ammonia at the bottom to levels between 1 and 5 mg/L ammonia nitrogen, which indicated that abundant nitrogen resources were converted from nitrate or released from the sediment under anoxic conditions (Jones et al. 1982) whereas the other group maintained similar ammonia levels at the bottom throughout the year, even under anoxic conditions (Beutel 2001).

The total phosphorus levels in the bottom water showed similar trends. More total phosphorus accumulated at the bottom during the stagnation in the summer months and was released from the sediment under anoxic conditions, but the trends were not as distinct as with ammonia.

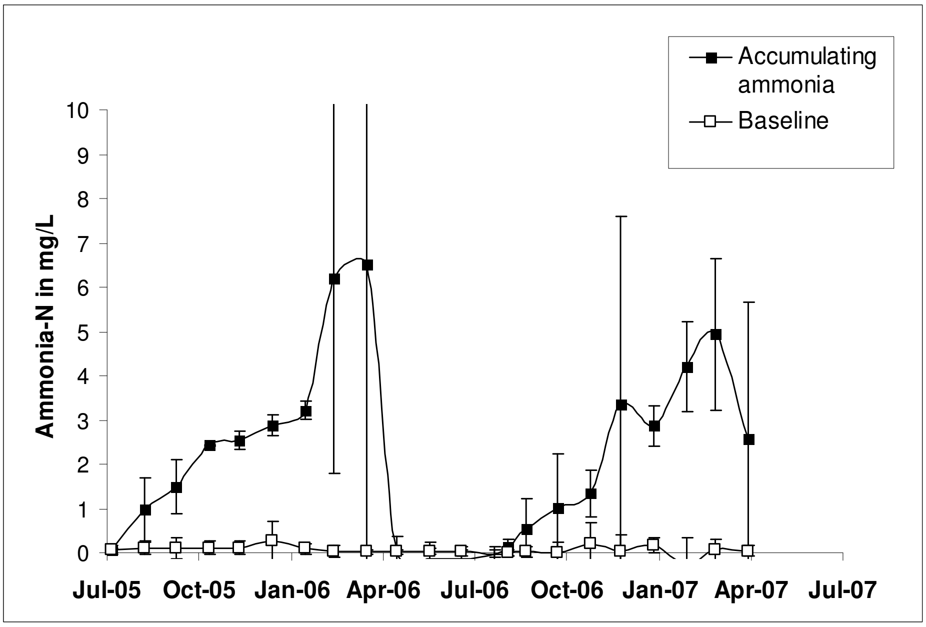


Figure 8.1: Ammonia nitrogen levels in the hypolimnion from July 2005 to April 2007. Ammonia accumulating group (black) n=5, baseline group with even ammonia levels throughout the year (white), n=11.

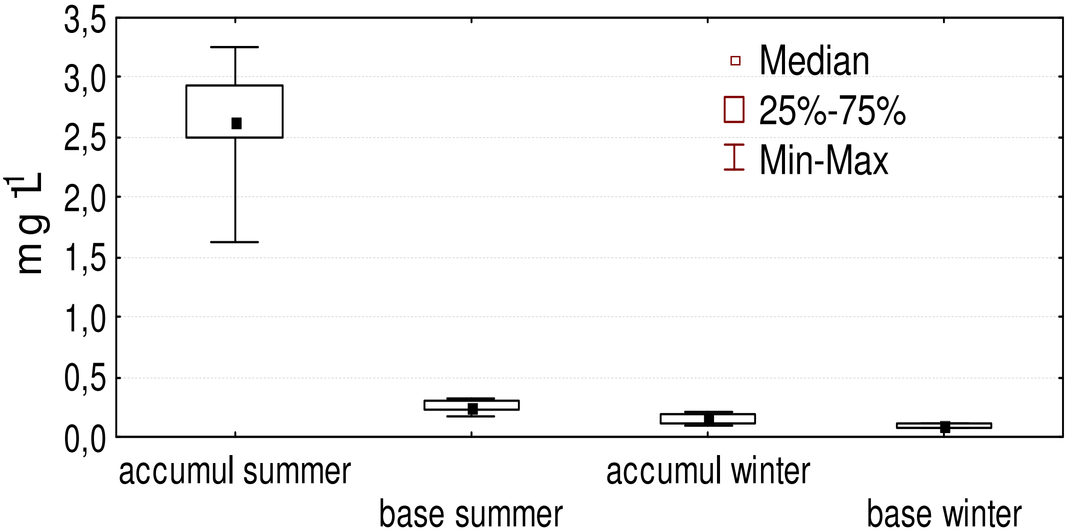


Figure 8.2: Median concentrations of ammonia nitrogen in the hypolimnion in reservoirs >1 mg/L ammonia (accumulated) and <1 mg/L (base) and in two main seasons summer (stagnation) and winter (reservoir turnover).

Three reservoirs had notably alkaline pH levels (>9) in the surface water in October. The pH could be one possible indicator for phytoplankton biomass in the epilimnion (surface layer) and its impacts on aquaculture production. None of the reservoirs had a pH of less than 6.

The information on reservoir hydrodynamics informed the timing of the sampling for the comparison of water quality and aquaculture data. The stagnation phase shows distinct differences among reservoirs and its beginning correlates with the harvest time.

8.3.2 Correlations among reservoir and fish production parameters

The 21 sites' surface areas varied between 1.3 ha and 13.4 ha at full supply level. Approximately half of the reservoirs had a surface area of less than 5 ha. The water column below the cages varied between 4.5 and 14 m. The surface area correlated positively with the maximum reservoir depth ($R^2 = 0.5091$, $p<0.05$) and the volume.

The FCR ratios varied from 1.1 to 2.5. Eighty percent of all production sites achieved a FCR below 2.0 and 50 % of all production sites achieved a FCR less than 1.5. The higher the FCR, the smaller were the maximum fish size (R^2=-0.53, $p<0.05$) and the growth rate (R^2=-0.59, $p<0.01$) per reservoir. If the mortality was high on the other hand, the FCR value was significantly increased (R^2=0.85, $p<0.01$).

The average growth rate of *O. mykiss* for the complete grow-out phase varied between 1.4 and 9.1 grams per day (or 1 to 1.6 % of total fish weight per day, calculated as average of 3 to 4 growth periods) in the different production systems. Sixty % of all production sites ranged between 4 and 6 g body weight addition per day. If many fish died or were lost during production, the growth rate was naturally affected as well (R^2=-0.45, $p<0.05$). The growth rate was closely linked to the maximum size of the fish harvested (R^2=0.72, $p<0.01$) which ranged from 695 to 1360 g total weight for 95 % of the projects (with one outlier that achieved only 420 g). The mortality as percentage of individuals lost in the production process varied between 5 and 50 %.

8.3.3 Correlations among water quality parameters

The water quality parameters showed strong correlations amongst each other. In a network map connecting all water quality parameters with existing correlations, ΔNH_3-N (difference between surface and near-bottom water ammonia at the end of October), temperature, ΔpH, ΔTP, ΔO_2 and alkalinity turn out as primarily connected nodal points with four or more correlative relations to other parameters.

ΔNH_3-N ranged from 0.01 to 2.13 mg/L ammonia nitrogen with four reservoirs (19 %) having values of higher than 1 mg/L ammonia nitrogen and seven reservoirs (33 %) having values of higher than 0.5 mg/L ammonia nitrogen. ΔNH_3-N correlated strongest ($p<0.01$) with surface pH and bottom total phosphorus. The higher the ammonia nitrogen levels at bottom, the higher the surface pH and the higher the bottom total phosphorus levels.

The bottom total phosphorus levels of the 21 reservoirs varied between 0.005 and 0.760 mg/L. 24 % of the reservoirs had a bottom-water phosphorus level of 0.5 mg/L total phosphorus or higher.

Five reservoirs had a pH difference between surface and near-sediment water of more than 2 units during the October 2006 measurements. Nine reservoirs had a pH difference of 1.5 units or more. The actual surface pH of six reservoirs exceeded the level of 8.4 (recommended upper limit for trout production), five of which exceeded the pH of 9 (measured between 1000 and 1600 hours). ΔpH could be a possible indicator of algal production in these small reservoirs. Photosynthesis influences pH directly by removing CO_2 and high algal biomass was observed in reservoirs of highest pH, most often dominated by *Ceratium hirundinella* and *Peridinium spp.* during winter.

The presence and abundance of *C. hirundinella*, correlated strongly with the total algal biomass in October 2006. Its presence correlated strongly with ΔpH (R^2=0.62, $p<0.01$) and also correlated with ΔNH_3-N (R^2=0.50, $p<0.05$) and ΔO_2 (R^2=0.55, $p<0.01$). *C. hirundinella* preferably dominated reservoirs with high loads of nitrogen at the bottom (Van Ginkel et al. 2001) whereas *Microcystis spp.* (primarily *Microcystis robusta*), suspected to be one of the main contributors to algal taint in the Western Cape reservoirs, preferred waters with relatively little ammonia (as indicator of nitrogen levels), but high total phosphate levels at the bottom (>1 mg/L TP).

8.3.4 Intercorrelations (reservoir morphometry, water quality, production and production problems)

The correlation analysis among reservoir information, fish production data, water quality data and problem observation data revealed little connection between the fish production data and the water quality information, good linkage between reservoir information and fish production data as well as between reservoir information and production problems. The production problems could furthermore be linked with single water quality parameters and single fish production parameters.

Among the water quality parameters, some predictive power for production performance could be attributed to ammonia nitrogen which indicated a higher mortality rate with higher ammonia levels (R^2=0.52, p<0.05). The surface total phosphorus levels correlated negatively with the fish weight at harvest time (R^2=-0.44, p<0.05) and the growth rate correlated negatively with the presence of *Ceratium hirundinella* (R^2=-0.66, p<0.01).

The reservoir surface area was not only the most readily measurable morphometric descriptor, but also had most correlations with other parameters as compared to reservoir maximum depth and volume. The surface area correlated with all production related information except the food conversion ratio. Larger reservoirs therefore favoured faster growth rates (R^2=0.43, p<0.05), larger fish weight at harvest time (R^2=0.44, p<0.05) and lower mortalities (R^2=-0.44, p<0.05).

At the water quality level, the reservoir surface area was a predictor of total phosphorus abundance in the near-bottom water (R^2=-0.63, p<0.01). The reservoir surface area also correlated strongly with the presence of oxygen related problems in the epilimnion (R^2=-0.55, p<0.01). Another factor to predict oxygen related problems at the production sites was ΔNH_3-N which correlated strongly with oxygen depletion in the surface water (R^2=0.55, p<0.01).

The algal taint, however, only showed correlations with the date of harvest (R^2=0.46, p<0.05). Ranking the data according to harvest time, it became detectable that all occurrences of tainted fish occurred when the harvest took place later than the 10th of November 2006. There were only three exceptions that were harvested in November without taint, all of which still had a water temperature of around 19 °C at the end of October 2006 and less than 20.5 °C at harvest.

8.4 Discussion

8.4.1 Reservoir hydrodynamics, nutrient distribution and impacts on water uses

The pattern of monomictic winter holomixis in Western Cape small water bodies was confirmed (Robarts et al. 1982). Therefore, the turnover period of Western Cape reservoirs took place in winter and correlated with the season for trout production which is dictated by the temperature requirements of the species. Due to water mixture, the nutrient levels of the epilimnion will be at their highest during fish production with consequently the highest algal biomasses and the highest risk of oxygen content fluctuations caused by phytoplankton biomass changes. Reservoirs with nutrient enriched sediments (agricultural runoff, leaf litter, tree stumps, aquaculture) are more likely to encounter internal loading during turnover with consequent high algal biomass development (nocturnal oxygen consumption) and die-offs (sudden oxygen depletion by bacterial decomposition).

During the stagnation phase in summer, highest nutrient contents were observed in the hypolimnion. Most hypolimnia of the studied reservoirs became anoxic during stagnation with accumulation of hydrogen sulphide and ammonia,. The reservoirs with highest nutrient levels showed high phytoplankton abundance also in summer when epilimnetic nutrient levels reduced relative to the winter period. At 36 % sites, the maximum depth reduced during summer by water extraction for irrigation to 5 m or less and no division in epi- and hypolimnion takes place. However, they maintained anoxic bottom water conditions throughout the summer despite the absence of a thermocline.

Water quality of water extracted from the hypolimnion (especially in drinking water reservoirs) can deteriorate extremely fast with external nutrient enrichment such as net-cage aquaculture itself. The main season for water extraction for irrigation is during the dry summer months when the water bodies are primarily divided into an epilimnetic and hypolimnetic layer.

8.4.2 Production performance

80 % of the production systems achieved FCRs better than 2.0 with half the projects performing satisfyingly by achieving a FCR of 1.5 or lower. A FCR between 1.1 and 1.5 should be the committed target with the high quality feeds that were applied to minimise nutrient inputs by unused feed. Nutrient (mainly nitrogen and phosphorus) losses to the environment are 2.5 times higher at a FCR of 2.0 than with a FCR of 1.0 (Beveridge 1996). Better feed management, indicated by a FCR approaching 1.0 in trout farming, did correlate positively with maximum fish size which weakens the idea of a “the more food the better”-approach. These results suggest that overall farming management (including feed management, net cleaning, food quality controls etc.) is one of the main factors deciding production performance next to water quality. A study by Snijders (2006) at 8 of the 21 production sites revealed that feeding often took place with fixed amounts rather than an ad lib strategy when feeding is adapted to weather or visibility conditions.

Single water quality parameters did not correlate with FCR, growth rate or maximum fish size. However, in cases of surface oxygen depletion, these events had a considerable effect on the overall mortality rates.

Market price is greatly dependent on the absence of algal taint. Algal taint is caused by cyanobacteria that release substances known as MIB and geosmin. Most abundant cyanobacteria species was *Microcystis robusta*. Linkages between *Microcystis spp*. occurrence end of October 2006 and algal taint could not be verified. With the limited algal information fed into the computation of this study, it was therefore not possible to predict algal taint by other factors than harvest time. The main factors of cyanobacterial presence in the study sites appear to be warmer temperatures (> 20 °C).

8.4.3 Indicators to avoid production losses

Surface area is a strong predictor of production success in terms of occurrence of surface oxygen depletion. There was a strong correlation between surface area and oxygen problems (R^2=-0.55, p<0.01). The dividing line between presence and absence of oxygen problems would be a minimum size of 3 ha (1700 fish ha^{-1}). Oxygen problems did not occur with reservoirs with a surface larger than 3 ha, but this observation does not consider accumulative effects and long-term developments, but only the original nutrient status. An Australian report by Gooley and Gavine (2003) takes a more cautious approach and recommends an environmental stocking density of a maximum of 375 fish/ha. Those recommendations probably follow the precautionary principle approach and would entail that a minimum surface area of 13 ha would be advisable for the small-scale farmers programme unit size of 5 tons production.

A second indicator for likely surface oxygen reduction (by strong correlation) would be hypolimnion ammonia nitrogen in the stagnation phase (R^2=0.55, p<0.01). Reservoirs without problems had less than 1 mg/L NH_3-N in the near-bottom water measured during the stagnation phase in summer (November and February). When a site has been chosen, this could be an predictor for long-term monitoring of changes in water quality and changes in the risk for oxygen depletion, together with surface total phosphorus in the stagnation phase (<0.4 mg/L).

Algal taint in *O. mykiss* net-cage production can according to the correlation analyses be avoided by harvesting the fish before November (10^{th} of November in 2006) with only exception of reservoirs that are cooler than 19 °C surface water temperature (2 m depth) in early November. Application of that rule would have allowed avoidance of 83 % of the occurrences of fish with algal taint experienced in 2006.

A second factor reducing algal taint is a general algal biomass that does not exceed 5 g/L, measured in mid-winter. Reservoirs with greater total biomass were usually dominated by *Ceratium hirundinella* during the cold season which is why presence of that species during periods of temperatures lower than 19 °C could be used as readily available indicator for overall algal biomass.

8.5 Conclusions

8.5.1 Site selection

Site selection criteria for water quality aspects of production should strictly orient themselves on minimum surface area (depending on level of cautiousness applied) and nutrient accumulation in the hypolimnion of the reservoir during stagnation (November to February). Smaller reservoirs could still be suitable for trout production, but would then need to be managed like production ponds with means for aeration and options for waste removal respectively sediment exchange (Sedgwick, 1995). Site selection and initially good water quality will also support a sufficient pigmentation of the fish flesh.

These recommendations allow risk minimized trout production in the first production years, but do not include or guarantee the long-term sustainability of these operations, the security that primary reservoir use can be maintained and the security of reservoir self-recovery.

This study was able to stress the importance of initial site selection. By readily available indicators such as reservoir size, but also more comprehensive information such as nutrient levels in the hypolimnion of the stratified reservoir, one can reduce the predictable production risks such as high mortalities by oxygen depletion.

8.5.2 Avoidance strategies on managerial level

The most feasible criterion to avoid economic losses by algal taint seems to be early harvesting and the finding of sites with cool water conditions.

Feeding management is the most important factor for production performance next to good ambient water quality. Good feeding management helps greatly to maintain favourable water quality conditions.

Continuous monitoring of sites as early warning system of water quality deterioration should be implemented. Hypolimnetic water conditions during the stagnation phase (oxygen, ammonia, total phosphorus) give the best indications on overall organic loading of a reservoir, the greatest risk with net-cage aquaculture operations, and likely inhibitor of long-term success.

8.6 References

Berg R., Thompson R., Little P.R., Görgens A.H. (1994). Evaluation of farm dam area-height-capacity relationships required for basin-scale hydrological catchment modelling. Water SA 20: 265-272.

Beutel M.W. (2001). Oxygen consumption and ammonia accumulation in the hypolimnion of Walker Lake, Nevada. Hydrobiologia 466: 107-117.

Beveridge M.C.M. (1996) Cage aquaculture. Oxford: Fishing News Books.

FAO (2007). State of World Fisheries and Aquaculture 2006. Rome, FAO.

Gooley G. J. and Gavine F. M (2003). Integrated Agri-Aquaculture Systems: A Resource Handbook for Australian Industry Development. RIRDC Publication No 03/012; RIRDC Project No. MFR-2A. Kingston, Rural Industries Research and Development Corporation.

Hoffman L.C., Swart J.J., Brink D. (2000). The 1998 production and status of aquaculture in South Africa. Water SA 26: 133-135.

