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# **Water Clarity in Two Shallow Lake Systems of the Central Plateau, Tasmania, Australia.**

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**University of Tasmania  
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## Abstract

Lake Crescent and Lake Sorell are two adjacent, large, shallow, mesotrophic lakes of the Central Plateau, Tasmania, Australia. Both lakes are in the same catchment and have similar geological and morphological characteristics. Early limnological work by Cheng and Tyler found trophic characteristics to be substantially different, despite their physical similarities. Historically, each lake exhibited strongly contrasting ‘stable-states’: Lake Sorell was a macrophyte-dominated clear water system, while Lake Crescent was turbid and dominated by phytoplankton. Cheng and Tyler dubbed this a “limnological paradox”.

Since the late 1990s, the quality of the trout fishery declined, nutrient and algal concentrations increased markedly and water clarity declined dramatically. These changes coincided with unprecedented low water levels due primarily to severe drought and competition for water by various users. The rapid decline in water clarity prompted this investigation to determine the underlying processes responsible for the degradation and to recommend management strategies to improve water quality.

Variables limiting light attenuation, turbidity and water clarity were measured from April 2000 to August 2002, and modelled using multiple linear regression. Regression coefficients were used to estimate the relative importance of each water quality component, and this analysis showed that high levels of inorganic suspensoids were largely responsible for the decline in water clarity. Although there were increases in nutrients and suspended sediment in these lakes in the late 1990s, detailed analysis of inputs to and outputs from these lakes suggested that these increases were derived from internal sources as inputs from the surrounding catchment were negligible.

The historical record of lake levels was analysed to quantify the areas of lake bed that would be in contact with the wave base under differing wind conditions and water levels. This showed that both lakes were more prone to wind effects after 1998 owing to the lower water levels. Further, prior to 1999 Lake Crescent was, on average, more turbulent and more prone to wind-driven resuspension events than Lake Sorell, which suggests a potential mechanism underlying Cheng and Tyler’s “limnological paradox”.

The physical disturbance of sediments from wind-driven waves was further investigated by calculating shear stress from wave theory, and quantifying relationships between shear stress and suspended sediment concentration. Shear stress characteristics across the lake basins were modelled under various lake levels and wind speeds, and the magnitude of shear stress increased dramatically at lower lake levels.

DYRESM-CAEDYM was used to develop a sediment resuspension model relating wind, lake-level and sediment flux that was then calibrated and verified against field observations. (The ecosystem model CAEDYM (Computational Ecosystem Dynamics Model) is coupled with the hydrodynamic driver DYRESM (Dynamic Reservoir Simulation Model) to accomplish these simulations). The model was used to ascertain the benefits of managing water levels in the lakes to ameliorate the affects of sediment resuspension and improve water quality. This modelling suggested that the “degraded” state of the lakes from the late 1990s was initiated and sustained by low lake levels leading to increased shear stress acting on the sediments. The increased nutrient concentrations and algal biomass were also found to result from low lake levels because external nutrient loading was insignificant. Conversely, the modelling showed that raising water levels would dramatically improve water clarity.

Alternative, trophically-based explanations of the differences between lakes Crescent and Sorell were examined by investigating the biotic interactions that influence water quality and ecosystem function. The aim was to determine if trophic cascades and stable-state theory would help explain the contrasting phycology of these two lakes.

The historical biological data from lakes Sorell and Crescent was reviewed and reanalysed, and contemporary data collected to compare the trophic structure of the “degraded” status at the end of the 1990s with the historical record. The strong contrasts in phytoplankton productivity and community composition evident between the lakes in the past were still prominent. The algal community of Lake Crescent has concentrations up to  $100 \times$  those of Lake Sorell, and is still dominated by diatoms while green algae dominate Lake Sorell. The zooplankton of Lake Crescent is

dominated by small cladocerans and copepods, while Lake Sorell has more frequent occurrences of larger cladocerans such as *Daphnia*. Lake Crescent also has an order of magnitude greater biomass of the zooplanktivorous fish, *Galaxias auratus*, than Lake Sorell, which leads to a much greater (up to 30 fold) predation pressure on large zooplankters. By contrast the pattern in biomass of the introduced piscivorous brown trout (*Salmo trutta*) between the two lakes is reversed. While some of these patterns are consistent with differences in the nature of top-down trophic cascades between the lakes, the accumulated evidence suggests that such relationships break down at the link between zooplankton and phytoplankton.

The empirical evidence collected suggests that zooplankton grazing had little effect on limiting phytoplankton productivity in either lake for any significant period of time, while the greater dominance of meroplanktonic diatoms and the greater susceptibility of Lake Crescent to wind-driven resuspension suggests a more parsimonious explanation of the persistent phytoplankton dominance in this lake.

DYRESM-CAEDYM was then employed to investigate plankton and meroplankton dynamics in lakes Sorell and Crescent, since this technique can be used to test 'N-P-Z' (nutrients-phytoplankton-zooplankton) models. The hypothesis tested was that developed above: that differences in resuspension combined with contrasts in the proportions of meroplanktonic phytoplankton were sufficient to explain the differences between the two lakes. Modelling of plankton and meroplankton dynamics in both lakes returned significant contrasts in algal productivity that were driven largely by contrasting sediment resuspension dynamics between the lakes.

It was concluded that the differing phytoplankton communities of the lakes are a result of contrasting sediment resuspension dynamics between the lakes, with a limited influence from contrasting levels of zooplankton grazing pressure.

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DYRESM-CAEDYM was developed at the Centre for Water Research (CWR) at the University of Western Australia. The models are freely available and can be downloaded from: [www.cwr.uwa.edu.au/~ttfadmin](http://www.cwr.uwa.edu.au/~ttfadmin)

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# Chapter 1 Introduction

## 1.1 Overview

Lake Sorell and Lake Crescent are two large, adjacent, shallow, productive lakes of the Central Plateau, Tasmania, Australia. They have similar morphometry and similar climate, geology, soils and vegetation in their catchments. The two lakes originated from natural lakes that have been altered at different times, primarily for supply of irrigation water since the 1830s (Cheng and Tyler 1973a).

In recent times, at high water levels, there has been a distinct contrast between the lakes, with Lake Sorell resembling a clear-water, macrophyte-dominated system and Lake Crescent showing characteristics of a turbid, phytoplankton-dominated system. Lake Sorell supported a highly productive and lucrative recreational trout fishery: an Inland Fisheries Commission angler survey of the 1992-1993 season found approximately 11 000 anglers visited Lake Sorell who caught a total of approximately 123 000 fish with an average catch rate of about 1.5 fish per angler per day (Gudde 2004). Catch rates in Lake Sorell had not changed significantly since the 1950s despite significant increases in anglers visiting the lakes each year (Gudde 2004).