Jones G.B., Simon B.W., Horsley R.W. (1982). Microbial sources of ammonia in freshwater lake sediments. Journal of General Microbiology 128: 2823-2831.

Jüttner F., Watson S.B. (2007). Biochemical and Ecological Control of Geosmin and 2-Methylisoborneol in Source-Waters. Applied and Environmental Microbiology 73: 4395-4406.

Robarts R.D., Ashton P.J., Thornton J.A., Taussig H.J., Sephton L.M. (1982). Overturn in a hypertrophic, warm, monomictic impoundment (Hartbeespoort Dam, South Africa). Hydrobiologia 97: 209-224.

Sedgwick S.D. (1995). Trout farming handbook. Oxford: Fishing News Books (Blackwell Science Ltd).

Snijders K. (2006). Establishment of on-farm code for water quality management for aquaculture systems. Limburg, Provinciale Hogeschool Limburg, Belgium, Departement Biotechnologie.

Utermöhl, H. (1958). Zur Vervollkommnung der quantitativen Phytoplankton-Methodik. Mitteilungen der internationalen Vereinigung der theoretischen und angewandten Limnologie 5: 567-596.

Van Ginkel C.E., Hohls B.C., Vermaak E. (2001). A Ceratium hirundinella (O.F. Müller) bloom in Hartbeespoort Dam, South Africa. Water SA 2: 269-276.

CHAPTER 9 OVERALL CONCLUSIONS - IMPACT ASSESSMENT OF NET CAGE OPERATIONS OF RAINBOW TROUT (ONCORHYNCHUS MYKISS) IN WESTERN CAPE RESERVOIRS

9.1 Background

Rainbow trout were historically introduced into Western Cape rivers in the late 19th century (Kingfisher 1922). Next to sport fishing activities, commercial production has since developed and increased, primarily in large reservoirs of KwaZulu-Natal, the Eastern Cape, the Northern Province, as well as the Western Cape Province (Stubbs 2007). Additionally, operations in private irrigation reservoirs commenced, with unpredictable success at first. In a world-wide comparison of the common factors for successful trout production, good water quality, cool water temperatures and the high flow of water through production systems (primarily ponds and raceways) are the most predominant (Pillay 2004). With cage aquaculture in reservoirs and lakes, water quality problems are a common occurrence (Podemski and Blanchfield 2006) and water quality related and other problems were also observed in many small irrigation reservoirs in the Western Cape (Table 9.1).

Table 9.1: Reasons for the temporary or permanent abandonment of 59 % of all production sites (i.e. 22 of 37 sites) from 1993 to 2007.

Farm sold	1
Theft	1
WQ: High Total suspended solids	3
WQ: Oxygen/algal growth	7
WQ: Water colour	1
WQ: Reduced fish growth/taint	6
WQ: Interference with other uses	2
Unknown	1

The Hands-On Fish Farmer's Co-operative Limited, harbouring most small-scale operations in the Western Cape, does not only produce fish, but attends to the pressing need of socio-economic growth (farm workers operate production in the small irrigation reservoirs) and ideally, promotes multiple water use of the numerous open water storage systems of the Western Cape (Salie 2004). Although faced with many challenges, some of the reservoirs with a history of continuous production success have kept the project and its vision alive.

The total number of HandsOn projects and the history of these projects is presented in Figure 9.1. The production history is reflected by the intensity of colour, white being new projects and black being the projects operating in the 4th year or longer. There was continuous growth and success from 1996 to 2004, where sites with consistently successful production were found and established. Since 2005, the number of sites used for more than four years has not increased as predicted, and many projects were abandoned after one or two production cycles. This indicated an erratic site selection process instead of rational choices based on intuitively gained experiences and available research results.

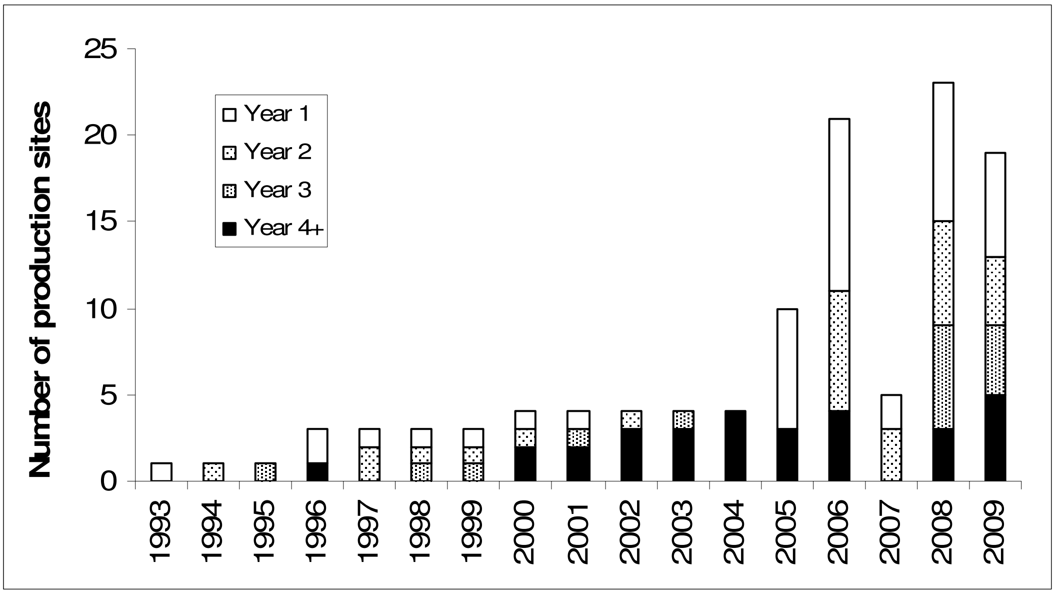


Figure 9.1: Number of production sites and their production history. 2007 was an exceptional year (fingerling supply problem). After gap years of no production, previous production years were added when production was resumed.

Due to water temperatures and given water quality of Western Cape reservoirs, grow-out of trout is only feasible during the winter months. However, during these months water temperatures are below the optimum growth temperature for trout. Trials undertaken in Australia recommended an environmental stocking density limit of 375 kg fish per ha surface area with small reservoirs (which relates to a minimum surface area of 13 ha with a 5 t production). With the Hands-On Fish Farmer's Co-operative Limited this limit was clearly exceeded with most projects located in reservoirs of <10 ha.

The aim of the current study was to determine how current net cage practice impacts on water quality of reservoirs in the Western Cape and to answer the question as to whether cage aquaculture is a suitable method for expanding usage of Western Cape irrigation reservoirs, especially seeing that they are already impacted upon by agricultural activities. Furthermore, to determine under which conditions the effects of production are compatible with international and South African Guidelines propagating sustainable production (ecologically, but consequently economically and socially).

The conclusions are divided into two sections. The first section focuses on conservation and the possible impact of aquaculture on small reservoirs. The second section focuses on conditions for optimised sustainable production, with minimum impacts.

9.2 Reservoir ecology and ecological function

9.2.1 Present South African legislation

The relevant recommendations and regulations related to water quality and aquaculture in open water systems can be found in the South African Water Quality Guidelines (DWAF 1996a, DWAF 1996b), the National Biodiversity Bill (DEAT 2004) and the Code of Conduct for Responsible Fisheries – Aquaculture Development (FAO 1995).

The Water Quality Guideline for Aquatic Ecosystems (DWAF 1996a) recommends, with regards to inorganic phosphorus as target; (i) a water quality range where the concentration of inorganic phosphorus should not be altered by >15 % (from concentrations under unimpacted conditions) at any time of the year, (ii) the trophic status should not increase (i.e. from eutrophic with 25-250 µg/L P to hypertrophic >250 µg/L P) and (iii) the amplitude and frequency of natural cycles in P concentrations should not be changed. While previous conditions of the reservoirs could not be compared to conditions with aquaculture production, the average reservoir without fish production could, however, be compared to the average production site, where the average surface total phosphorus concentrations increased by 112 % (with similar or slightly less increases regarding inorganic phosphorus). The amplitude of natural P cycles was therefore disturbed, while the frequency remained similar. The South African Water Quality Guideline regarding agricultural water use by aquaculture (DWAF 1996b), states that the acclaimed policy is:… the maintenance of all water resources in their current condition with the low effect range (< 15% increase in inorganic P) as target water quality range for South African waters.

The Biodiversity Bill (DEAT 2004) regulates the introduction of alien species (such as rainbow trout) into different environments and permits are required to operate trout production. A permit holder is required to prevent species from escaping, which is a possibility of reservoirs feeding into rivers directly.

The Code of Conduct for Responsible Fisheries (FAO 1995) propagates sustainable aquaculture practices. The precautionary approach is recommended where the absence of scientific knowledge will rather underestimate than overestimate production potential. At this point, the simplified permit for private irrigation reservoirs is based on the assumption that a 5 t production of trout will be of low impact. However, the results of the current study show that the permitting process needs a sound evaluation of the initial water quality conditions in the respective reservoirs. Governmental assessment and monitoring aiming to set criteria for suitable reservoirs for cage production are lacking and would be necessary as well as the determination of maximum environmental stocking densities. Surface water, but also sediment integrity, will also need to be respected.

9.2.2 The suitability of Western Cape reservoirs for trout-cage production (Chapter 3)

Parameters that influence successful fish production include optimal ranges of temperature, pH, oxygen and hardness, and the absence of free ammonia. As published by Maleri et al. (2008), the optimum temperatures for rainbow trout (*Oncorhynchus mykiss*) leave a suitable grow-out period for trout form April to October where the surface water temperatures of the reservoirs range within the 14 to 21 °C window. The surface water pH during these winter months is suitable, similar and related to the occurrence of free ammonia which remains below toxic concentrations for rainbow trout during winter. The softness (absence of carbonate and bicarbonate) of Western Cape reservoirs is below the recommended value (2 to 55 mg $CaCO3/L$ instead of the recommended >200 mg $CaCO3/L$), but has no immediate negative effects. Soft water increases the risk of diseases and allows more sudden water quality changes regarding pH and ammonia concentrations than hard water would (reduced buffer capacity).

The oxygen concentrations in the reservoirs were lower than 6.0 mg/L on occasion during the study period, mostly in small reservoirs (<5 ha) of hypertrophic status; problems related to oxygen occurred in 29% of 21 reservoirs during the study period. The reduction in oxygen was related to excessive algal growth.

The initial evaluation of water quality status of most reservoirs shows a suitability for trout-cage production, but often excludes the seasonal changes in phytoplankton presence and surface oxygen values. A safe way to monitor the suitability of a reservoir would be to measure oxygen and sample phytoplankton on a monthly basis in the core production period from May to October one year prior to production commencement.

The period of introduction of additional phosphorus by agriculture (via runoff) and aquaculture (directly into the water body) falls into the winter period, which is also the turnover period of these monomictic reservoirs. Both the latter cases enhance occurrences of phytoplankton peaks during the winter period (dependent on the quantities and geomorphological and hydrological conditions), rather than during the two intermediate periods between turnover and stagnation period. As Chapters 4 and 6 show, the effects caused by aquaculture in the first and second production winters already exceeded the “background” effects caused primarily by runoff from agricultural land, which indicates that cage aquaculture affects itself negatively when operated in unsuitable sites.

9.2.3 Conclusions of effects of aquaculture found (Chapters 4-6)

Eutrophication in Western Cape reservoirs leads to elevated surface phosphorus concentrations, extended hypolimnetic anoxia, higher fluctuations in surface pH, pH levels >9 and phytoplankton communities dominated by single species that are larger in diameter (>100 µm). These developments in the reservoir are independent of the source of nutrients into the reservoir and the input rate (addition of nutrients per year).

The main drivers of eutrophication in the studied reservoirs without aquaculture are agriculture and sometimes informal settlements. Trees and birds were not shown to have impacts on parameters in the water, but increased total phosphorus in the sediments. Net-cage aquaculture in a high number of reservoirs (84%) was shown to have impacts that match the impacts of other external sources and exceeds them in specifically unsuitable sites (e.g. <3 ha in surface area). When considering that agricultural land use is already present in the area (and can certainly be optimised in minimising nutrient losses), the introduction of net-cage agriculture needs to consider that the combined input should not exceed the environmental capacity of nutrient introduction into the reservoirs.

With high likeliness, the increased sedimentation of aquaculture waste and anoxic conditions will impact on the macrozoobenthos population. The sediment composition changes towards a higher organic proportion with aquaculture waste, which could have positive effects on sediment processes and binding capacities. This needs to be investigated further since negative effects on the resuspension rate occur at the same time. There also seems to be impacts of aquaculture on bacterial presence in surface water (i.e. a higher dissolved organic component), indicated by an increase in the BOD_5 value (biological oxygen demand in five days). In bottom water, there was seasonally no oxygen to be consumed to indicate aerobic microbial presence, however the general microbial activity in sediments of different reservoir use would be interesting.

On the phosphorus budget side, there was a clear indication that with some sites, the additional introduction of phosphorus into the reservoirs was high (adding 30 to 90 % to what is already introduced). Obviously, optimum conditions with optimum FCR, feed with the lowest feasible phosphorus content and good water quality conditions for trout (oxygen, temperature) support lowest effects, however four of the six studied sites were not found suitable for 5 tons production of trout.

Eutrophication of the reservoirs with the described changes in water quality has adverse effects to irrigation systems (filtering), drinking water quality (filtering and odour) as well as aquaculture itself (oxygen supply to fish and algal taint in produced trout flesh).

9.2.4 Thresholds for production success and stocking limits (Chapters 6-8)

When trying to find optimal reservoirs for trout production, several angles were used to find optimal conditions or thresholds to optimise conditions respectively. Are reservoirs outside these conditions chosen, the risks for surface oxygen depletion, algal taint in harvested fish flesh, increased mortalities or problems to the other reservoir uses will increase.

With regards to phosphorus budgets, two reservoirs with surface areas of 6.5 and 7.3 ha were successful sites, with water exchange rates of approx. 7 to 9 times per year, and this holds especially true with suboptimal FCRs of >1.1 and fish feed of >1 % phosphorus content. Reservoirs with lower exchange rates are prone to increase phosphorus concentrations in sediment and surface water with the sufficiently described consequences. Reservoirs >10 ha and water exchange rates of >5 times per year are the recommended minimum criterion for a sustainable production site.

When evaluating the criterion of “no algal taint in harvested fish” to identify successful production sites, the depth of the reservoirs as well as the hypolimnetic water temperature played an important role. As a consequence, reservoirs should have a deepwater zone of >9 m depth and hypolimnetic temperatures of <17℃ during production. The findings suggest an advantage of granite based reservoir basins, sites of initially low surface phosphorus concentrations and diverse phytoplankton communities.

When using production history and occurrence of algal taint and surface water oxygen depletion as criterion, the recommended minimum surface area would be >3 ha to avoid mortality incidents by oxygen depletion in the studied reservoirs. Well suited sites were identified as low in hypolimnetic ammonia during the stagnation phase (January and February) with less than 1 mg/L and an algal biomass in winter (April to August) that should not exceed 5 g/L.

9.2.5 Environmental impact assessment of trout net cages in small reservoirs

The general water quality status of the studied Western Cape reservoirs is eutrophic to hypertrophic and reflects enrichment in lowland rivers (Hart & Hart 2006). Approximately 20 % of the reservoirs harbour conditions that would be an immediate threat to natural or stocked fish or water bird health (pH and toxic ammonia concentrations, mostly in summer). Harmful algal blooms were not observed. Greater bird numbers, with a diversity >10 species, were specifically sighted at reservoirs with an extended flat water zone. Flat reservoir shores are naturally rare due to the common practice of minimising the reservoir surface area to minimise evaporation losses. Aquaculture did not result in water quality conditions which affected bird health directly, but on the contrary, the cage structures served as additional breeding areas during the summer months, and attracted fish eating birds (cormorants) during winter and spring. According to Davies (1997), reservoirs with greater habitat variability, better water quality, less turbidity and lower pesticide levels were more attractive to birds in the Elgin area (similar to current study area). The two reservoirs with the most birds present in the current study were of the lowest trophic state found (which would have been among the better water quality reservoirs of the study by Davies (1997) when comparing nitrate concentrations). Davies studied primarily sites <1 ha in surface area where reservoirs are naturally more turbid, with a higher concentration of nutrients and pesticides. With respect to water birds, aquaculture does not seem to have an adverse effect on mean numbers and species diversity.

Other impacts of aquaculture that need to be considered are the introduction of resistant bacteria. Antibiotic treatment of fingerlings in hatcheries was minimised and not generally applied. The quality of wood, i.e. pre-treatment, used for cage construction could be of concern due to the introduction of harmful substances (e.g. antifouling agents) and their absence should be verified. The cage netting could lead to bird entanglement, although this was only observed once. The escape of trout into receiving waters was not studied, but certainly needs further attention. The catchments where rainbow trout permits are granted, usually have a history of rainbow trout stocking in the respective natural rivers. Whether these feral alien species are likely to increase further under current production practices, should be monitored.

Increased eutrophication by external sources (agriculture and aquaculture in different paces depending on the reservoir characteristics) affected reservoir uses. An increased need to backwash filters was often stated by farm owners and workers and was coinciding with increased nutrient status of the respective reservoirs. In one example, drinking water quality was directly affected by aquaculture operations when hypolimnetic water extraction was applied, presumably aquacultural waste spoiled the water (unpleasant odour). As a general consequence to eutrophication, a change of water extraction location to the surface becomes necessary.

In conclusion, sustainable production of rainbow trout in cages was shown to be possible at suitable sites and can suffice the ”low impact” rule (<15 % phosphorus increase) under very specific conditions only. Production should therefore be re-evaluated in all other circumstances. Freshwater is a precious and diminishing resource in the Western Cape which is already under pressure in its current state. Safe drinking water and irrigation water use are seen as priority to reservoirs that were constructed to serve these purposes. Agriculture is already affecting many of the lowland sites and evaluation of agricultural practices with regards to minimised nutrient losses is recommended. Net-cage aquaculture should only be operated when the joint combination of uses and practices (including aquaculture) allow that the nutrient increase in the reservoir remains below the indicated boundaries.