Both systems are naturally turbid water bodies by Tasmanian standards with Lake Sorell having a ‘milky’ appearance and a susceptibility to both lake bed and shore disturbance during times of high wind. Lake Crescent, in comparison, maintained an algal biomass at least an order of magnitude higher than Lake Sorell. The increased algal biomass is believed to have maintained a background turbidity of 10 – 20 NTU higher on average in Lake Crescent than Lake Sorell (Cheng and Tyler 1973a). Anecdotal evidence suggested that wind events reduced water clarity in both lakes but such events were short lived, with water clarity returning to background levels within days of the wind event ceasing.

During the late 1990s, water quality in both lakes declined markedly. Turbidity in both systems increased and aquatic macrophytes all but disappeared from Lake Sorell. The recreational trout fishery began to fail which resulted in serious declines in visitation rates to Lake Sorell by recreational fishers. Moreover, there were fears



expressed that Lake Sorell would “flip” into an alternative phytoplankton-dominated “stable state” if the changes seen in the late 1990s persisted.

This recent degradation in water quality prompted this research project with the aim of investigating the mechanisms responsible for the dramatic decline in water quality experienced by the lakes between the period from 1999 to 2002 and to support appropriate management initiatives to improve the ecological health of both lakes, in particular the return of Lake Sorell to a clearer water state and the restoration of its trout fishery. As a result of these investigations, there was also an opportunity to re-evaluate the possible reasons for the striking ecological contrasts between the two systems which led Cheng and Tyler (1973a) to term the differences between these lakes “a limnological paradox”. The ultimate goal being the restoration of the aquatic macrophyte community in Lake Sorell and an improvement in the biological integrity of the system that will ultimately culminate in an improvement in the health of the recreational trout fishery.

Following is a brief outline of each of the chapters presented in this thesis with an overview of their sequence and how they fit together.

The first task was to assemble the existing data on water quality in these two lakes and initiate a more intensive, regular sampling program (which spanned 2000 – 2002) to quantify the temporal changes in the major water quality variables. This information is presented in Chapter 2. Additionally, because concerns have been expressed about changes in land use in the catchments, the imports and exports of nutrients and sediment were estimated and budgets constructed to help identify the likely origin of increased water column loading of nitrogen, phosphorus and suspended sediment.

Chapter 3 further develops the relationships between key variables influencing turbidity, light attenuation and water clarity with quantitative estimates made of the contribution key parameters make to defining the values of turbidity, light attenuation and Secchi depth. Chapter 3 subsequently documents interactions between water level in both lakes and trends in turbidity and suspended particulate material (SPM), with special emphasis on the role of colloidal material.

The data assembled in Chapters 2 and 3 suggest strongly that reductions in lake water level coupled with wind-driven resuspension of sediments can largely explain the observed increases in nutrients, SPM and phytoplankton. Consequently, Chapter 4 quantifies the level of physical disturbance applied to the lake bed under varying lake levels to help determine the degree to which low water levels may influence the hydrodynamic environment of the lakes and ascertain if increased rates of sediment resuspension may be responsible for the severe reduction in water clarity. This component of work is further developed with the application of a coupled hydrodynamic-ecological model called DYRESM-CAEDYM (DYnamic REservoir Simulation Model and ComputAtional Ecosystem DYnamics Model). DYRESM-CAEDYM was used to model in detail the process of sediment resuspension in both lakes and to identify possible management options to ameliorate it.

The remaining two chapters address the ecological differences between these lakes, and the likely mechanisms that have maintained these differences. Chapter 5 examines the trophic structure of the two lakes, with particular emphasis on the contrasts in top-down interactions. The aim is to assess the possible role of biotic interactions in regulating the differences in algal community structure and productivity that were a prominent characteristic of the lakes for the period prior to 1999.

The information gathered for Chapters 3, 4 and 5 was then used for more detailed modelling using DYRESM-CAEDYM in Chapter 6. The aim here was to determine whether “bottom up” processes engendered by contrasts between the two lakes in their susceptibility to resuspension events were sufficient to explain the strong contrast in algal productivity between Lake Sorell and Lake Crescent.

Chapter 7 forms the concluding chapter and is a synopsis of the main findings of each chapter with an emphasis given to the final conclusions made from the study and the long-term management implications the findings of this project have on Lake Sorell and Lake Crescent. The approach taken during the course of the project and the methods developed are also evaluated in their applicability to possible future applications to shallow lake systems in Tasmania, Australia and elsewhere.

The remainder of this introduction chapter consists of a literature review of the areas of shallow lake ecology that were most relevant to this project. Included is an overview of the history of lakes Sorell and Crescent, followed by a discussion on shallow lake ecology and the special limnological characteristics of shallow lake systems. Emphasis is then given to the phenomena of sediment resuspension and the influence this process may play on shaping shallow lake ecosystems under differing climatic and hydrological conditions.

## **1.2 Lake Sorell and Lake Crescent**

Lakes Sorell and Crescent lie on the eastern margin of Tasmania's Central Plateau, an extensive dolerite-dominated feature of the island. Both lakes are natural basins with small catchments (details in Chapter 2) and land use has mostly revolved around some grazing (for sheep and cattle) and minor selective-cut logging of the *Eucalyptus delegatensis* forests. The lakes have had their water levels manipulated since the 1830s (Cutler, Kinrade et al. 1990) with a combination of deepening of the outflow channel and the construction of a weir on the outflow of Lake Crescent resulting in an effective operating range of ~2.0 m at full supply level (FSL). Subsequently, these lakes have had their levels regulated to supply irrigation water and a small amount of town water to downstream users of the Clyde River during the summer and early autumn months (late December through March). Prior to the mid-1990s, however, the most intensive human activity on the lakes was recreational fishing for introduced trout, chiefly in Lake Sorell which was the most popular trout fishery in Tasmania for much of the latter half of the 20<sup>th</sup> Century.

A significant amount of work has been carried on the ecology of lakes Sorell and Crescent with Burrows (1968) and Cheng and Tyler (1973a; 1973b; 1976a; 1976b) documenting various aspects of water quality and trophic status from 1967 to 1971. Additionally, the Inland Fisheries Service (IFS) of Tasmania has sampled fish and the water column for various purposes over the last 10 years, with an increase in the frequency and extent of water sampling occurring over 1999 in the face of a rapid decline in water quality.

Both lakes are polymictic (Cheng and Tyler 1976b) and are highly variable aquatic environments due to their large surface area and shallow average depth (Chapter 2, Figure 1 and Figure 2, Table 2). Cheng and Tyler (1973a) documented strongly contrasting trophic characteristics between the two systems, with the phytoplankton differing in species composition, population structure, biomass and seasonal phenomena despite the lakes being adjacent and having comparable water chemistry, mixis and land use within their catchments.

The difference between the lakes was significant enough to influence both the public's perception of water clarity between the lakes and their ecology. The greater macrophyte cover of Lake Sorell was likely being due to lower levels of light attenuation that would result from a reduced standing crop of phytoplankton in Lake Sorell compared to Lake Crescent (Spence 1982), and one of the littoral wetlands in Lake Sorell has been listed as a RAMSAR site.

Although Cheng and Tyler (1973a) documented the striking contrast in phytoplankton dynamics between the lakes, they were unable to unequivocally explain why the lakes to behaved in such a manner. They called the contrast "a limnological paradox".

However, there were other differences in the ecology of the two lakes which resonate with late 20<sup>th</sup> century research on food webs. Compared with the phytoplankton-dominated Lake Crescent, the clearer-water Lake Sorell had much denser populations of the piscivorous trout but fewer of the endemic golden galaxias (which was presumed to be zooplanktivorous). There were also anecdotal data suggesting differences in the zooplankton between the two lakes which were suggestive of "top-down" control in Lake Sorell and "bottom-up" control in Lake Crescent.

In the mid 1990s European Carp were illegally introduced to Lake Crescent, and manipulating water levels to minimise the opportunity for spawning was an additional management goal. This coincided with increased demand for irrigation water, and a series of dry years which culminated in record low water levels and major changes to the water clarity of Lake Sorell in particular. The littoral wetlands dried, the trout fishery failed, and there was concern for the future of the fish, invertebrates and wetland plants that were endemic to these lakes. Following from these changes,

legitimate concern was whether both these shallow lakes had changed to a new stable configuration of being turbid and dominated by phytoplankton.

Preceding the 'switch', Lake Sorell had high aquatic macrophyte biomass that provided an important resource for aquatic invertebrates and vertebrates. It is believed that the high biomass of aquatic macrophytes in Lake Sorell was a significant contributor to the lakes highly productive trout fishery, as the weed beds provided a significant resource for the aquatic invertebrates in the lake that constituted the main dietary item of the trout (Stuart-Smith 2001). The degradation in water quality that occurred in the late 1990s and the resulting pressure placed on the light climate is likely responsible for the whole-scale disappearance of aquatic plants from the main lake basin of Lake Sorell (Heffer 2003). The loss of aquatic macrophytes is likely a key factor in the decline of the trout fishery in Lake Sorell over recent years.

### **1.3 Shallow lake ecology**

The distinction between shallow and deep lakes refer, in large, to significant contrasts in specific ecological processes that occur within the systems and result in shallow lakes behaving quite differently compared to their deep counterparts (Naselli-Flores 2003). Scheffer (1998), arbitrarily categorised shallow lakes to have an average depth of less than 3 m, although in some instances this may be seen as a simplistic definition in distinguishing a shallow lake from a deep lake (Padisak and Reynolds 2003).

Shallow lakes rarely stratify, at least to the point where they maintain any persistent vertical density gradient (Padisak and Reynolds 2003) and this is largely due to their turbulent and well-mixed nature (Scheffer 1998). In a deep lake, which stratifies over summer, the epilimnion is isolated from the hypolimnion thus reducing interactions of the epilimnion with the sediments. In a deep lake, macrophyte growth is usually restricted to the littoral zone, and plant biomass and cover is relatively small. However, in a shallow lake, the epilimnion is continuously in contact with the sediments, and the entire lake bed can be in the littoral zone with macrophyte growth possible throughout. The significant sediment-water interactions and the potential importance of macrophytes make the functioning of a shallow lake significantly different to that of a deeper lake and increase the importance of the littoral communities in driving primary productivity (Wetzel 2001).

Shallow lakes are generally characterised by both abundant submerged macrophytes and clear water at low nutrient levels (pristine state), or abundant phytoplankton and turbid water at high nutrient levels (degraded state). At intermediate nutrient levels they are dominated by either submerged macrophytes or phytoplankton (Jeppesen, Jensen et al. 1990).

In the normal dynamics of a shallow lake, ecosystem processes and stability are maintained even in the face of moderate and continuous disturbances that occur within the lake and its catchment (Carpenter and Cottingham 1997). This resilience results from negative feedback mechanisms that maintain the lake in either one of the two relatively stable, equilibrium states. The result is a system that may exist in two alternate states, termed ‘alternative stable states’ (Scheffer 1990) and have become popular over the last decade in the study of shallow lake limnology.

The hypothesis of alternative stable states stems from the theory that shallow lakes typically exist in two states: a clear-water, macrophyte-dominated state, and a turbid-water phytoplankton-dominated state, with each state possessing negative feedback loops capable of keeping the system in a stable ‘equilibrium’ (Scheffer 1990).

In the clear-water, macrophyte-dominated state, macrophytes compete for nutrients with phytoplankton; they provide cover for zooplankton from zooplanktivorous fish, in turn increasing zooplankton biomass which increases grazing pressure on phytoplankton (Portielje and Rijdsdijk 2003). Macrophytes reduce turbulence at the sediment surface from wind driven waves thus reducing resuspension of sediments (Jackson and Starrett 1959; James and Barko 1990; James and Barko 1994; Koch and Beer 1996). All these factors help maintain the system in a stable ‘macrophyte dominated’ clear-water state.

The alternative ‘stable’ state is a turbid, phytoplankton-dominated state. High phytoplankton concentrations lead to the shading of macrophytes. Light limitation leads to a decline in macrophyte biomass and the feedback loops which maintained the clear-water state start to breakdown (Blindow, Hargeby et al. 1998). With a decline in macrophyte cover, zooplankton become easy prey for fish. The wave

buffering capacity of macrophytes are lost and sediments are more easily disturbed. With the loss of aquatic plants, the competition for nutrients between macrophytes and phytoplankton is reduced thus increasing the potential for an algal dominated system (Moss 1990).

The theory of alternative stable states, pioneered by Scheffer (1990), is largely dependent on aquatic macrophytes playing a critical role in influencing ecological processes typical of shallow lake ecosystems, such as an existence of a significant competition between phytoplankton and aquatic macrophytes for growth-limiting light and nutrients. The theory also appears partly biased towards top-down trophic interactions that culminate in a significant limitation of phytoplankton standing crop by zooplankton grazing. A system that exhibits classic characteristics of being in either a clear-macrophyte or turbid-phytoplankton dominated state but does not possess the above mentioned controlling factors to limit primary productivity, may ultimately need to be investigated and described in a different light with less of a reliance on stable-state theory to explain ecosystem function and limitation of primary productivity.

There are several possible reasons for a switch from a clear-water to a turbid state. A major physical or biological disturbance, outside the moderate disturbances that a lake ecosystem may be resilient to, may 'tip' the balance in favour of the turbid, phytoplankton dominated state. The disturbance erodes the stabilising positive feedback loops maintaining the lake in a stable equilibrium state (Scheffer 1990) and causes the whole ecosystem to become unstable (Kristensen, Sondergaard et al. 1992).