9.2.6 Overall impact analysis regarding reservoir ecology and ecological function

Open water storage of rain and rainfall fed river water in reservoirs of the Western Cape represents the greatest source of fresh water during the dry season. Large reservoirs are primarily used for industrial and domestic use, while small private farm reservoirs are used for irrigation. They are usually <15 ha in surface area. Despite being man-made, small farm reservoirs form an integral part of agricultural landscapes, but are, however, not recognised as reservoirs of biodiversity (Brainwood & Burgin 2009). Natural inland freshwater lakes are no common feature in the Western Cape, but rivers and coastal wetland systems are (Dalgliesh et al. 2004), where farm reservoirs can serve as stepping stones between these systems.

The influence and role of irrigation reservoirs in the hydrological landscape is still primarily unknown (Hart & Hart 2006), and so is the ecological state of many farm reservoirs (Brainwood et al. 2004). Fish, macroinvertebrates and water birds are the groups that could profit most obviously from additional water bodies, however, within certain limits. Birds prefer flat water zones in water bodies, macrophyte rich waters and waters of medium to good water quality (Davies 1997). For example, Theewaterskloof Dam, a large drinking water reservoir for Cape Town, has no water bird specific value due to its low water quality caused by decaying tree stumps and reservoir immaturity (Hart & Hart 2006). Also, most indigenous fish are extremely water quality dependent and need structures to pass in-flow reservoirs.

Generally, open freshwater structures are poorly managed in South Africa and their individual independent status is not well understood (Hart & Hart 2006). Their study showed that the water quality of small farm reservoirs is developing towards highly eutrophic and hypertrophic conditions. Under the currently existing agricultural practices, however, in some reservoirs, 5 t production units of trout added the same amount of phoshorus and therefore doubled or tripled the already introduced phosphorus which accelerates eutrohpication. Hence, the additional pressure caused by aquaculture production needs sensible decisions and should be propagated under application of the precautionary principle. The question remains as to whether cage aquaculture of trout at the recommended scale of 5 t can be continued under certain circumstances or should it be discontinued entirely. The full decision making process, however, would require a balance between economic growth and water use efficiency versus protection of one of the most precious water resources in the country (irrigation water and drinking water).

9.3 Conclusions regarding responsible aquaculture production

Trout are dependent on good water quality and the Western Cape depends on good water quality resources. Site selection of suitable sites is one of the most important task before production starts. Successful sites can avoid production problems in terms of project disruption, additional efforts, adverse effects for primary reservoir uses as well as economical setbacks and social instability.

No negative impact of aquaculture production of 5 t units of trout could be found within two of six (33 %) of intensively studied production sites (monthly samples) and 4 of 26 (15 %) of seasonally studied sites (three samples per year). Therefore it can be concluded that sustainable trout production is possible in suitable Western Cape reservoirs. The characteristics of suitable sites were: a surface area of >6 ha with water exchange rates >6 times per year (Chapter 4 and 6) and >10 ha with water exchange rates <3 times per year (Chapter 4). A water exchange rate of <2 times per year is not recommended. To prevent interferences with surface irrigation water quality, the minimum depth (lowest water level in summer) of a suitable site should be >6 m to prevent the top sediment mixing with surface water or breaking stagnation. Shale and sandstone based basins or sediment surface material of a small grain size is more suitable to bind additional phosphorus waste from aquaculture production at a faster rate (Chapter 4). However, retention capacities depend more strongly on hypolimnetic water conditions than catchment rock types (Chapter 5). Greater surface areas will usually entail that the potentially hypolimnetic sediment area (>6 m depth) will also be increased so that the sediment area unimpacted by agricultural waste will exceed (at least twice) the directly impacted area. Optimal conditions for trout production and optimal conditions for ecosystem well-being are closely linked due to the high demand of good water quality conditions for trout.

As an additional tool for estimating the suitability of sites, a carrying capacity model as introduced by Beveridge (1984) yielded good results that were similar to the outcome of a data intensive mass balance approach. Water exchange rate and reservoir volume were again the most important components feeding the calculation, and they need to meet certain minimum standards.

There may be health hazards linked to fish consumption due to the water quality situation in certain areas. The study by Davies (1997) showed that all but one of the 28 Elgin sites had critical effect values of Endosulfan, with 14 sites actually exceeding the acute effect value. In general, the introduction of Endosulfan and similar substances should be banned or strictly controlled. With regards to this study, Endosulfan is an insecticide which is most toxic to fish and accumulation of Endosulfan in trout flesh would be of concern to the environment and to consumers of these fish. Knowledge of average concentrations prior to production would be recommended, and the same would be valid for any applied chemical.

To minimise the risk of tainted trout flesh which reduces marketability of the fresh product, criteria similar to sustainable successful production apply (Chapter 7). Algal species which cause algal taint occur with elevated phosphorus concentrations. The four successful production sites had average annual surface total phosphorus concentrations of 0.047, 0.114, 0.118 and 0.137 µg/L, which exceeded the recommended maximum surface phosphorus. However, overall phytoplankton biomass was low in reservoirs of higher exchange rates. Reservoirs which exceed a minimum depth of 6 to 7 m until late summer remained below the triggering temperature for cyst remobilisation. When no history of algal taint producing species is known, the likeliness of taint is decreased. Granite dominated basins and catchment areas are more suitable (possibly due to more unfavourable conditions for cyst survival in the sediment). A high Total nitrogen :Total phosphorus ratio can also indicate a reduced chance of cyanophyte presence.

Once suitable sites are found, good management practices are required. A training manual was published parallel with trout farming development (Salie et al. 2008). Further recommendations and comments with regard to common management practices will follow in the next section.

When water quality deteriorates, recovery in most cases is a lengthy process. The additional nutrient input of aquaculture, which was shown to impact on certain systems (2-3 ha), augments the sediment phosphorus resources and, depending on the overall nutrient balance, takes years to decrease to pre-production levels (see Reservoirs 25, 26 and 31). With the reduction of external nutrient loading, internal loading takes over and maintains an elevated phosphorus seasonal circulation. Mitigating measures and measures of phosphorus removal are either cost-intensive or have not been proven to bring sufficient success. For instance, a trial with a bacterial kit provided by Amitek Solutions (Pty) Ltd. was unsuccessful in removing nutrients from enriched hypolimnetic water. Despite successes of the product with effluent and waste water treatment, the suitability of bacterial kits for nutrient reduction in irrigation reservoirs depends completely on concurrent oxygenation of the bottom area and artificial destratification. Nitrogen could then be successfully removed from the systems while phosphorus would remain. Phosphorus removal could otherwise be achieved e.g. with hypolimnetic water removal, sediment dredging or introduction of phosphorus binding substances. Consequently, production systems that consider water enrichment in combination with phosphorus removal from the systems are no cost-effective options at this point.

9.4 Recommendations – management options

Dedicated site selection is the management choice with the greatest impact on the long-term success of trout cage production with the minimum environmental impact. Sufficient information on optimal sites should be available, not only via information made available in the course of the current study, but also recommendations from sources such as the FAO, and experiences from Scotland, Canada, the USA, Australia, Iran and other countries (Chapter 2). The precautionary principle would imply that water quality deterioration (of which the consequences are paid for by private farm owners in the current production scenario and by the public in terms of overall quality of water resources) will be minimised after careful consideration of all available information sources and risks for water resources in general and the water use of the respective reservoir in particular.

When the most important water reservoir parameters (exchange rate, surface area and volume and minimum water depth >6 m during all seasons) are fulfilled at a site, other water quality issues that may hinder optimal trout production, might be rectified with improvements in the hydrological management of a reservoir (e.g. in cases were inflowing water carries a high sediment load). The phosphorus balance of reservoirs with ready structures for hypolimnetic water withdrawal (and sufficient water reserves in summer to spare a proportion of the water) could reduce phosphorus by extracting hypolimnetic water (stratification needs to remain established). Withdrawal for irrigation purposes would be the preferred path. In case of water structures receiving the hypolimnetic water, however, downstream water quality changes need to be considered. With surface water withdrawal, no effects of downstream water quality would be expected from the results of the model in Chapter 6. With hypolimnetic water withdrawal, effects on downstream waters would be anticipated, dependent on the phosphorus and nitrogen content (as well as parameters not considered in this thesis) of the water released.

Next to site selection, feed quality (with considerations of all available products on the market) and feeding management are the next most important stepping stones towards successful production (Papatryphon et al. 2004). Training courses for feeding management need to be in place and should be intensified with regular on-site controls. Interviews with small-scale farmers revealed that production practices often diverge from optimum feeding techniques (Snijders 2006). Also, net cleaning was often neglected for several months and dead fish were not removed from the nets.
Another option to improve FCRs might be the reduction of cage size to optimise production. The common stocking density in South African cages (5 t production units) does not exceed 8 kg/m^3. However, a stocking density of 20 to 80 kg/m^3 is common elsewhere and can usually be applied without negative effects on fish health and weight increase (North et al. 2006). It goes without saying that fish feed could be applied more effectively with smaller cage areas, keeping the same depth.

Good site selection, an early season introduction of trout in autumn (as soon as surface temperatures guarantee trout wellbeing) and an early harvest (before surface water temperatures increase >19 °C) are the safest options to avoid algal taint (Chapter 5).

Should problems with water quality arise (disease outbreak, surface oxygen depletion), an emergency plan needs to be in place that evaluates first steps and ready alternative options.

9.5 Future research

The following suggestions on future research are made as a consequence of the results and experiences of the current study:

- Investigation of sediment processes in terms of immobilising and remobilising phosphorus need to be investigated further. As a suggestion, bacterial processes might play an important role in phosphorus remobilisation.
- The sedimentation rate in general and additionally underneath cages would need to be investigated to further define the nature of waste arriving on the sediments. Even more important are phosphorus release experiments under different conditions (pH, anoxic/oxic), mimicking hypolimnetic conditions (e.g. bottle experiments).

- The safety of trout flesh produced in Western Cape irrigation reservoirs for consumption by humans (e.g. pesticide concentrations) needs to be verified.
- Features/structures/characteristics that increase the ecological importance of freshwater reservoirs as stepping stones in landscape ecology could support the value of initially man-made structures.
- Options for the optimisation of water flow management of irrigation reservoirs and improvement of overall nutrient budgets need to be investigated.
- Production options to remove additional phosphorus from farm reservoirs without further nutrient input could solve the currently encountered nutrient enrichment altogether.
- Improvement of cage structures – methods to catch waste or remove waste from the sediment (suction pumps/hypolimnetic water extraction with development of additional water use for the extracted water) - would be an alternative option to maintain the production method in Western Cape reservoirs.
- Catchment nutrient flows studying land use and its impact on nutrient availability to open water resources and studying runoff water quantity and quality would support the current findings and could decrease the overall ecological degradation of reservoirs regarding nutrient status and algal growth (this includes economical aspects regarding costs for filtering and other costs of irrigation systems).

9.6 References

Brainwood, M. A., Burgin, S., and Maheshwari, B. L. (2004). Temporal variations in water quality of farm dams: impacts of land use and water sources. Agricultural Water Management 70(2): 151-175.

Brainwood, M. A. and Burgin, S. (2009). Hotspots of biodiversity or homogeneous landscapes? Farm dams as biodiversity reserves in Australia. Biodiversity and Conservation 18: 3043-3052.

Dalgliesh, C., Steytler, N., and Breetzke, B. (2004). Western Cape - State of the Environment Overview Report. Report No. 329585/1. 2004. Pretoriar, Department of Environmental Affairs and Development Planning.

Davies, H. (1997). An Assessment of the Suitability of a Series of Western Cape Farm Dams as Waterbird Habitats. Cape Town, University of Cape Town, Conservation Biology Department.

DEAT (2004). Biodiversity Bill. South Africa.

DWAF (1996b). South African Water Quality Guidelines. Volume 6: Agricultural Water Use: Aquaculture. Pretoria, Department of Water Affairs and Forestry.

DWAF (1996a). South African Water Quality Guidelines. Volume 7: Aquatic Ecosystems. Pretoria, Department of Water Affairs and Forestry.

FAO (1995). Code of Conduct for Responsible Fisheries. Rome, FAO.

Hart, R. and Hart, R. C. (2006). Reservoirs and their management: a review of the literature since 1990. Water Research Commission Report No. KV 173/06. Pretoria, Department of Water Affairs and Forestry.

Kingfisher (1922). A Trout Fisher in South Africa. Cape Town: F.W. Flowers & Co.

Maleri, M., Du Plessis, D., and Salie, K. (2008). Assessment of the interaction between cage aquaculture and water quality in irrigation storage dams and canal systems. WRC Report No. 1461/1/08. Pretoria, Department of Water Affairs and Forestry.

North, B. P., Ellis, T., Turnbull, J. F., Davis, J., and Bromage, N. R. (2006). Stocking density practices of commercial UK rainbow trout farms. Aquaculture 259: 260-267.

Papatryphon, E., Petit, J., Kaushik, S. J., and van der Werf, H. M. (2004). Environmental impact assessment of salmonid feeds using Life Cycle Assessment (LCA). Ambio 33(6): 316-323.

Pillay, T. V. R. (2004). Aquaculture and the environment. Oxford, UK: Blackwell Publishing Ltd.

Podemski, C. L. and Blanchfield, P. J. (2006). Overview of the environmental impacts of Canadian freshwater aquaculture. A Scientific Review of the Potential Environmental Effects of Aquaculture in Aquatic Ecosystems - Volume 5. Canadian Technical Report of Fisheries and Aquatic Sciences. Ontario, Department of Fisheries and Oceans Canada.

Salie, K. (2004). Improving pre-harvest management for better post-harvest quality in small-scale rainbow trout farming in the Western Cape Province, South Africa. Unpublished report.

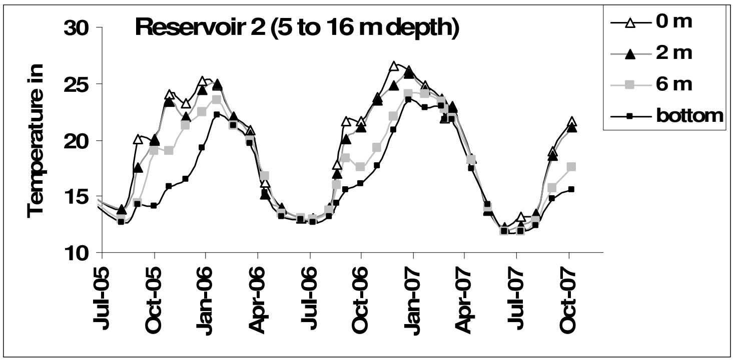
Salie, K., Resoort, D., Du Plessis, D., and Maleri, M. (2008). Training Manual for Small-Scale Rainbow Trout Farmers in Net Cage Systems on Irrigation Dams with Reference to Production, Fish Health and Water Quality. Water Research Commission Report TT 369/08. Pretoria, Department of Water Affairs and Forestry.

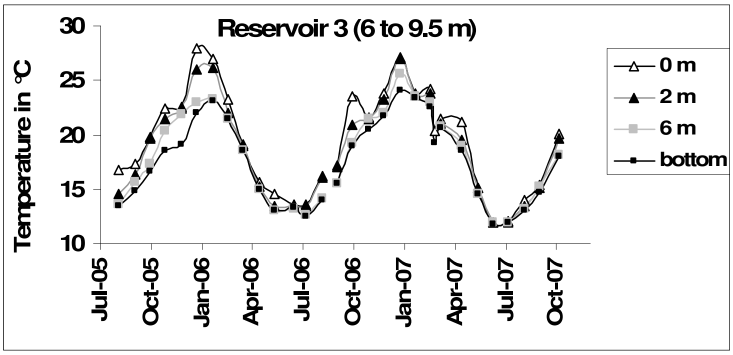
Snijders, K. (2006). Interview questions and answers collected in the process of the B.Sc. Thesis "Establishment of on-farm code for water quality management for aquaculture systems". Limburg, Provinciale Hogeschool Limburg, Belgium, Departement Biotechnologie.

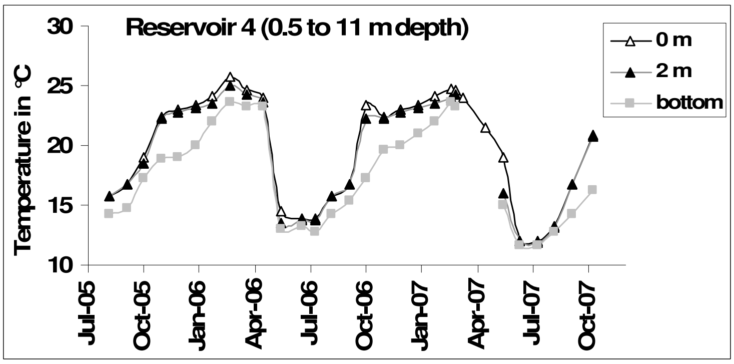
Stubbs, G. (2007). Production figures of rainbow trout in South Africa. Western Cape Trout Association Meeting, May 2007.