Such disturbances include increased nutrient loading; changes to the trophic characteristics of the system, such as a significant increase in the recruitment of a large piscivour; or, in the case of extreme low lake levels, increased predominance of sediment resuspension (Blindow, Hargeby et al. 1998; Bachmann, Hoyer et al. 1999). All of these may cause a shift between the clear, macrophyte-dominated state to a turbid state. Water levels have been recognised as being fundamental in the functioning of shallow lakes and have an over-riding effect on the ecology, functioning and management of shallow lakes (Coops, Beklioglu et al. 2003).

In the face of these changes, the long established equilibrium breaks down and the physical process of sediment resuspension may predominate. Phytoplankton dominance becomes significant as competition with aquatic macrophytes is removed. Additionally, progressive shading by increased phytoplankton and epiphytes may further reduce the occurrence of aquatic macrophytes (Lowe, Battoe et al. 2001).

Improvements in the light climate, that could eventuate if lake levels returned to an optimum level, may result in an increase in phytoplankton productivity with the possibility of high phytoplankton biomass giving rise to a turbid phytoplankton dominated state (Hellstrom 1991). In extreme cases significant blue-green algal blooms may eventuate, further degrading the environmental integrity of the system.

Determining the importance of trophic interactions and the relationships between physical and biological processes, allows a complete understanding of what drives ecosystem function. This knowledge can be used to direct management initiatives to reduce the chance of phytoplankton dominance and guide systems towards the desired macrophyte dominated, clear-water state. Lakes may switch several times between clear and turbid phases with years to decades passing between transition phases (Blindow, Andersson et al. 1993).

#### **1.4 Sediment resuspension**

In numerous cases, sediment resuspension has been regarded as critical in degrading water clarity and increasing light attenuation in shallow lakes (Lick 1982; Luetlich, Harleman et al. 1990; Hellstrom 1991; Hawley and Lesht 1992; Kristensen, Sondergaard et al. 1992; Hamilton and Mitchell 1996; Murphy 2001). It has also been identified as important in eroding the long-term, equilibrium states that shallow lakes may develop over time in the absence of significant disturbance. With the cases of Lake Sorell and Lake Crescent, the degradation in water clarity and the large-scale loss of aquatic macrophytes in Lake Sorell warrants a detailed investigation into the significance that sediment resuspension has on influencing water quality and the light climate of both lakes.

The water quality in lakes and rivers depends on the concentration and distribution of dissolved and particulate material (Bailey and Hamilton 1997). Therefore, the process



of sediment resuspension has significant ramifications for the biological and chemical processes occurring in a water body (Blom, Van Duin et al. 1992; Kristensen, Sondergaard et al. 1992; Bloesch 1995; Horppila and Nurminen 2001). Sediment resuspension also has implications for sediment accumulation, an important process that influences the distribution of fine grained material throughout a water body (Hakanson 1982). This is especially important to management as most types of nutrients and pollutants have a high affinity for small organic and inorganic particles (Hakanson 1981).

The turbulence generated by wave action at the sediment surface of shallow lakes has the potential to resuspend lake bed substrate, increasing concentrations of suspended sediment, nutrients and algal cells. This in turn decreases water clarity and light penetration and limits primary productivity (Hellstrom 1991; Arfi, Guiral et al. 1993) as well as increasing the concentration of plant nutrients (Demers, Therriault et al. 1987; Hamilton and Mitchell 1997) and in some instances chlorophyll (Hamilton and Mitchell 1988; Carrick, Aldridge et al. 1993). Wind driven waves may also affect aquatic plants directly by physically uprooting the plant or indirectly through erosion of sediment (Keddy 1982).

Wind is the driving force in wave generation in inland lake systems. The stronger the intensity of the wind, the greater the size of the wave (C.E.R.C. 1977), and wind generated waves are usually the key controlling force influencing sediment resuspension in inland lake systems, although sediment surface currents and bioturbation may be significant (Scheffer 1998).

Resuspension of sediments occurs when a wave is said to 'feel' the bottom. Water movement generated as a wave travels in one direction influences sediments when the depth of water is less than one-half of the wave length. After this point, the orbits of the water particles become elliptical rather than circular, and there is an oscillatory horizontal motion of water immediately over the bottom sediments which may be sufficient to initiate resuspension (Carper and Bachmann 1984). The bigger the wave the greater is the depth at which the wave potentially 'feels' the bottom.

Sediment entrainment occurs when the force of water movement at the lake bed exceeds the stabilising forces of the sediments (Hawley and Lesht 1992; Vlag 1992; Flindt and Kamp-Nielsen 1998) and the energy exerted at the lake bed is sufficient to overcome the cohesive nature of the sediments (Bloesch 1995). The force exerted at the sediment surface is termed shear stress. Shear stress is a function of wind speed, water depth and fetch (Hamilton and Mitchell 1997) with the extent and frequency of resuspension events being determined by the depth characteristics of the lakes (Carper and Bachmann 1984).

Sediment resuspension only takes place if the current-induced shear stress at the sediment surface is greater in force than the cohesive forces holding the sediments together (Bloesch 1995). This force is termed the critical shear stress (Scheffer 1998). Sheng and Lick (1979) found a critical shear of approximately  $0.05 \text{ N m}^{-2}$  to represent the threshold force necessary to initiate sediment resuspension in Lake Erie. This is a similar value to that identified by Arfi et al. (1993) in Lake Ebrië, and is termed the value as the ‘characteristic’ critical shear stress in evaluating the likelihood of sediment resuspension in Lake Pamvotis by Romero et al. (2002). Above this point, the rate of sediment resuspension has been shown to be linearly related to the shear stress at the lake bed (Hamilton and Mitchell 1996; Hamilton and Mitchell 1997).

The critical velocity needed for resuspension depends on the type of sediment. Fine silts and non-cohesive deposits are more easily resuspended than sand and well-consolidated deposits. Consolidation of sediments occurs as sediments are left undisturbed. Frequently disturbed sediments remain unconsolidated and are more easily resuspended (Hawley and Lesht 1992). Due to the frequency and extent of resuspension in a shallow water body, consolidation of sediments is rare (Bengtsson and Hellstrom 1992). As a consequence, the critical shear stress needed to initiate resuspension in a shallow lake is often less than that required for a deeper water body (Bengtsson and Hellstrom 1992).

One of the most important and direct effects of sediment resuspension are changes to the light climate caused by increased turbidity (Kristensen, Sondergaard et al. 1992; Vlag 1992; Blom, van Duin et al. 1994). Sediment resuspension can also enhance algal growth by keeping algae suspended in the water column that had previously

settled out (Hamilton and Mitchell 1997). Several studies have shown that algal biomass and phytoplankton community composition are strongly influenced by sediment resuspension dynamics (Gabrielson and Lukatelich 1985; de Jonge and van Beusekom 1995). Sediment resuspension has also been shown to influence phosphorus concentration in the water column (Kristensen, Sondergaard et al. 1992).

In order to estimate the rate of resuspension and deposition of sediments it is necessary to obtain information on the following (Sheng and Lick 1979):

1. The bottom shear stress generated by both oscillatory currents and steady currents from wind driven waves.
2. Sediment composition.
3. The rate of resuspension and sedimentation as function of shear stress and sediment composition.

For a lake without significant inflows or outflows, the concentration of sediments in the water column is dependent on rates of sediment resuspension, sedimentation and the background concentration of seston and suspended solids present in the water column (Bailey and Hamilton 1997). To describe this balance between sedimentation and resuspension, Hawley and Lesht (1992) contended that, assuming the water is well mixed horizontally and vertically, the depth-integrated change in suspended sediment concentration ( $S$ ) with time can be written as the difference between the two vertical fluxes of entrainment ( $E$ ) and deposition ( $s$ ):

$$D \frac{dS}{dt} = E - s \quad \text{Equation 1.1}$$

Where  $D$  is the total water depth,  $E$  is the upward sediment flux due to resuspension, and  $s$  is the downward flux due to sedimentation. Usually the component of sedimentation in the equation is set to the product of the particle settling rate and the sediment concentration (Hawley and Lesht 1992):

$$s = v (C - C_{\text{bak}}) \quad \text{Equation 1.2}$$

Where  $v$  is the settling velocity and  $C_{\text{bak}}$  is the background concentration of non-settling suspended solids that may be attributed to material such as motile forms of algae and fine colloidal material. The expression therefore takes into account the component of suspended solids that is present in the water column that does not behave as a free-falling particle but has the ability to remain in suspension under quiescent conditions due to forces other than direct settling and entrainment.

The sinking rate or settling velocity of a particle or  $v$  ( $\text{m s}^{-1}$ ) under quiescent conditions may be adequately described by Stokes' Law, where the force of downward motion is equal to the drag force resisting motion (Hakanson and Jansson 1983). Assuming that the shape of the particle approximates a sphere, a general form of Stokes' Law is as follows:

$$v = \frac{g d_s^2 (\rho - \rho_s)}{18 \mu} \quad \text{Equation 1.3}$$

where  $g$  is the gravitational constant ( $9.81 \text{ m s}^{-2}$ );  $d_s$  is the diameter of a sphere with the same volume as the settling particle ( $\text{m}$ );  $\rho_s$  is the particle density ( $\text{kg m}^{-3}$ );  $\rho$  is the density of water ( $\text{kg m}^{-3}$ ); and  $\mu$  refers to the kinematic viscosity of water (at  $20^\circ\text{C}$ ,  $0.001005 \text{ kg m}^{-1} \text{ s}^{-1}$ ; (Stumm and Morgan 1996)).

Equation 1.1 of Hawley and Lesht (1992) is similar to the expression of Scheffer (1998) who describes the change in concentration of suspended sediment also as a simple mass-balance mathematical function of the form:

$$\frac{dS}{dt} = \frac{\Gamma}{D} - \frac{v}{D} S \quad \text{Equation 1.4}$$

where  $\frac{dS}{dt}$  describes the rate of change of concentration of particles in the water column in units of  $\text{g m}^{-3} \text{ day}^{-1}$ .  $\Gamma$  describes the return of sediments to the water column through resuspension in  $\text{g m}^{-2} \text{ day}^{-1}$ . Sedimentation or the loss of particles to the

sediments is proportional to the ratio of the fall velocity  $v$  ( $\text{m day}^{-1}$ ) divided by the water depth  $D$  (m).

The equilibrium concentration ( $S^*$ ) of suspended sediment in the water column occurs when the rate of resuspension is equal to the rate of deposition. Following from equation 1.4 (Scheffer 1998):

$$S^* = \frac{\Gamma}{v} \quad \text{Equation 1.5}$$

If the influence of wind induced currents on sediment resuspension ( $\Gamma$ ) are known then the above equations allow for the development of relationships to determine changes in suspended sediment concentration within a water body (Scheffer 1998).

Two processes of wind-driven waves are responsible for the resuspension of sediments. The first is through wind-induced circulation patterns in the lake, which cause shear stresses at the bottom by which sediments are entrained (Aalderink, Lijklema et al. 1984). These currents will be termed ‘steady currents’ (Herzfeld and Hamilton 2000). The second mechanism is through wind induced waves causing oscillating water movements which are attenuated with increasing depth (Aalderink, Lijklema et al. 1984). These currents will be termed ‘oscillating currents’ (Herzfeld and Hamilton 2000).

In shallow lakes, the bottom shear stresses associated with horizontal currents or steady currents are generally not of the magnitude to initiate sediment resuspension (Luettich, Harleman et al. 1990) and would only occur under very extreme wind conditions. For example, Bengtsson et al. (1990) found wind speeds exceeding  $30 \text{ m s}^{-1}$  were necessary to produce steady currents of the magnitude needed to initiate sediment resuspension, and then only in very shallow water. A similar result was found by Hawley and Lesht (1992) who found bottom shear stress caused by oscillatory currents to be 3-10 times the shear stress produced by steady currents.

Therefore, generally speaking, the dominant force acting on the lake bed is due to the oscillatory currents from wind driven waves. However, despite the reduced influence of steady currents in regards to direct erosion of the lake bed, these currents have been demonstrated to be very important in the redistribution of sediments throughout a water body (Bailey and Hamilton 1997).

The area of sediment resuspension has been defined in previous studies as the point at which the wave base comes in contact with the lake bottom (Gons, Veenigen et al. 1986). At depths greater than this point, the oscillatory currents produced by the motion of the wave have no influence on the lake bed. When the wave travels into areas where the depth is less than one half the wavelength, the wave base comes into contact with the lake bed and sediment entrainment may occur (Carper and Bachmann 1984; Arfi, Guiral et al. 1993; Scheffer 1998).

Hakanson (1982) defines three areas in a lake which differentiate between areas of sediment resuspension and areas of sediment accumulation: 'erosion', 'transportation' and 'accumulation', each of which reflects environments of differing energy, turbulence and exposure. The areas of erosion are high energy areas where sediment resuspension dominates and there is no deposition of fine material. The sediments in the erosional area are usually very resilient to disturbance and are comprised of such material as bare rock, sand and well consolidated clays. Areas of transportation occur where fine material is deposited, but may be resuspended at times of high wind exposure. Sediments in the transportation area are more diverse, possibly consisting of fine sands and coarse silt. The area of accumulation is where fine material is continuously deposited and reflects areas of little or no exposure. The accumulation area is therefore 'outside' the influence of oscillatory currents and sediments are usually very fine silts and mud of high organic and water content.