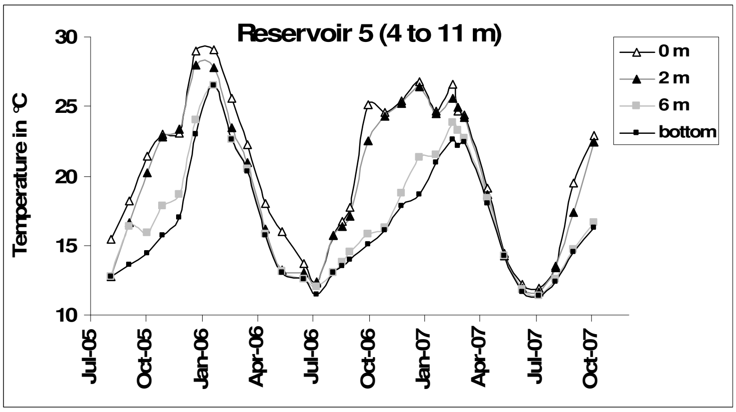
CHAPTER 10 APPENDIX

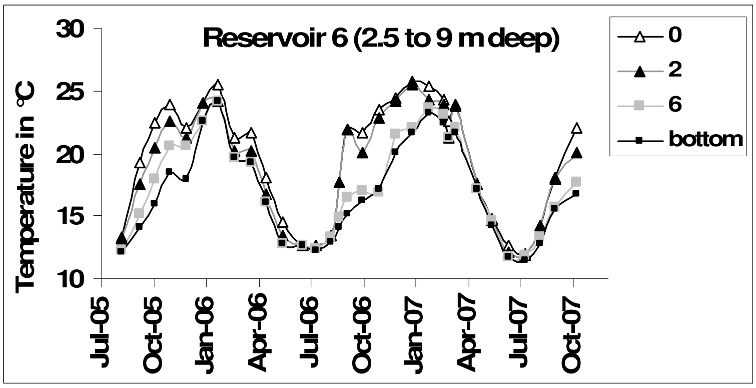
10.1 Water temperatures of Reservoirs 2 to 6 (referring to section 3.4.1)



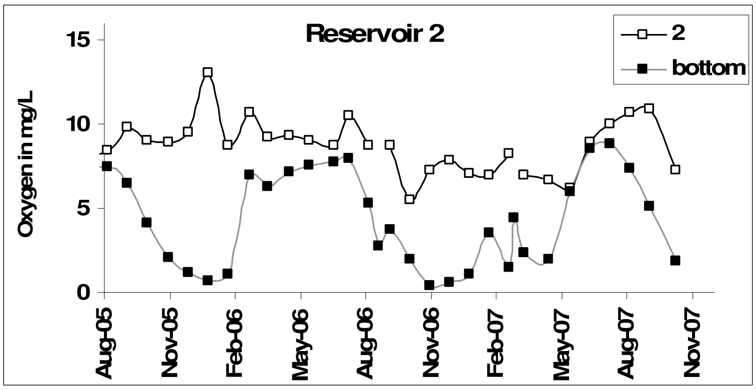


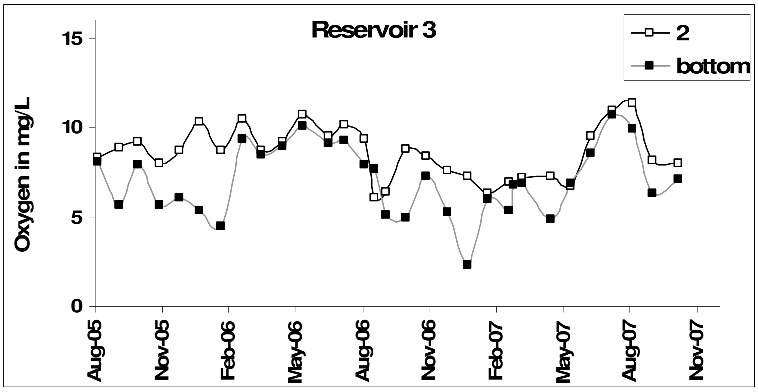


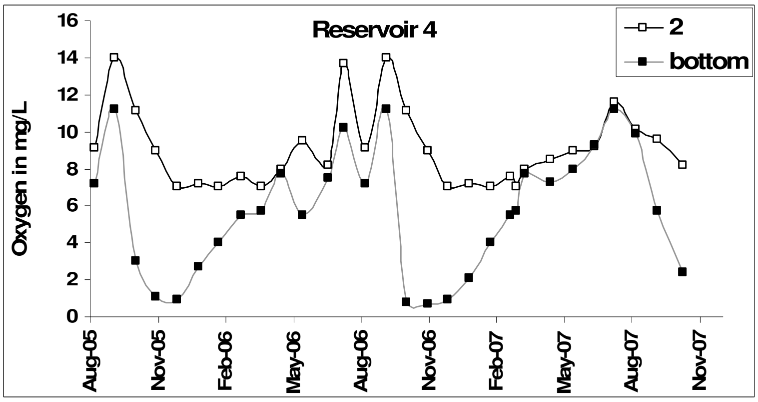


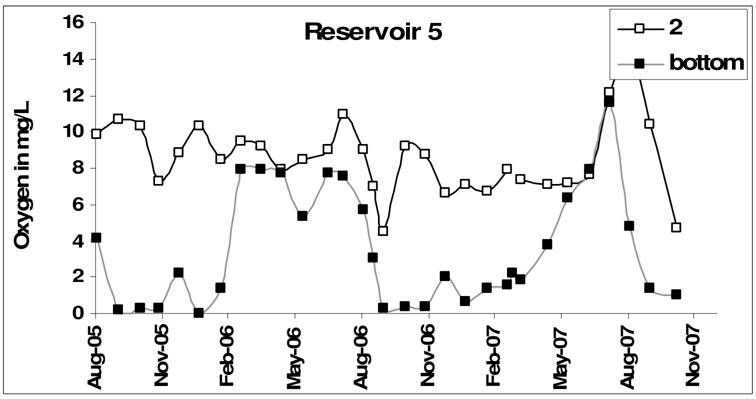

Reservoir 6 (2.5 to 9 m deep)
Temperature in °C
30
25
20
15
10
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07
0
2
6
bottom

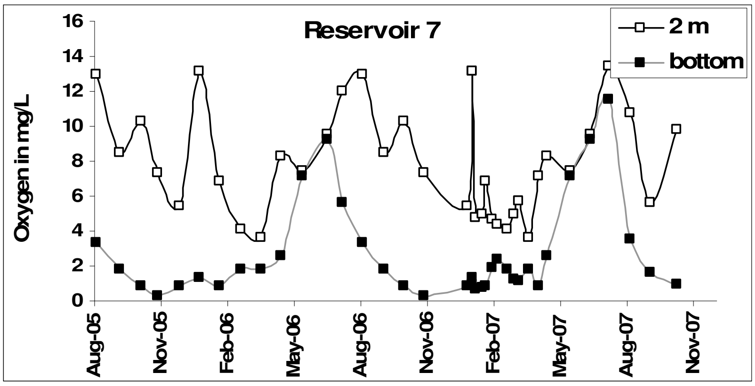
10.2 Oxygen distribution of Reservoirs 2 to 5 and 7 (referring to section 3.4.1)



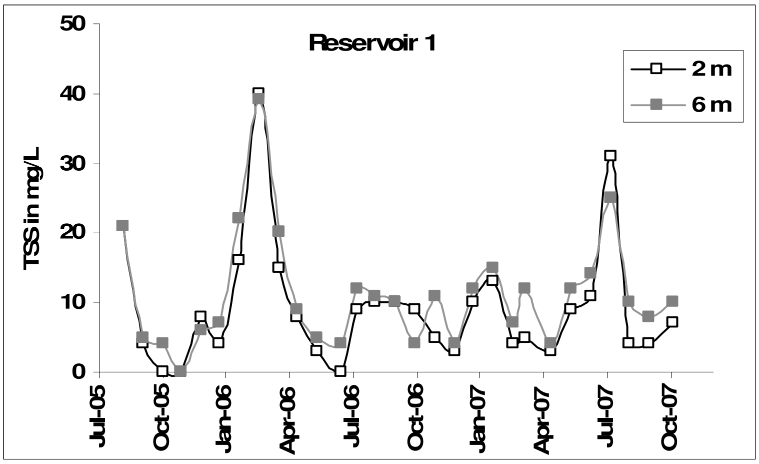


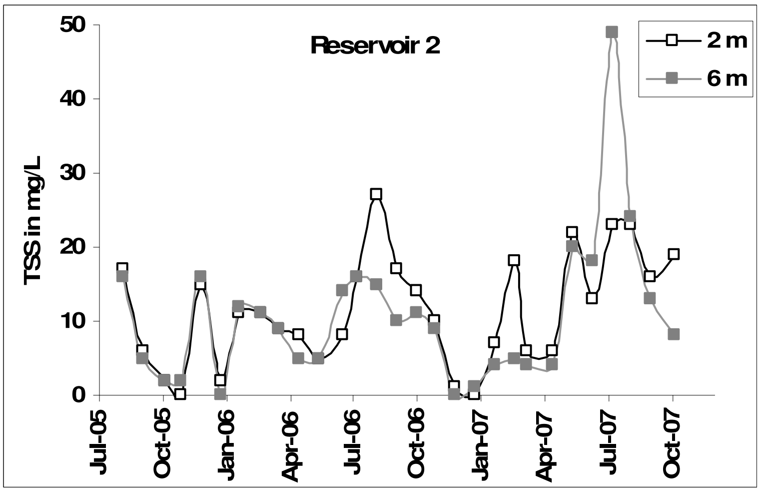


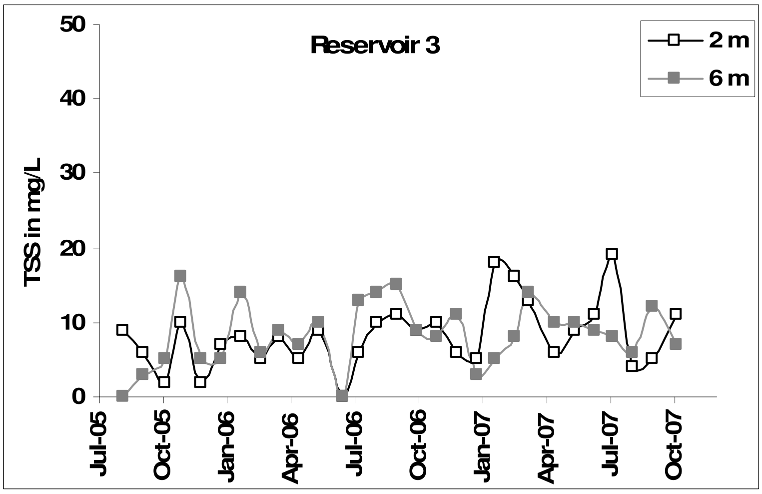


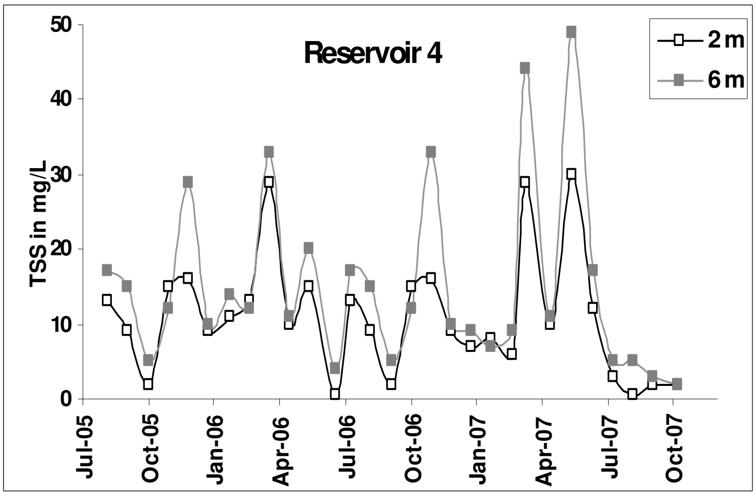

Reservoir 7
2 m
bottom
Oxygen in mg/L
16
14
12
10
8
6
4
2
0
Aug-05
Nov-05
Feb-06
May-06
Aug-06
Nov-06
Feb-07
May-07
Aug-07
Nov-07

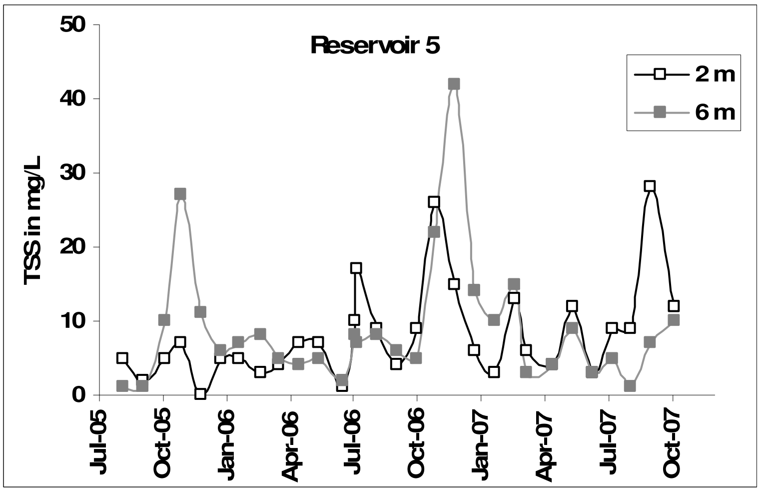
10.3 TSS (Total suspended solids) distribution of Reservoirs 1 to 7 (referring to section 3.4.2)

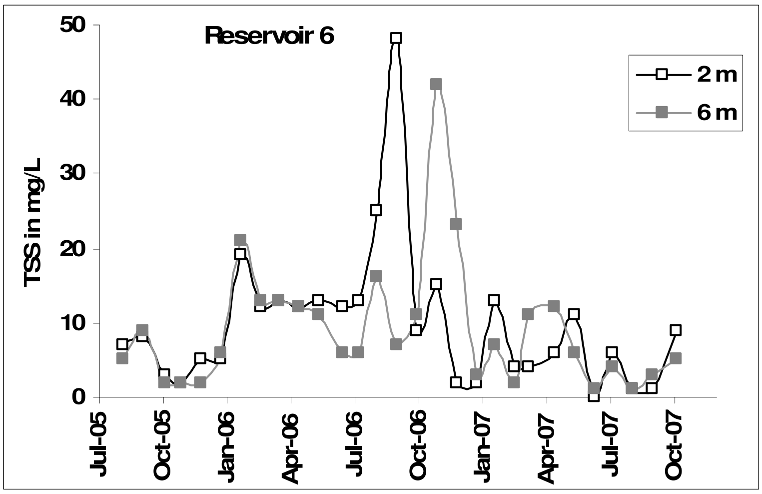


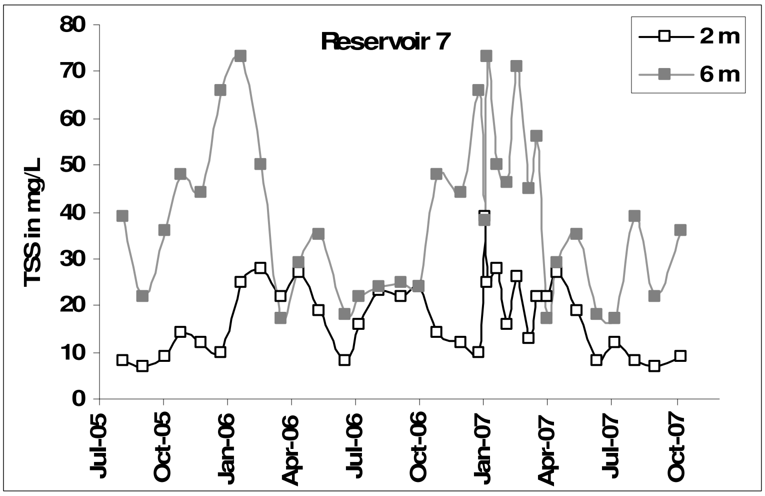




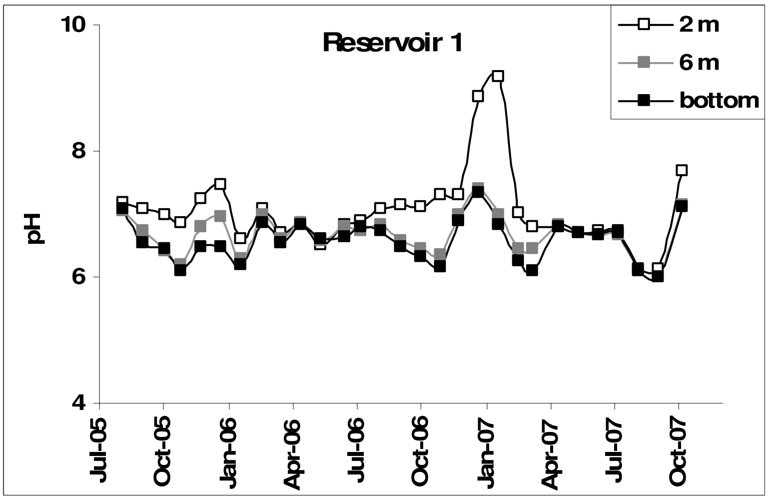

Reservoir 4
2 m
6 m
TSS in mg/L
0
10
20
30
40
50
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

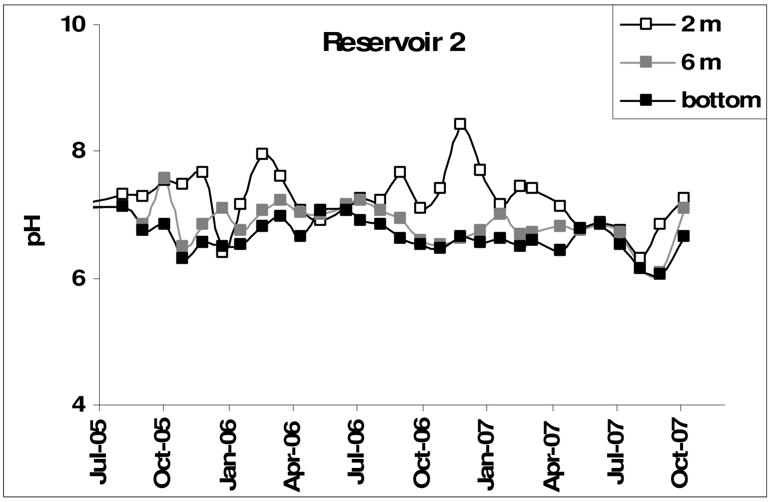

Reservoir 5
2 m
6 m
TSS in mg/L
0
10
20
30
40
50
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

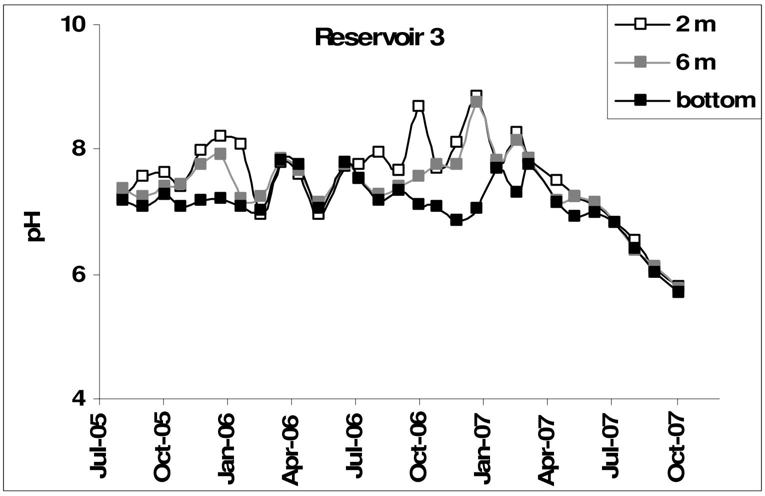

Reservoir 6
2 m
6 m
TSS in mg/L
0
10
20
30
40
50
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