The transition between the transportation and the accumulation zone therefore defines the boundary between sediment resuspension and sediment deposition; or alternatively the limit of influence of wind-driven waves. This point has been termed the 'critical' limit or 'critical' depth (Hakanson 1981). To determine the transition between the deposition and transportation/erosion areas it is necessary to know the

threshold water velocity required for an oscillatory wave to resuspend fine particles (Rasmussen and Rowan 1997).

Resuspension of sediments occurs when the force of water movement at the lake bed exceeds the stabilising forces of the sediments (Hawley and Lesht 1992; Vlag 1992; Flindt and Kamp-Nielsen 1998). Above this critical point, the cohesive nature of the sediments is overcome and sediment entrainment into the water column results (Bengtsson and Hellstrom 1992). With the initiation of sediment entrainment, the rate of sediment resuspension has been shown to be linearly related to the shear stress at the lake bed (Hamilton and Mitchell 1996; Hamilton and Mitchell 1997).

In the absence of direct measurements of water velocities at the lake bed, numerous studies have successfully applied wave forecasting models based on the Sverdrup-Munk-Bretschneider (SMB) method (C.E.R.C. 1977) to inland lakes to model wave dynamics (Sheng and Lick 1979; Hamilton and Mitchell 1996; James, Best et al. 2004). This allows estimates of wave characteristics, mean current velocities and shear stress at the sediment surface to be made as a function of wind speed, wind fetch and water depth. Having done so, significant relationships have been derived between the degree of shear stress under varying conditions and the rate of entrainment of sediments (Aalderink, Lijklema et al. 1984; Ostubo and Muraoka 1988; Luettich, Harleman et al. 1990; Blom, Van Duin et al. 1992; Hawley and Lesht 1992; Hamilton and Mitchell 1996; Bailey and Hamilton 1997).

### Steady currents

Given information on the magnitude of the mean current velocity  $\bar{U}$  ( $\text{m s}^{-1}$ ), the time averaged shear velocity at the lake bed  $u_{*c}$  due to steady currents can be determined by (Herzfeld and Hamilton 2000):

$$u_{*c}^2 = \frac{f_c \bar{U}^2}{8} \quad \text{Equation 1.6}$$

where  $f_c$  is a current friction factor defined as:

$$f_c = 0.24 \left[ \log \left[ \frac{12D}{k_s} \right] \right]^{-2} \quad \text{Equation 1.7}$$

where  $D$  is the mean depth (m) and  $k_s$  is the bed roughness (m), which assuming is under flat bed conditions, is given by:

$$k_s = 2.5d \quad \text{Equation 1.8}$$

where  $d$  is the diameter of the particle (m).

The calculation of bed shear stress ( $t_c$ ) for steady currents based on the time averaged shear velocity ( $u_{*c}$ ) is outlined shortly.

### Oscillatory currents

Approximation of the oscillatory currents produced by waves at the lake bed may be determined by (Herzfeld and Hamilton 2000):

$$u_{*w}^2 = 0.25 f_w u_m^2 \quad \text{Equation 1.9}$$

where  $u_{*w}$  is the time averaged shear velocity ( $\text{m s}^{-1}$ ),  $u_m$  is the maximum orbital velocity ( $\text{m s}^{-1}$ ) as calculated by linear wave theory, and  $f_w$  is the wave friction factor defined as:

$$f_w = \exp \left[ 5.213 \left[ \frac{k_s}{a} \right]^{0.194} - 5.977 \right] \quad \text{Equation 1.10}$$

where  $a$  is the maximum bottom amplitude also calculated from linear wave theory.



### Bed shear stress

The bed shear stress or the force exerted at the sediment surface may be calculated from the shear velocity due to either steady currents or oscillatory currents according to Herzfeld and Hamilton (2000):

$$\tau = \rho u^2 \quad \text{Equation 1.11}$$

where  $\tau$  is the shear stress ( $\text{kg m}^{-1} \text{s}^{-2}$ ),  $\rho$  is the density of water ( $\text{kg m}^{-3}$ ), and  $u^2$  is the time averaged shear velocity ( $\text{ms}^{-1}$ ) for either steady or oscillatory currents. Therefore the total applied shear stress at the sediment surface is equal to:

$$\tau = \rho (u_{*c}^2 + u_{*w}^2) \quad \text{Equation 1.12}$$

In order to calculate  $a$ , the maximum bottom amplitude, and  $u_m$  the maximum orbital velocity, laminar wave theory is used. From the formulae of (C.E.R.C. 1977), it follows that:

$$a = \frac{H}{2 \sinh\left(\frac{2\pi D}{L}\right)} \quad \text{Equation 1.13}$$

where  $H$  is the wave height (m) and  $L$  the wavelength (m), and:

$$u_m = \frac{\pi H}{T \sinh\left(\frac{2\pi D}{L}\right)} \quad \text{Equation 1.14}$$

When Airy wave theory is able to be applied, the wavelength,  $L$ , may be calculated by (Hamilton and Mitchell 1996):

$$L = \frac{gT^2}{2\pi} \tanh\left(\frac{2\pi D}{L}\right) \quad \text{Equation 1.15}$$

Where  $T$  is the wave period (s). As  $L$  appears on both sides of the equation, an approximation of equation 1.15 is given by (Herzfeld and Hamilton 2000):

$$L = 2\pi D \frac{\left[\tanh\left(\frac{4\pi^2 D}{gT^2}\right)\right]^{0.5}}{\left(\frac{4\pi^2 D}{gT^2}\right)} \quad \text{Equation 1.16}$$

According to C.E.R.C. (1977), wave height and period may be calculated from wind speed, fetch and depth by:

$$H = \frac{0.283 U^2}{g} \tanh\left[0.530 \left[\frac{gD}{U^2}\right]^{0.75}\right] \tanh\left[\frac{0.0125 \left[\frac{gF}{U^2}\right]^{0.42}}{\tanh\left[0.530 \left[\frac{gD}{U^2}\right]^{0.75}\right]}\right] \quad \text{Equation 1.17}$$

$$T = \frac{2.4\pi D}{g} \tanh\left[0.833 \left[\frac{gD}{U^2}\right]^{0.375}\right] \tanh\left[\frac{0.077 \left[\frac{gF}{U^2}\right]^{0.25}}{\tanh\left[0.833 \left[\frac{gD}{U^2}\right]^{0.375}\right]}\right] \quad \text{Equation 1.18}$$

where  $H$  is the wave height (m), and  $T$  the wave period (s).  $F$  is the fetch (m),  $U$  the wind speed 10 m above the surface of the water ( $\text{m s}^{-1}$ ) and  $D$  the lake depth (m).  $g$  is the gravitational constant ( $9.81 \text{ m s}^{-2}$ ).

Having determined shear stress under changing environmental conditions with the use of laminar wave theory, it is possible to develop relationships relating the magnitude of shear stress to changes in sediment concentration of the water column by modelling the process of sediment entrainment and sediment deposition. Following from equation 1.4 and 1.5, Bailey and Hamilton (1997) describes a sediment entrainment-deposition model of the form:

$$C_{n+1} = C_{(n)} + \Delta C_{ent(n+1)} - \Delta C_{dep(n+1)} \quad 1.19$$

Where  $C_{n+1}$  is the concentration ( $\text{kg m}^{-3}$ ) following a time step of  $\Delta t$  seconds,  $C_{(n)}$  is the initial concentration ( $\text{kg m}^{-3}$ ),  $\Delta C_{ent(n+1)}$  ( $\text{kg m}^{-3}$ ) is the change in concentration due to sediment entrainment over  $\Delta t$ , and  $\Delta C_{dep(n+1)}$  ( $\text{kg m}^{-3}$ ) is the change in concentration due to deposition over  $\Delta t$ .

The change in concentration due to sediment entrainment can be described as a function of water depth ( $D$ ) (m), rate of entrainment and shear stress (Bailey and Hamilton 1997):

$$\Delta C_{ent} = \frac{1}{D} * K_r \left( \frac{\tau - \tau_{crit}}{\tau_{ref}} \right) \Delta t \quad 1.20$$

Where  $\tau$  ( $\text{N m}^{-2}$ ) is the bottom shear stress and  $\tau_{crit}$  is the critical threshold shear stress. The excess shear stress in the equation  $(\tau - \tau_{crit})$  is divided by a reference shear stress  $\tau_{ref}$  to make the term in parenthesis dimensionless (Luettich, Harleman et al. 1990).  $K_r$  is the sediment entrainment parameter ( $\text{kg m}^{-2} \text{s}^{-1}$ ).

The change in concentration due to sediment deposition ( $C_{dep}$ ) is a function of the initial SPM concentration ( $C_{(n)}$ ) ( $\text{kg m}^{-3}$ ), water depth ( $D$ ) (m) and the depth averaged settling velocity ( $\omega$ ) ( $\text{m s}^{-1}$ ) (Bailey and Hamilton 1997):

$$\Delta C_{dep} = C_{(n)} \left( 1 - \exp \left( -\frac{\omega \Delta t}{D} \right) \right) \quad 1.21$$

Combining equations 1.20 and 1.21 becomes (Bailey and Hamilton 1997):

$$C_{(n+1)} = \frac{1}{D} K_r \left( \frac{\tau - \tau_{crit}}{\tau_{ref}} \right) \Delta t + C_{(n)} \exp \left( -\frac{\omega \Delta t}{D} \right) \quad 1.22$$

Therefore, from mass-balance calculations and laminar wave theory, it is possible to adequately model the dynamics of sediment entrainment and deposition by combining information on readily measurable variables such as sediment characteristics, bathymetric data and local meteorological conditions. Due to the nature of the information needed for the development and application of sediment resuspension models, the models are largely system specific. However, once calibration and model validation has been achieved, the dynamics of sediment resuspension in a shallow lake system can be largely understood. Such a tool provides the means to forecast potential benefits of management scenarios, such as lake level or wind fetch manipulation, that may be aimed at limiting the area of lake bed exposed to above-critical shear stress created by oscillatory currents derived from wind driven waves.

## 1.5 Water clarity and the light climate

A result of an escalation in the extent and frequency of sediment resuspension in a shallow lake is the increase of suspended sediment loading to the water column, which can lead to significant changes in the light climate. Solar radiation or light availability is one of the most ecologically important parameters in a shallow lake. Nearly all the energy that drives lake metabolism is derived from light utilised in photosynthesis (Wetzel 2001). The viability of phytoplankton and aquatic macrophytes is largely governed by the amount of light available for photosynthesis, as growth is only viable if photosynthesis outweighs respiration (Spence 1982).

Light availability largely determines the area of a lake colonisable by aquatic macrophytes (Spence 1982) and therefore has a significant influence on the extent and viability of the macrophyte community. Light also influences algal productivity; if nutrients are not limiting, light may be the key variable limiting algal growth (Scheffer 1998).

As light is a critical aspect driving ecosystem productivity and defining ecosystem characteristics, understanding the light climate and factors limiting light availability in a shallow lake ecosystem is necessary if management initiatives are to be undertaken to improve the ecological health of a water body.

Two things can happen to a photon of light when it travels underwater, it can be absorbed or it can be scattered. Although water itself contributes to the absorption and scattering of light, the optical properties of turbid lakes depend largely on suspended particles and dissolved substances (Kirk 1980).

Inorganic suspensoids do not effectively absorb light and therefore have a reduced influence on light attenuation. However, by scattering light they increase the effective path length that a light photon travels, in turn increasing the likelihood of the photon being absorbed (Kirk 1994). Absorption effectively removes light from the water column, and is due to all coloured material, such as dissolved organic substances (gilvin and gelbstoff) and algal pigment (chlorophyll) (Scheffer 1998).

Turbidity in essence is a direct measure of the amount of light scattering in the water column (Gippel 1989) and by definition, is largely independent of dissolved colour. Water clarity as perceived by the human eye, is dependent on all components present in the water column. As a result, turbidity, Secchi depth, and light attenuation are influenced by the same types of material, but to differing degrees.

The components present in the water column that influence the light environment as characterised by measuring variables such as turbidity, Secchi depth and light attenuation include inorganic suspensoids, detritus, phytoplankton, dissolved colour (e.g. humic acids) and colloids.

Information on each of these fractions and their relative contribution to light attenuation is important in understanding the ecological processes that limit light availability in a water body. For example, in a lake that has lost its aquatic macrophytes through increased shading, management initiatives to promote macrophyte growth would be different if light attenuation was governed by high algal biomass compared to a seston-dominated light-limited system.

By measuring each fraction over an extended period of time, it is possible to develop a model relating the relative contribution or importance of each fraction to observed optical properties of the water column. This is achieved by fitting parameter values to field data using an optimisation algorithm (Buiteveld 1995), or, more commonly, using multiple linear regression analysis (Kirk 1985; Gerbeaux and Ward 1991; Blom, van Duin et al. 1994; Buiteveld 1995; Scheffer 1998).

Multiple linear regression allows the regression coefficient for each independent variable (IV) to be determined. For example, in the case of modelling light attenuation, the relationship may be written as:

$$\text{Light attenuation (m}^{-1}\text{)} = a \text{ (inorganics)} + b \text{ (detritus)} + c \text{ (algae)} + d \text{ (dissolved colour and colloids)} + e \quad \text{Equation 1.23}$$

Where  $a, b, c$  and  $d$  are the regression coefficients for each discrete fraction that contributes to the observed degree of light attenuation and  $e$  is the residual error term.