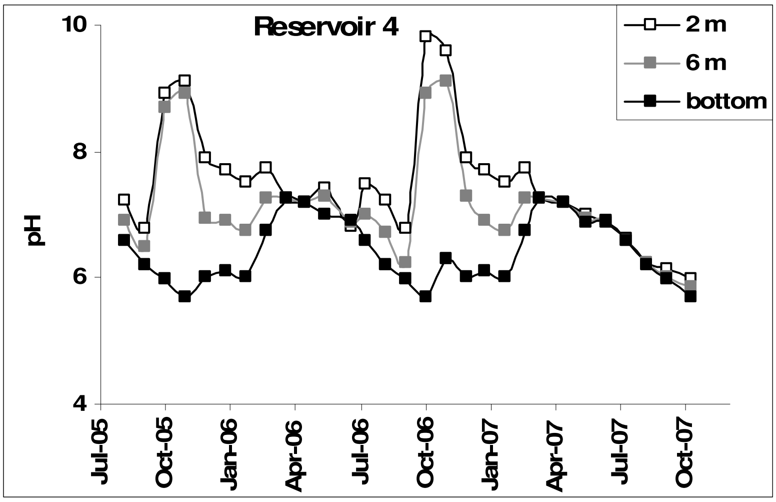

Reservoir 7
TSS in mg/L
2 m
6 m
0
10
20
30
40
50
60
70
80
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

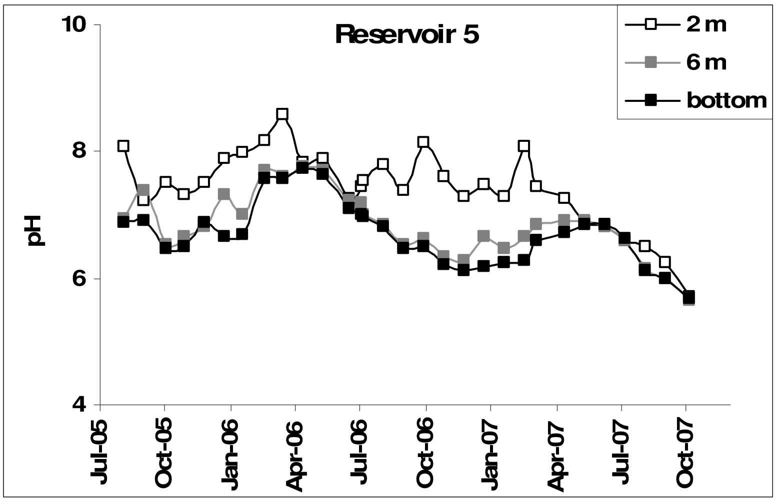
10.4 pH distribution of Reservoirs 1 to 7 (referring to section 3.4.2)

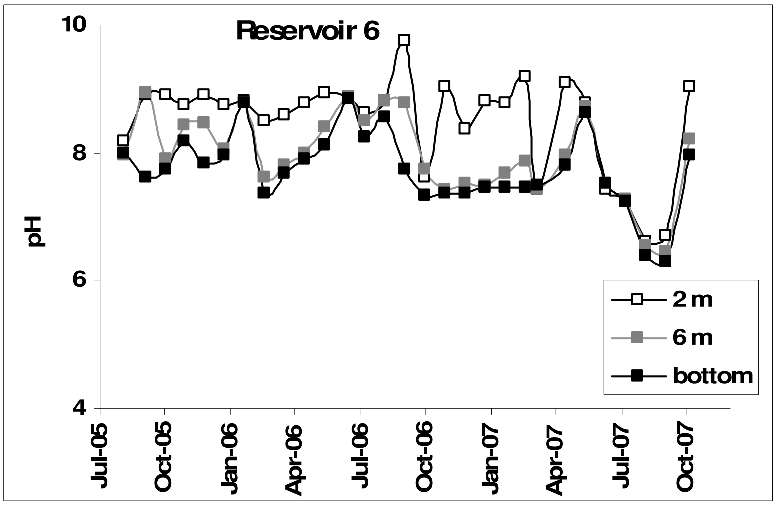


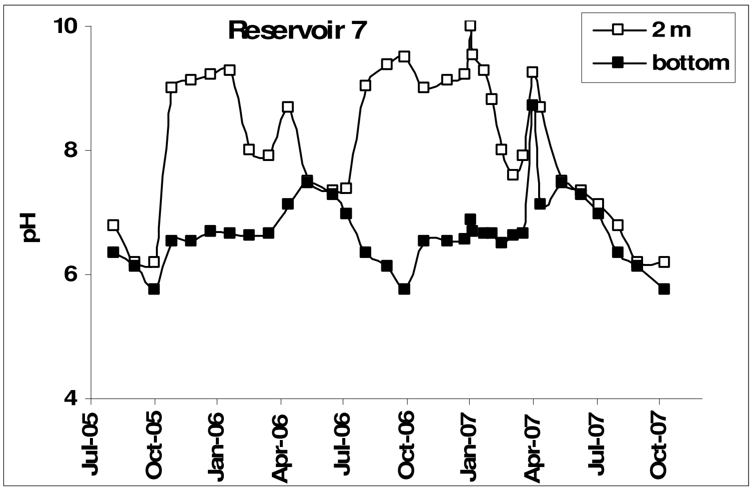




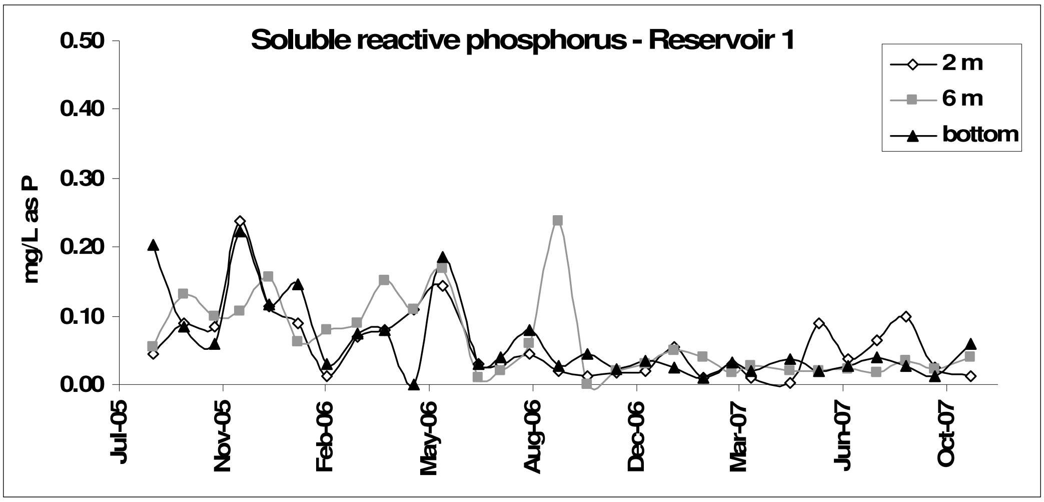

Reservoir 4
2 m
6 m
bottom
pH
10
8
6
4
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

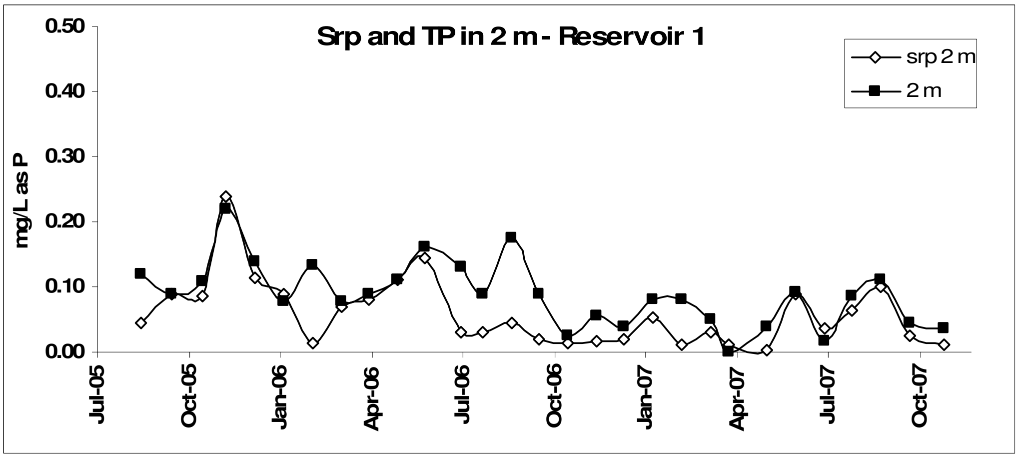

Reservoir 5
2 m
6 m
bottom
pH
10
8
6
4
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

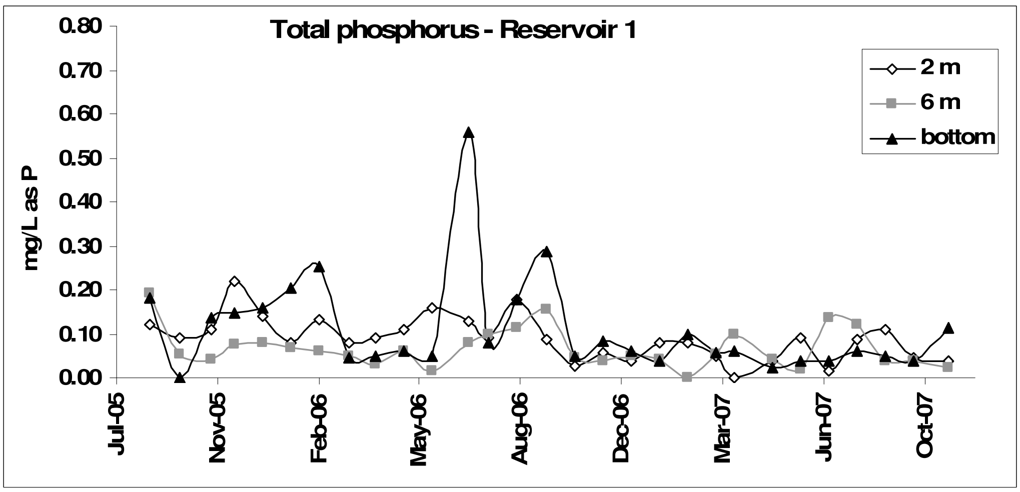

Reservoir 6
2 m
6 m
bottom
pH
10
8
6
4
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

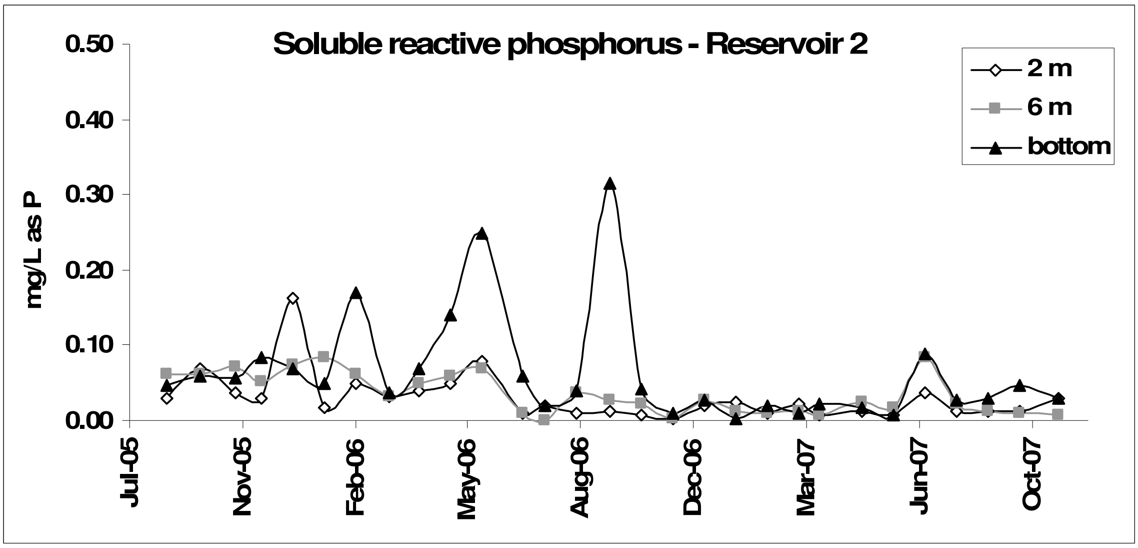

Reservoir 7
2 m
bottom
pH
10
8
6
4
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

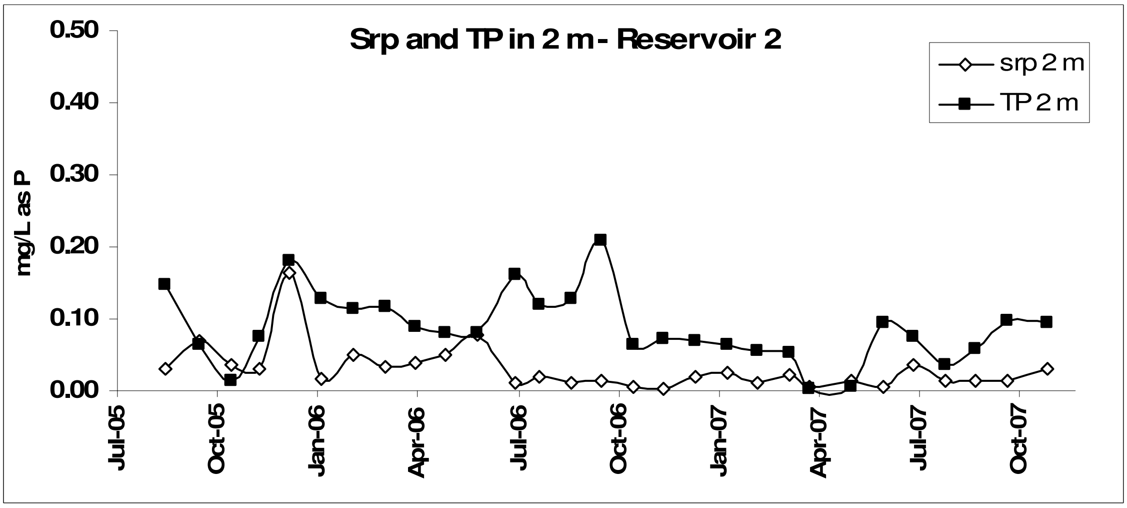
10.5 Soluble reactive phosphorus (srp) and total phosphorus (TP) of Reservoirs 1 to 7 (referring to section 3.4.3)

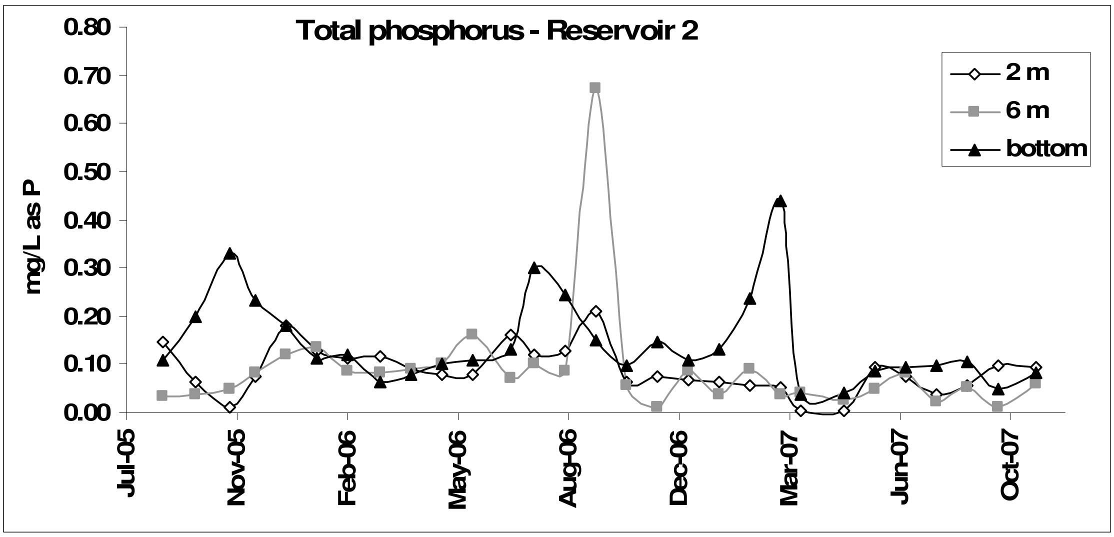


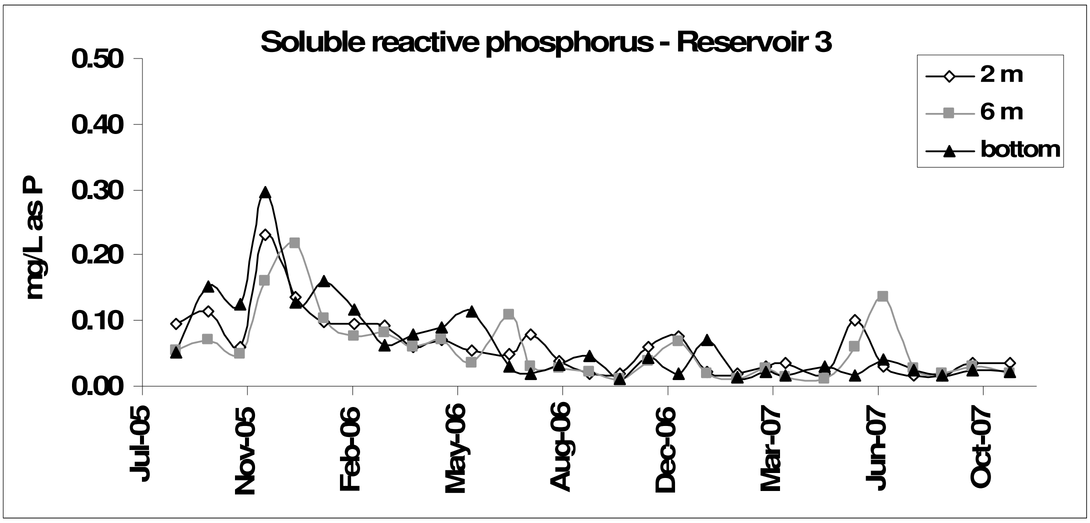


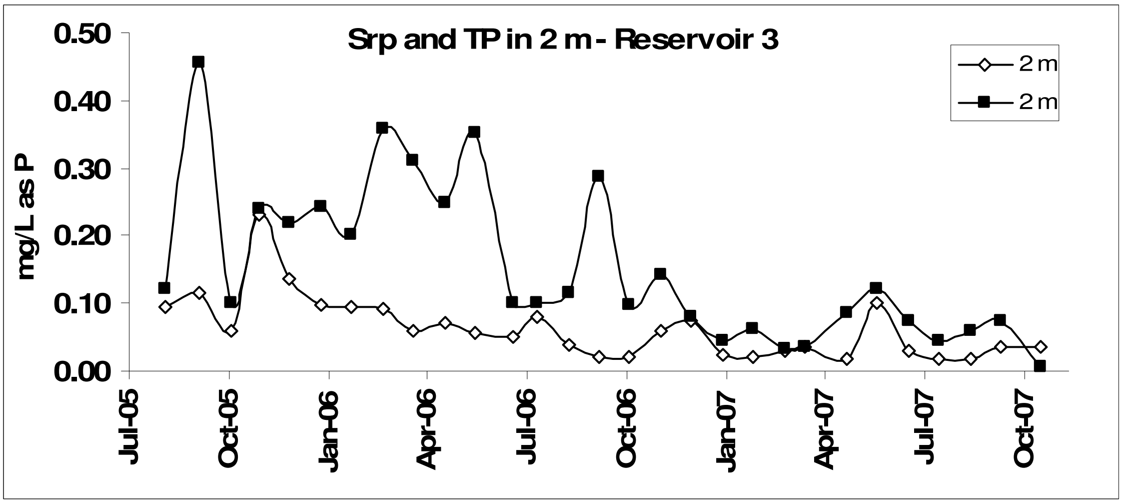


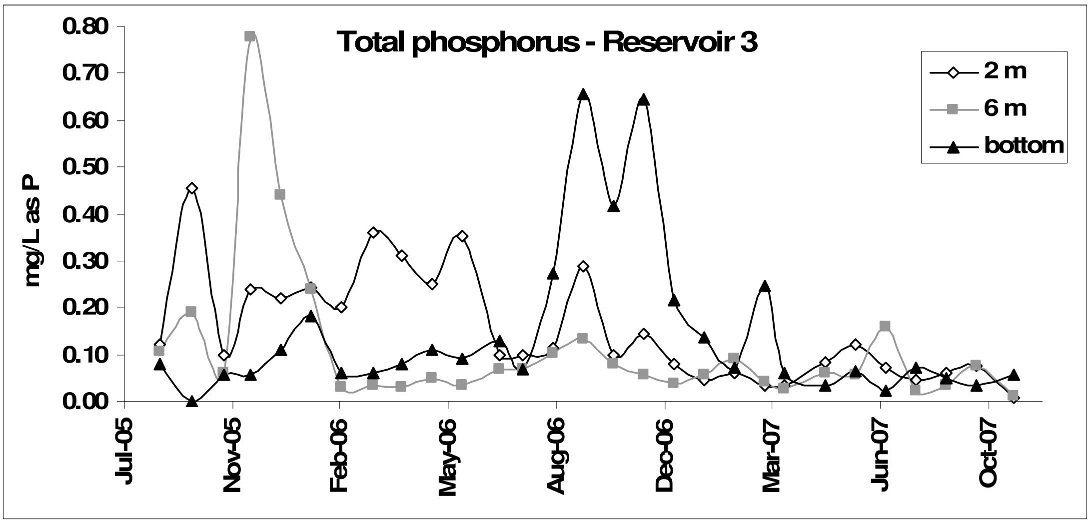

Soluble reactive phosphorus - Reservoir 2
mg/L as P
0.50
0.40
0.30
0.20
0.10
0.00
2 m
6 m
bottom
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

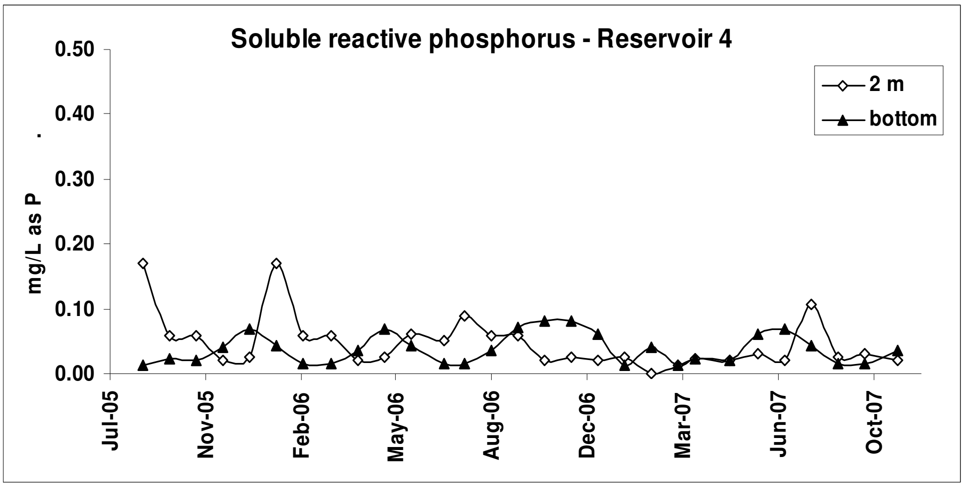

Srp and TP in 2 m - Reservoir 2
mg/L as P
0.50
0.40
0.30
0.20
0.10
0.00
srp 2 m
TP 2 m
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

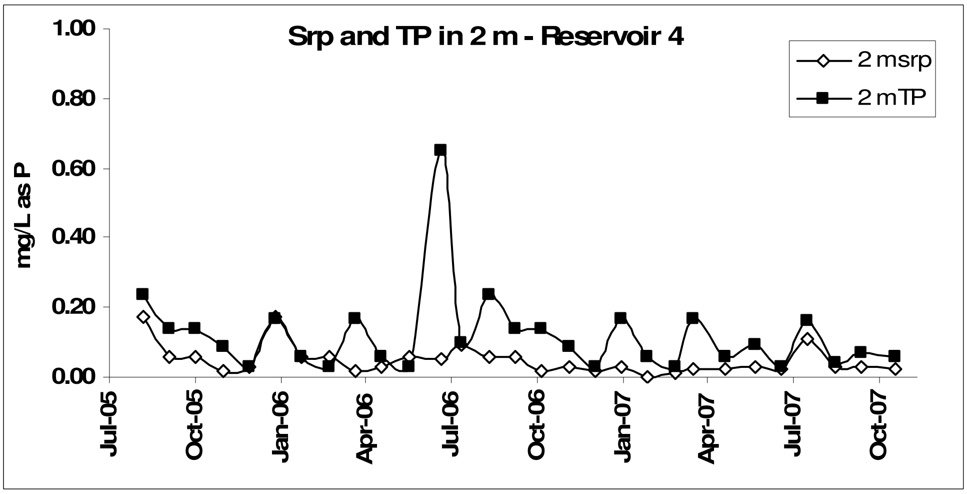

Total phosphorus - Reservoir 2
mg/L as P
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
2 m
6 m
bottom
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

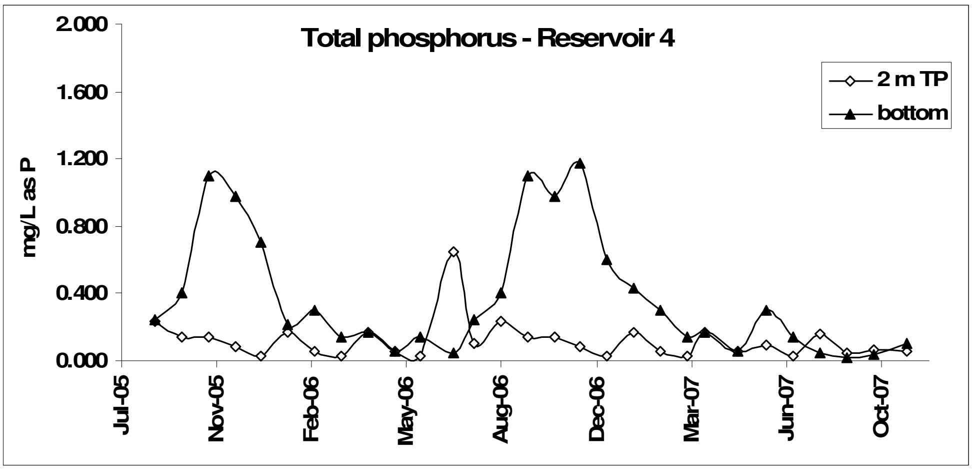

Soluble reactive phosphorus - Reservoir 3
0.50
0.40
0.30
0.20
0.10
0.00
mg/L as P
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07
2 m
6 m
bottom

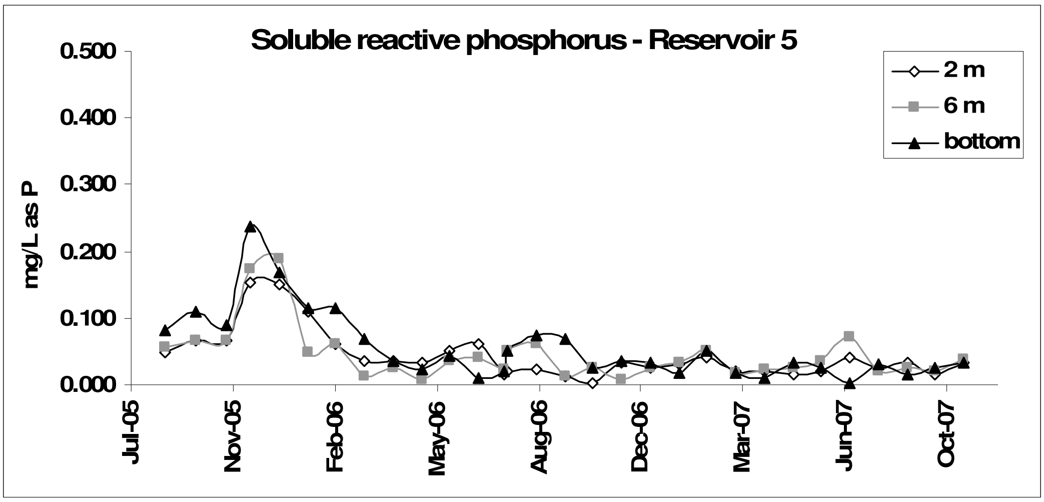

Srp and TP in 2 m - Reservoir 3
0.50
0.40
0.30
0.20
0.10
0.00
mg/L as P
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07
2 m
2 m

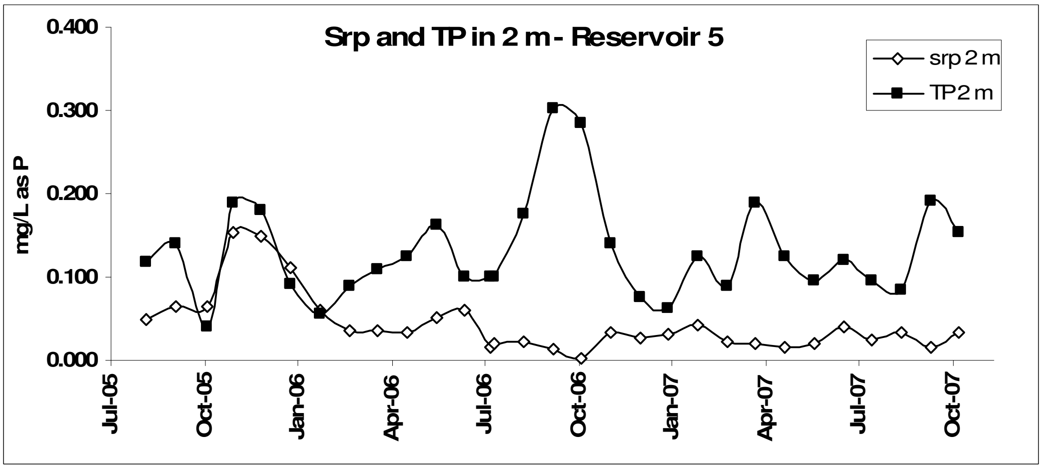

Total phosphorus - Reservoir 3
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
mg/L as P
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07
2 m
6 m
bottom

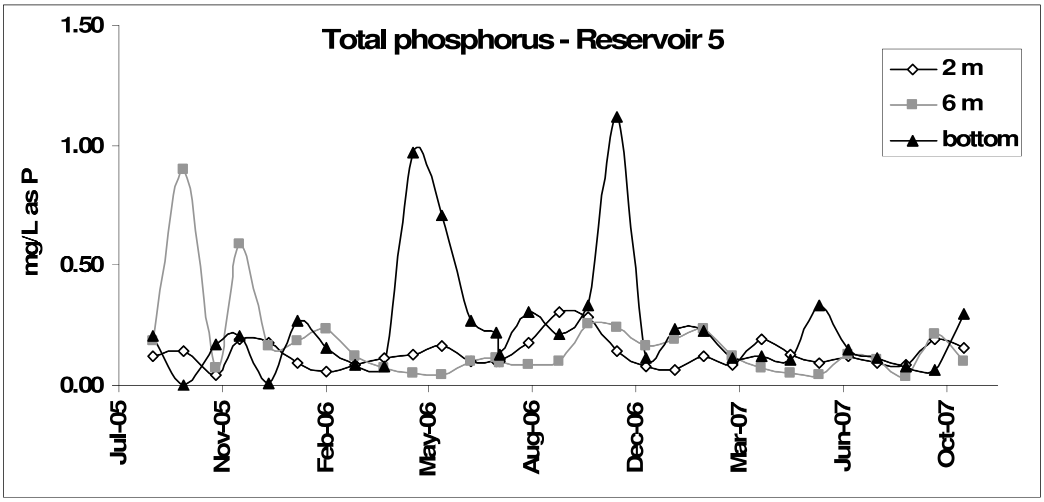

Soluble reactive phosphorus - Reservoir 4
2 m
bottom
mg/L as P
0.50
0.40
0.30
0.20
0.10
0.00
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

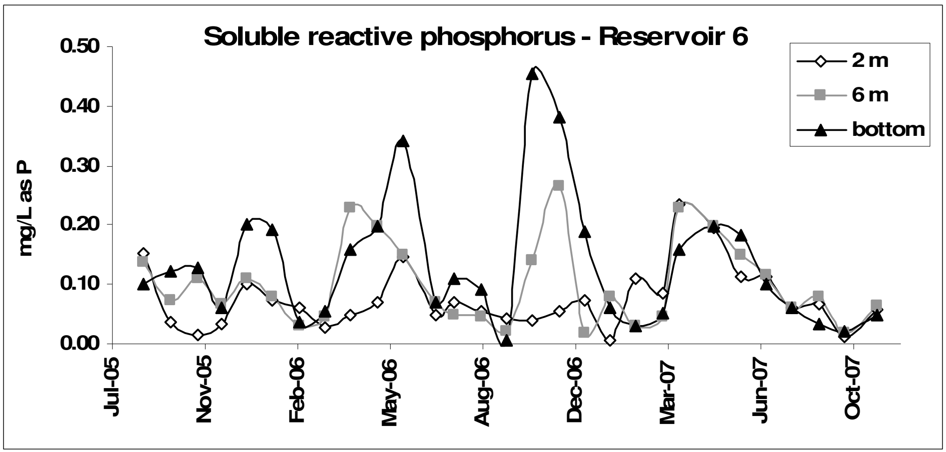

Srp and TP in 2 m - Reservoir 4
2 msrp
2 mTP
mg/L as P
1.00
0.80
0.60
0.40
0.20
0.00
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

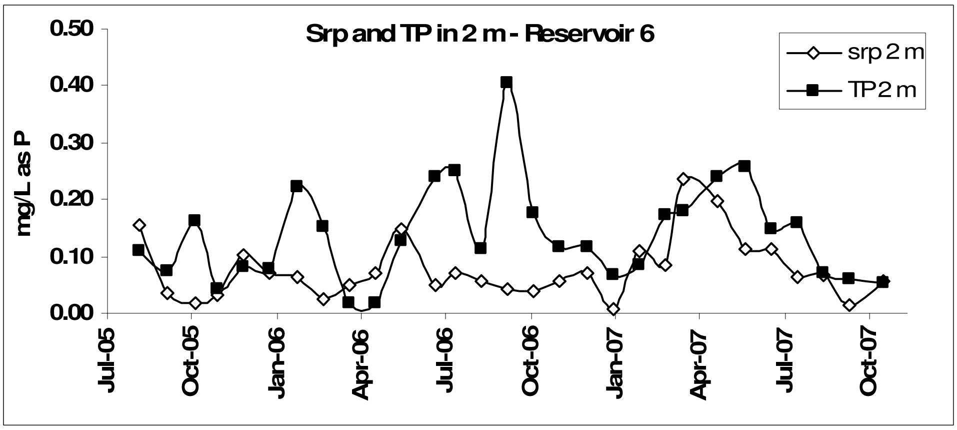

Total phosphorus - Reservoir 4
2 m TP
bottom
mg/L as P
2.000
1.600
1.200
0.800
0.400
0.000
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

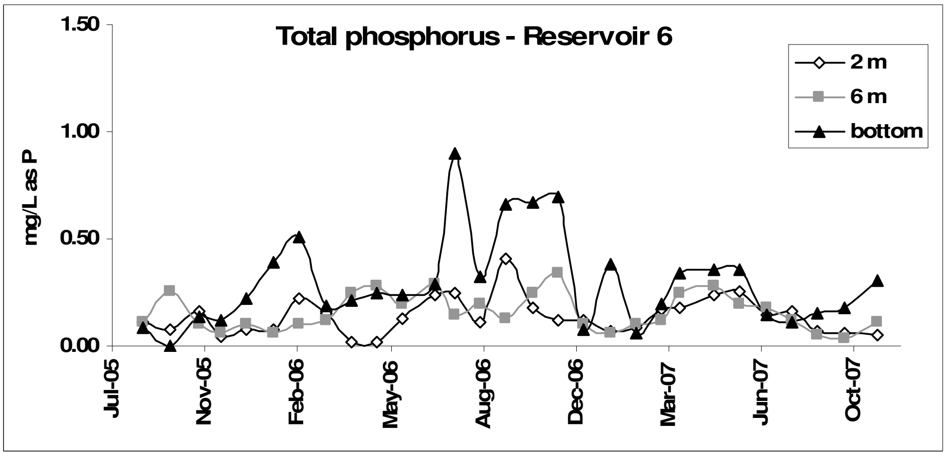

Soluble reactive phosphorus - Reservoir 5
mg/L as P
0.500
0.400
0.300
0.200
0.100
0.000
2 m
6 m
bottom
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

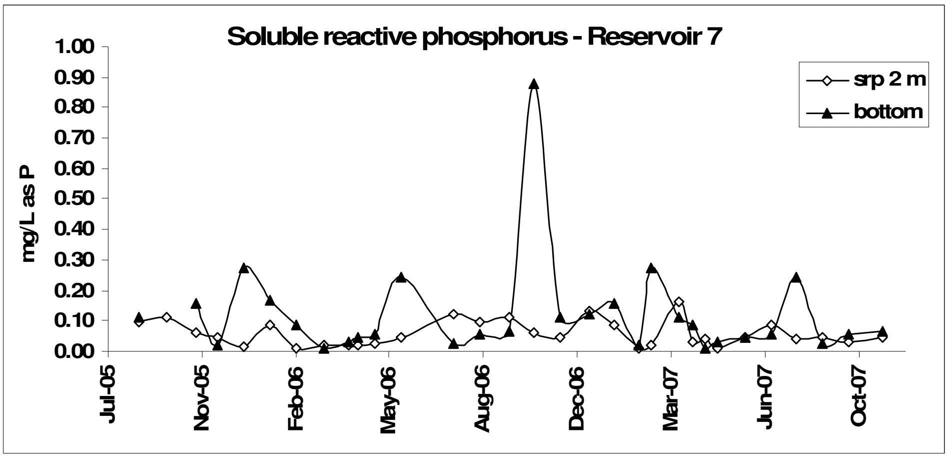

Srp and TP in 2 m - Reservoir 5
mg/L as P
0.400
0.300
0.200
0.100
0.000
srp 2 m
TP 2 m
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

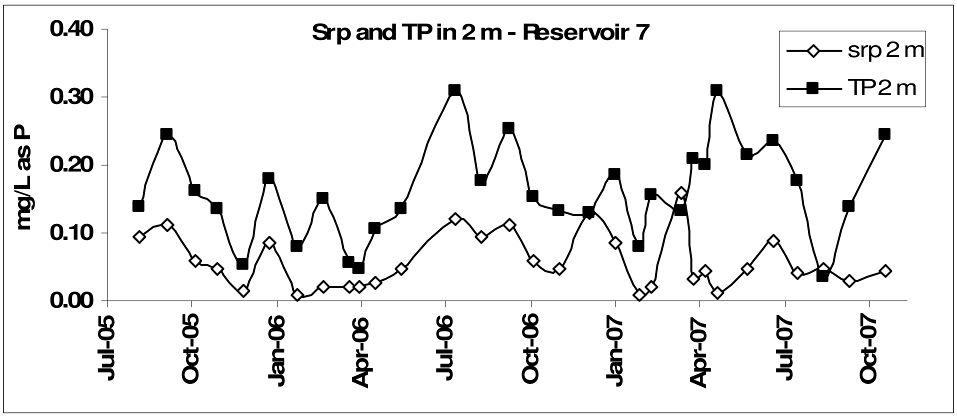

Total phosphorus - Reservoir 5
mg/L as P
1.50
1.00
0.50
0.00
2 m
6 m
bottom
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

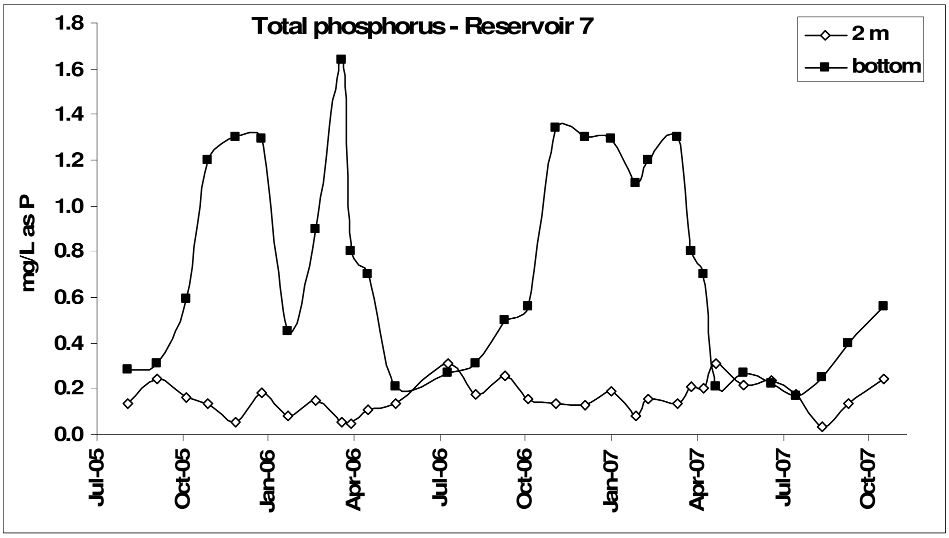

Soluble reactive phosphorus - Reservoir 6
mg/L as P
0.50
0.40
0.30
0.20
0.10
0.00
2 m
6 m
bottom
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07


Srp and TP in 2 m - Reservoir 6
mg/L as P
0.50
0.40
0.30
0.20
0.10
0.00
srp 2 m
TP 2 m
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

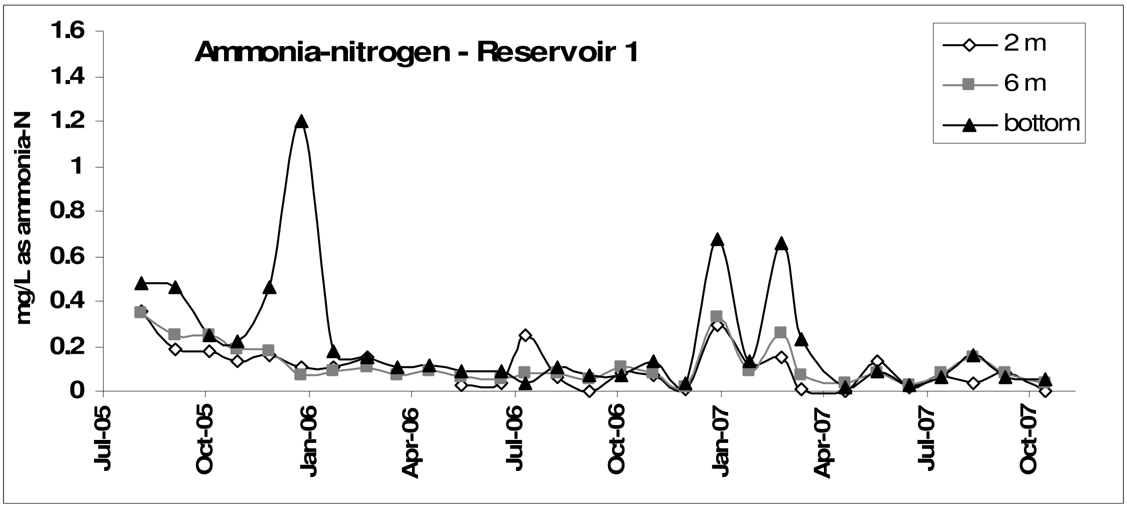

Total phosphorus - Reservoir 6
mg/L as P
1.50
1.00
0.50
0.00
2 m
6 m
bottom
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

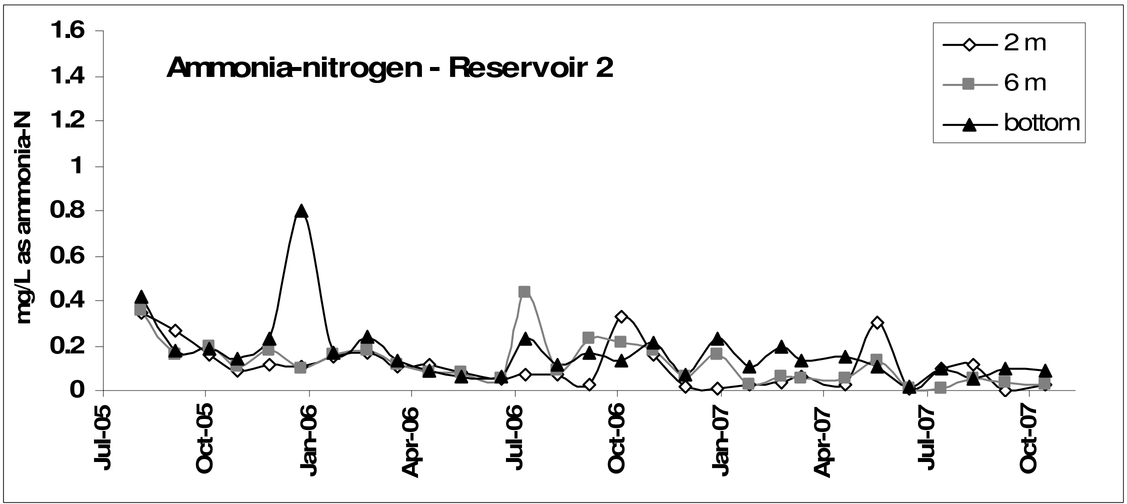

Soluble reactive phosphorus - Reservoir 7
srp 2 m
bottom
mg/L as P
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
Jul-05
Nov-05
Feb-06
May-06
Aug-06
Dec-06
Mar-07
Jun-07
Oct-07

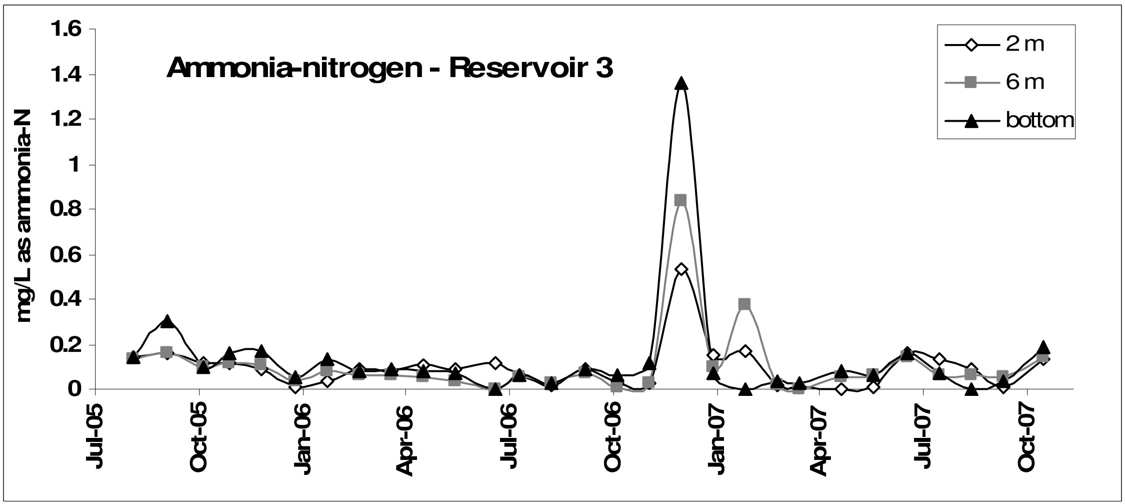

Srp and TP in 2 m - Reservoir 7
srp 2 m
TP 2 m
mg/L as P
0.40
0.30
0.20
0.10
0.00
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

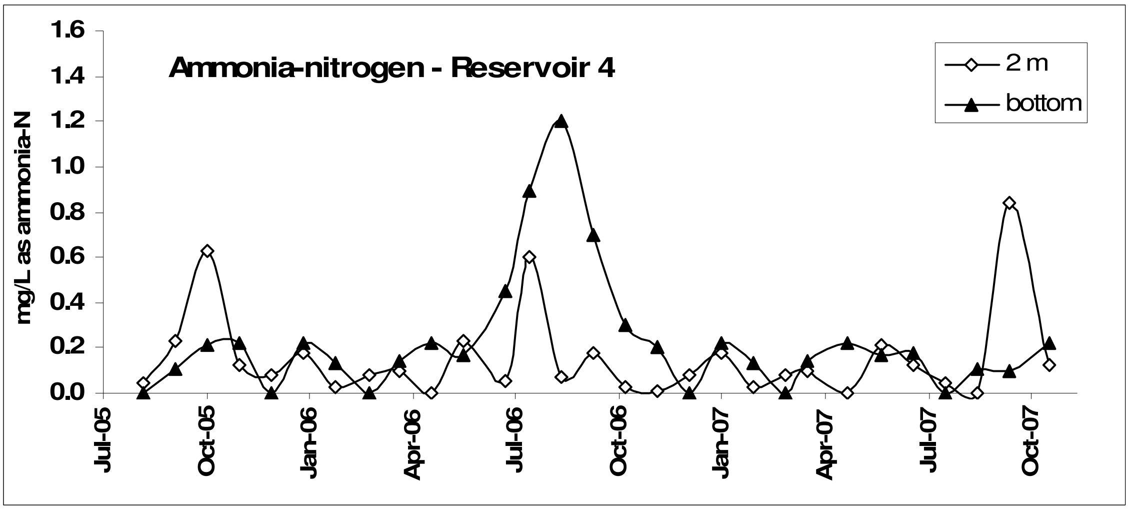

Total phosphorus - Reservoir 7
2 m
bottom
mg/L as P
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

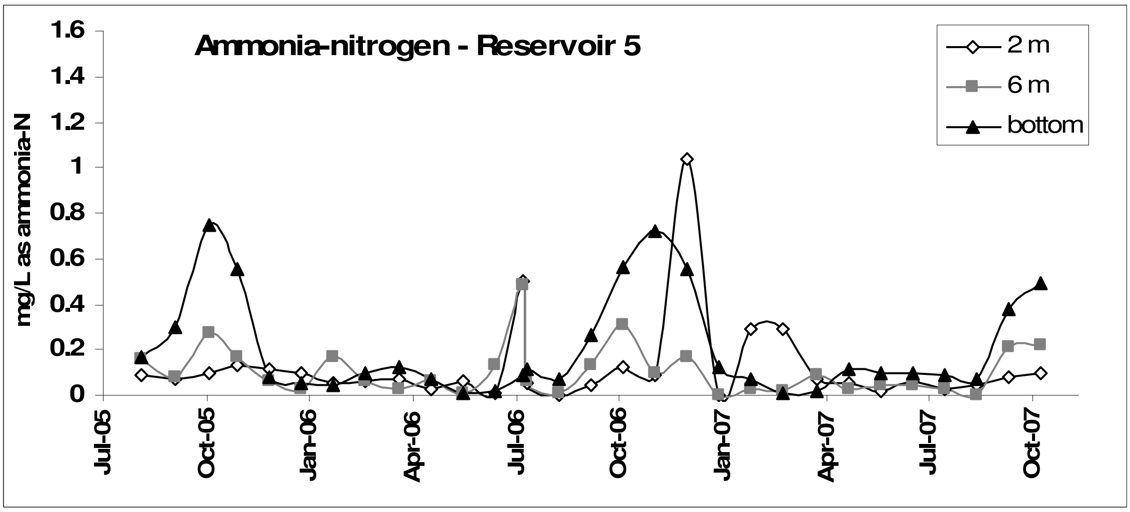
10.6 Ammonia nitrogen of Reservoirs 1 to 7 (referring to section 3.4.3)

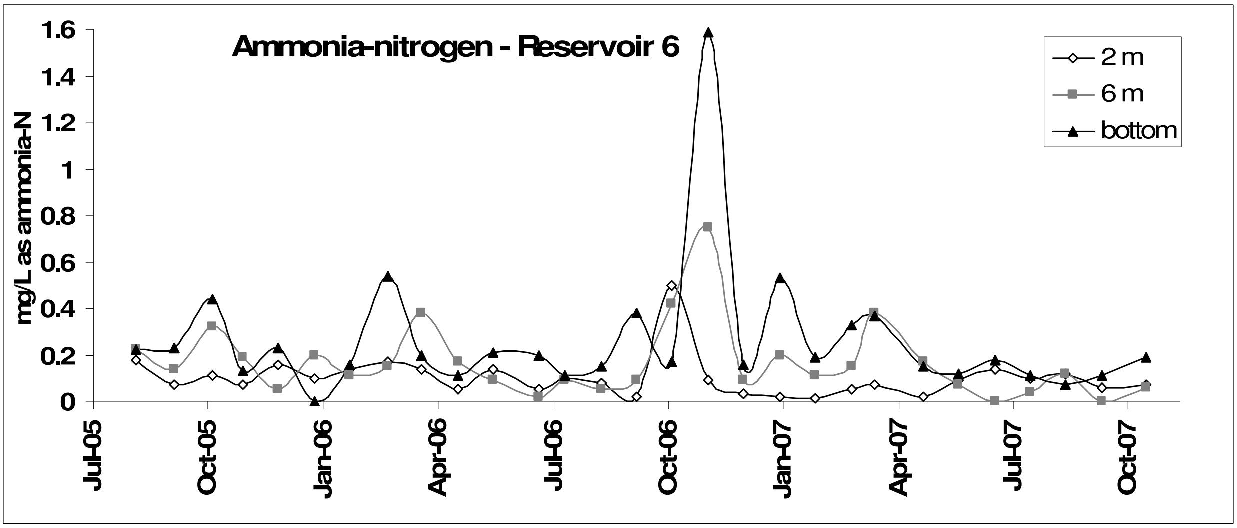




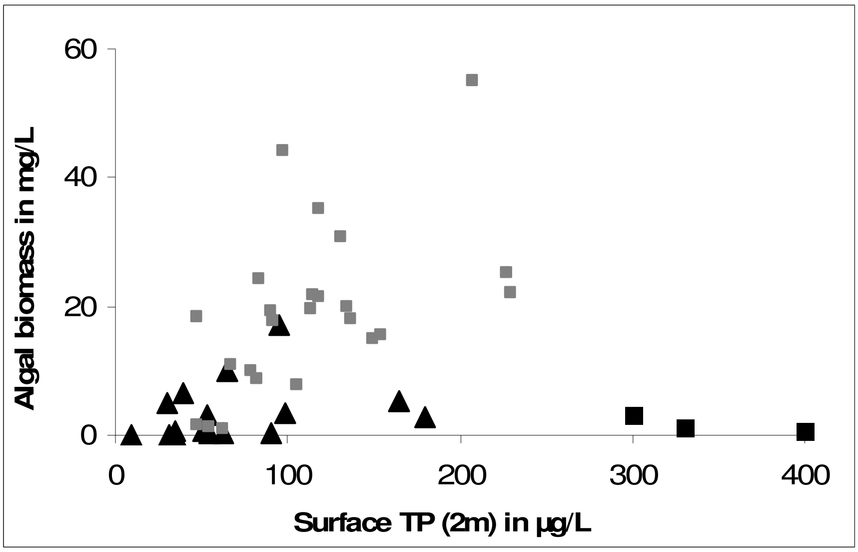



Ammonia-nitrogen - Reservoir 4
mg/L as ammonia-N
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
2 m
bottom
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07


Ammonia-nitrogen - Reservoir 5
mg/L as ammonia-N
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
2 m
6 m
bottom
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07


Ammonia-nitrogen - Reservoir 6
mg/L as ammonia-N
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
2 m
6 m
bottom
Jul-05
Oct-05
Jan-06
Apr-06
Jul-06
Oct-06
Jan-07
Apr-07
Jul-07
Oct-07

10.7 Surface TP versus algal biomass (referring to section 4.4.3)



Black triangles: non-production reservoirs

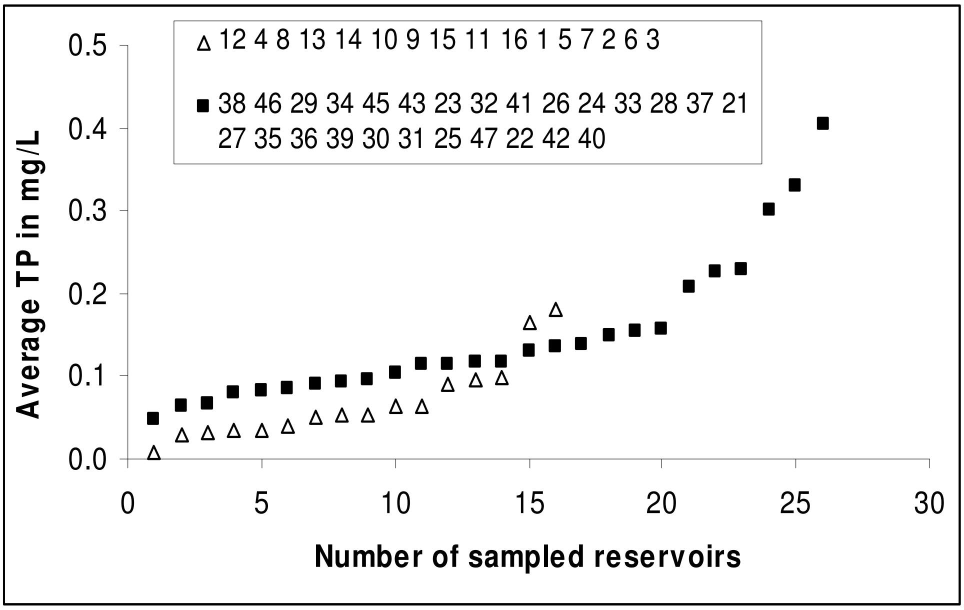
Grey squares: production reservoirs except for the hightest three phosphorus concentration reservoirs

Black squares: production reservoirs – the three high phosphorus, but exceptionally low algal biomass reservoirs

10.8 Comparison of monthly mean to mean of six samples over two years used for overall comparison

	Average of 6 values (plus stdev)	Average of 26 values (plus stdev)	Average of 6 values (plus stdev)	Average of 26 values (plus stdev)	Average of 6 values (plus stdev)	Average of 26 values (plus stdev)
	pH 2m	pH 2m	Avg TP 2 m	Avg TP 2m	Avg NH3 bt	Avg NH3 bt
			in µg/L	in µg/L	in µg/L	in µg/L
Reservoir 1	6.8 ± 0.4	7.1 ± 0.7	66 ± 21	63 ± 42	193 ± 231	230 ± 258
Reservoir 2	7.7 ± 0.5	7.2 ± 0.9	102 ± 53	83 ± 48	120 ± 74	124 ± 99
Reservoir 3	8.2 ± 0.8	7.5 ± 1.3	107 ± 33	119 ± 32	83 ± 21	97 ± 78
Reservoir 4	7.5 ± 0.4	7.3 ± 0.8	119 ± 34	121 ± 39	210 ± 261	229 ± 271
Reservoir 5	8.1 ± 0.5	7.4 ± 1.7	121 ± 42	129 ± 37	143 ± 148	206 ± 222
Reservoir 6	8.2 ± 0.7	8.5 ± 0.7	149 ± 29	153 ± 33	282 ± 167	261 ± 286
Reservoir 7	8.1 ± 0.7	7.9 ± 0.8	149 ± 36	142 ± 41	1148 ± 1176	1356 ± 1704
Reservoir 22	6.9 ± 0.6	6.7 ± 0.4	145 ± 122	114 ± 98	238 ± 65	235 ± 157
Reservoir 23	6.3 ± 0.3	6.6 ± 0.7	106 ± 73	118 ± 80	268 ± 57	255 ± 134
Reservoir 24	7.3 ± 0.5	7.6 ± 0.6	58 ± 34	67 ± 34	250± 117	237 ± 82
Reservoir 26	8.0 ± 1.2	8.1 ± 1.0	320 ± 226	283 ± 152	1890 ± 709	2134 ± 1140
Reservoir 27	8.0 ± 0.6	8.1 ± 0.7	139 ± 40	131 ± 50	2544 ± 2499	2570 ± 1887
Reservoir 32	7.4 ± 0.5	7.3 ± 0.4	82 ± 24	88 ± 48	147 ± 68	170 ± 139

10.9 Average of six total phosphorus values (3 samplings per year per reservoir – 2006 and 2007 – March, June and October/November)



Caption: The reservoirs (16 non-production reservoirs represented by white triangles and 26 production reservoirs represented by black squares) were sorted according to absolute value. The legend indicates the reservoir numbers in the same order as the single squares/triangles in the graph.

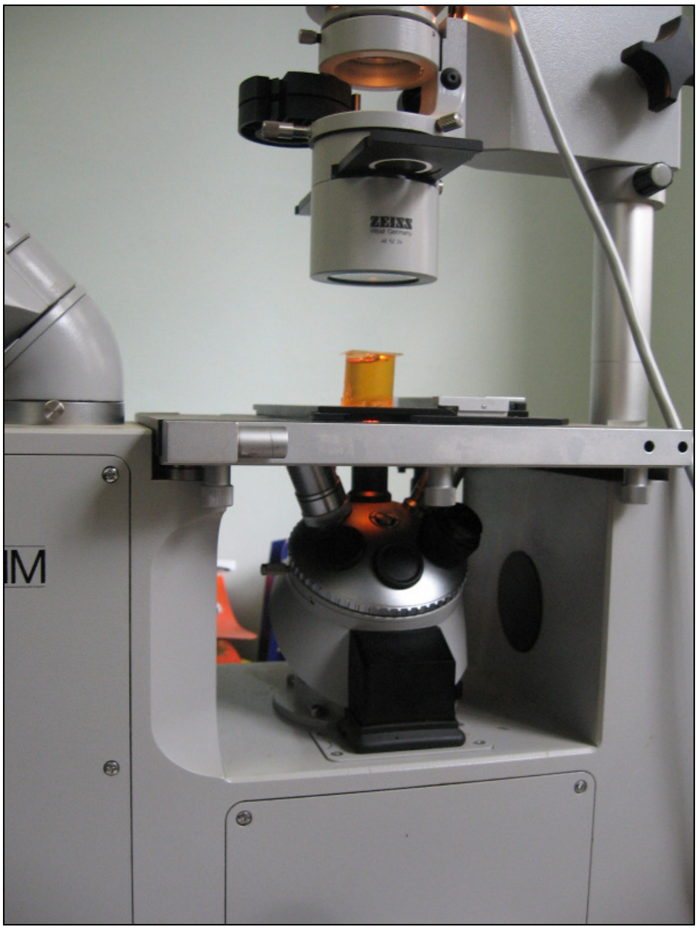
10.10 Pictures of cage system, field and laboratory equipment

Picture 1: Cage system with two cages each 5 by 10 by 4 meters. In the foreground the float with which the cages were accessed.

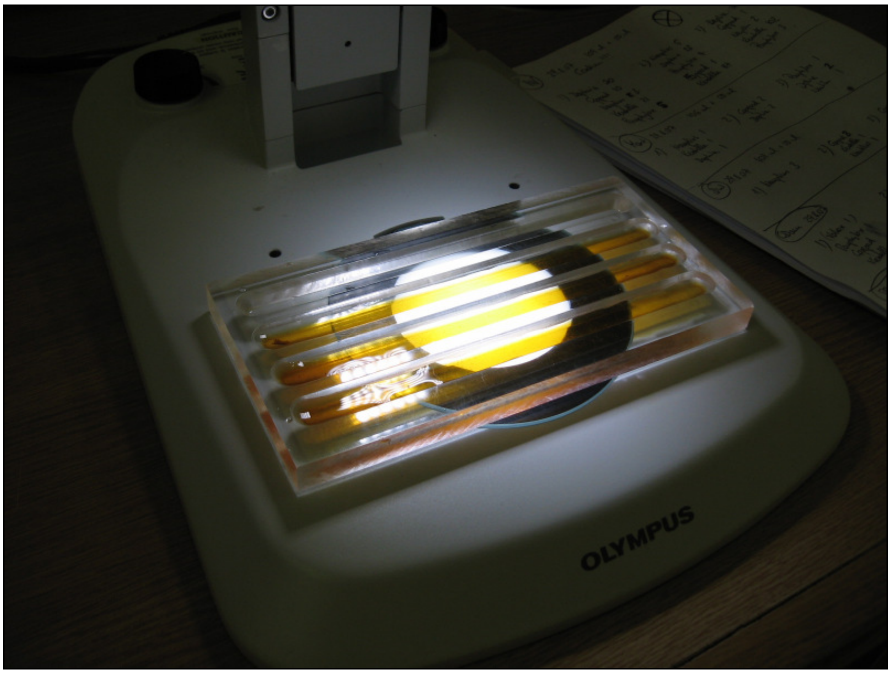
Picture 2: Inflatable boat with which reservoirs were accessed, including typical field work equipment.

Picture 3: Measuring temperature and oxygen concentration of water samples from 2 m, 6 m and near the bottom depths. Samples were collected using a 1.5-L water sampler with a single line trigger mechanism.

Picture 4: Conductivity and pH measurements in the laboratory.



Picture 5: Inverted microscope with phytoplankton counting chamber.



Picture 6: Counting zooplankton using a stereomicroscope.

Picture 7: Outflowing water structure located at reservoir bottom (reservoir empty).

10.11 Further information on production reservoirs (Chapter 4)

	Location	Surface area in ha	Elevation in m (a.m.s.l.)	Water management area	Underlying rock type of the catchment	Surrounding land use
21	**Elgin**	2.1	289	Breede - Bot	Shale, sandstone	orchards
22	**Drakenstein**	6.5	190	Berg - Kromme	granite, alluvial, shale	olives, forestry
23	**Franschhoek**	7.3	240	Breede - Riviersonderend	sandstone, shale	fynbos
24	**Elgin**	5.8	257	Breede - Palmiet	shale, sandstone	orchards
25	**Franschhoek**	6.3	226	Breede - Riviersonderend	sandstone, shale	fynbos, burning
26	**Stellenbosch**	1.6	161	Berg - Eerste	granite, shale	vineyards
27	**Stellenbosch**	4.2	141	Berg - Eerste	granite, shale	vineyards
28	**Franschhoek**	4.1	191	Berg - Paarl Mountain Runoff	granite, alluvial, shale	vineyards
29	**Elgin**	3.2	264	Breede - Palmiet	Shale, sandstone	orchards
30	**Elgin**	3.5	243	Breede - Palmiet	Shale, sandstone	orchards
31	**Stellenbosch**	1.2	190	Berg - Eerste	granite, shale	vineyards, forest, pasture
32	**Elgin**	6.9	137	Breede - Bot	Shale, sandstone	orchards
33	**Worcester**	5	386	Breede - Hex and Smalblaar	Shale, sandstone	fynbos
34	**Tulbagh**	0.9	123	Olifantsdoring - Welgemoeds	Shale, sandstone	fynbos
35	**Rawsonville**	6.4	255	Breede - Romans	Shale, sandstone	vineyards
36	**Ceres**	13.4	1038	Olifantsdoring - Houdenbek	sandstone, shale	fynbos
37	**Rawsonville**	2.3	390	Breede - Romans	Shale, sandstone	orchards
38	**Elgin**	1.4	250	Breede - Palmiet	sandstone, shale	orchards
39	**Ceres**	7.8	530	Breede - Titus	sandstone, shale	fynbos
40	**Ceres**	14.6	949	Olifantsdoring - Upper Olifants and Dwars	sandstone, shale	fynbos
41	**Stellenbosch**	5.2	264	Berg - Eerste	granite, sandstone	fynbos
42	**Malmesbury**	4.7	140	Berg - Diep	shale, sand, granite	pastures, industry
43	**Ceres**	5.3	1233	Gouritz - Touws and Groot	shale, sandstone	fynbos
44	**Somerset West**	16.8	180	Berg - Lourens	granite, sandstone	vineyards
45	**Hermanus**	3.1	42	Breede - Bot	ferricrete, silcrete, sand, shale	orchards, vineyards, vegetables
46	**Worcester**	5.6	216	Breede - Slanghoek	shale, sandstone	vineyards

10.12 Information on Reference (#1-16) and Production Reservoirs (#21-46)

Reservoir #	Water sampling	Phyto and zoo samples (without and with *)	Sediment samples	Water sampling	Elevation	Surface area	Water exchange
Schedule	**monthly**	**monthly**	**twice per year**	**three times per year**	**in m a.m.s.l.**	**in ha**	**times per year**
1	x	x*	x		165	1.7	2
2	x	x*			132	6.9	1.5
3	x	x*	x		152	7.4	0.8
4	x	x			210	4.3	1.5
5	x	x*	x		248	1.8	0.8
6	x	x*	x		75	11	0.8
7	x	x			207	1.4	1.3
8				x	312	240	0.3
9				x	128	2.3	10
10				x	225	6.9	2
11				x	255	7.4	3
12				x	254	8	3
13				x	558	7.1	0.7
14				x	738	7.3	1.5
15				x	224	2.2	0.7
16				x	949	16.5	0.5
21	x	x			289	2.1	1
22	x	x*	x		190	6.5	2
23	x	x*	x		240	7.3	4
24	x	x*	x		257	5.8	0.7
25	x	x			226	6.3	2
26	x	x*	x		161	1.6	0.5
27	x	x*	x		141	4.2	0.3
28	x	x			191	4.1	0.4
29	x	x			264	3.2	0.7
30	x	x			243	3.5	0.7
31b	x	x			210	1.8	2
32	x	x*	x		137	6.9	2
33				x	386	5.0	2
34				x	123	0.9	50
35				x	255	6.4	1.2
36				x	1038	13.4	2
37				x	390	2.3	1
38				x	250	1.4	0.5
39				x	530	7.8	1
40				x	949	14.6	2
41				x	264	5.2	10
42				x	140	4.7	0.6
43				x	1233	5.3	4
44				x	180	16.8	2
45				x	42	3.1	4
46				x	216	5.6	0.7

10.12 continued: Information on Reference (#1-16) and Production Reservoirs (#21-46);

The values are an average of the six samplings in March, June and October/November 2006 and 2007, respectively.

Reservoir #	Production in kg*	Env. stocking density in kg/ha	Years of production	TP 2 µg/L	TP bt µg/L	NH3 bt mg/L	max. peak of alg. biomass	months anoxic <1 mg/L oxygen
1	0	0	0	65	55	0.15	4.00	2
2	0	0	0	99	155	0.12	3.40	2
3	0	0	0	180	160	0.055	2.70	0
4	0	0	0	30	140	0.08	5.10	3
5	0	0	0	91	95	0.175	0.34	3
6	0	0	0	165	260	0.11	5.20	5
7	0	0	0	95	160	0.09	17.00	3.5
8	0	0	0	31	35	0.01	0.14	2
9	0	0	0	51	135	0.02	0.65	2
10	0	0	0	39	130	0	6.62	3
11	0	0	0	54	54	0.13	0.23	1
12	0	0	0	9	35	0.02	0.05	1
13	0	0	0	35	19	0	0.05	2
14	0	0	0	35	60	0.13	0.50	1
15	0	0	0	53	66	0.06	3.08	2
16	0	0	0	63	66	0.06	0.19	1
21	5000	2381	2	135	444	0.32	5.51	7
22	5000	769	7	114	131	0.58	19.51	4
23	5000	685	2	118	183	1.92	21.38	2
24	5000	862	2	67	123	0.6	10.79	3.5
25	5000	794	2	157	110	0.17	1.48	4
26	5000	3125	2	207	613	1.15	55.06	7.5
27	5000	1190	11	131	624	0.28	30.84	8.5
28	5000	1220	1	118	385	0.84	35.17	5
29	5000	1563	2	92	296	0.45	17.83	8
30	5000	1429	2	115	303	0.43	21.83	8
31b	5000	2778	2.5	105	250	0.7	20	8
32	5000	725	2	79	141	0.3	9.95	3.5
33	5000	1000	8	91	261	0.3	19.21	4
34	5000	5556	3	47	63	0.01	18.22	2
35	5000	781	1	54	191	0.26	1.09	4
36	15000	1119	1	82	110	0.1	8.82	4
37	5000	2174	1	154	309	0.08	15.42	6
38	5000	3571	1	405	442	0.26	49	4
39	5000	641	1	97	210	0.07	44.06	5
40	10000	685	1	63	130	0.38	0.96	2
41	10000	1923	5	229	79	0.46	0.74	0.5
42	5000	1064	1	330	461	0.65	1.26	7
43	5000	943	1	84	100	0.11	24.37	2
44	60000	3571	7	300	170	0.26	3.19	4
45	5000	1613	4	137	201	0.63	3.67	2
46	5000	893	2	150	219	0.13	9.60	5

*) 5000 kg estimation: usually ranged from 4800 to 6400 kg