The regression coefficients, combined with the value for the IV in question, are proportional to the influence that that variable has on determining the magnitude of the dependent variable (DV), with the DV being turbidity, light attenuation or Secchi depth. By combining the product of each regression coefficient with observed or average concentrations of its respective IV, many researchers model each fraction's contribution to the magnitude of the observed DV in question (e.g. Kirk 1985; Gerbeaux and Ward 1991; Blom, van Duin et al. 1994; Scheffer 1998).

Having established the variables responsible for reducing water clarity, the next step is to determine the mechanisms and pathways that result in the variable(s) having a disproportionate affect on the light climate. Each independent variable that limits light is, to an extent, dependent on different driving regulatory mechanisms that characterise the ecosystem (Scheffer 1998). Increased levels of suspended particulate material, for example, may be influenced by internal and external processes, such as entrained sediment from resuspension or inputs of sediment from streams (Markensten and Pierson 2003). Similarly, high algal standing crops may result from

increased phosphorus loading and eutrophication of the water body (Sondergaard, Jensen et al. 2003) that may be derived from internal loading (through sediment disturbance) or external sources (such as high loads of nutrients from disturbed catchments).

The important role algae and suspended sediment play in light availability and the undesirable consequence of increased levels potentially giving rise to a turbid, phytoplankton-dominated system makes obtaining information on nutrient and sediment cycles in lakes extremely important. Developing nutrient and sediment budgets for lakes and their surrounding catchments provides the information needed to assess the likely importance of internal and external nutrient and sediment dynamics and is of great consequence for successful lake management.

## **1.6 Nutrient, sediment loads and lake wide budgets**

Shallow lakes have a high ratio of sediment surface area to water column which implies that nutrient cycling from the sediments is more important in shallow lakes than in deep lakes (Sondergaard, Jensen et al. 2003). Understanding the processes responsible for increased eutrophication and their relative magnitudes is important if management initiatives focus on bottom-up control of algal productivity through the limitation or removal of nutrients. To achieve this, the estimation of nutrient and sediment budgets is necessary in identifying the sources of nutrients in a lacustrine system.

Increased external and internal nutrient loading can be a major reason for the stable state of a shallow lake ecosystem to change (Scheffer 1998) with numerous studies showing that high phosphorus loading leads to high phytoplankton biomass, turbid water and undesirable biological changes (Sondergaard, Jensen et al. 2003). Often, at low nutrient levels, the lake is in a stable, clear-water equilibrium, with macrophytes predominating. If nutrients are limiting, increased eutrophication may lead to a significant increase in pelagic and epiphytic phytoplankton. Algae effectively compete with aquatic macrophytes for light, and at a high biomass will lead to increased levels of shading and ultimately a decline in macrophyte cover (Blindow 1992). The loss of macrophytes from the system may also lead to a loss in biodiversity, changes in fish community structure and a limitation on the effectiveness

of top-down grazing control of phytoplankton by zooplankton (Sondergaard, Jensen et al. 2003).

Phosphorus is considered to be the key factor causing eutrophication of our waterways (Beaulac and Reckhow 1982; Cosser 1989) as phosphorus has been identified as the main nutrient limiting algal growth in many Australian freshwater water bodies (Wallbrink, Olley et al. 1996).

Phosphorus release from sediments, or internal loading, is an important process leading to eutrophication of a water body. There are considered to be two dominant mechanisms for this. First, diffusion through the water-sediment interface of phosphates dissolved in the pore water, and , second, resuspension of particles followed by desorption of phosphate from the particles in the water column (Aalderink, Lijklema et al. 1984). The latter process will dominate in shallow lakes (Blom, Van Duin et al. 1992) owing to a well-oxygenated upper-sediment layer (that reduces and eliminates anoxic conditions at the sediment surface) and increased turbulence at the lake bed (which entrains sediments and increases particulate phosphorus concentrations in the water column). However, in the absence of sediment resuspension, it must not be assumed that the nutrient flux from the sediments is negligible. At times of calm weather and reduced turbulence, molecular diffusion of nutrients from the sediment pore water will become more significant (Scheffer 1998) and would likely be an important process in nutrient exchange between the benthos and the water column.

High input of phosphorus into a water body usually indicates disturbances within a catchment. This is because phosphorus is typically immobile in all but very sandy soils, as it is readily absorbed to clay minerals and other soil components (Force 1992). Establishing the sources and sinks of nutrients and sediment in a lake, along with quantification of nutrient and sediment loads entering from the lake's catchment allows identification of where nutrient and sediment loads may be problematic.

Elimination of high external loads of phosphorus may lead to little change in internal nutrient levels as high rates of internal loading drive nutrient dynamics (Sondergaard, Jensen et al. 2003). Sondergaard et al. (2003) note that the phosphorus pool in the



sediments may be more than  $100 \times$  the pool present in the lake water column, further highlighting the importance of sediment-water interactions in a shallow lake.

Increased eutrophication can have significant impacts on the stability of a shallow lake ecosystem (Scheffer 1998). The degree of influence increased nutrient loading has on primary productivity is dependent on both bottom-up and top-down trophic interactions. Understanding the role biotic processes play on influencing primary productivity is important in determining the impact increased eutrophication has on lake productivity.

## 1.7 Ecological modelling

The modelling framework chosen for this project was the ecological modelling software DYRESM-CAEDYM (DYnamic REservoir Simulation Model and ComputAtional Ecosystem DYnamics Model) developed by the Centre for Water Research (CWR) at the University of Western Australia. DYRESM is a one-dimensional hydrodynamic mixing model that has successfully been used to predict the vertical distribution of temperature, density and salinity in several lentic waters around the world (e.g. Hejzlar, Balejova et al. 1993; Hamilton and Schladow 1997; Schladow and Hamilton 1997; Bo-Ping, Armengol et al. 2000; Gal, Imberger et al. 2003). The vertical distribution of temperature and salinity is achieved through a series of Lagrangian layers (Hamilton and Schladow 1997), and it is a process-based hydrodynamic model based on well-defined physical relationships and, therefore, requires little, if any calibration.

CAEDYM is a detailed water quality model which is used to describe the biological and chemical characteristics at a short or long time step. It is a self-contained ecological model that has been designed to link to a suite of hydrodynamic models, although it can be used independently when specific ecological processes are to be examined and no spatial resolution is required. The model has been set up largely for assessments of eutrophication, being of the 'N-P-Z' (nutrients-phytoplankton-zooplankton) model format. CAEDYM is an advance on the traditional N-P-Z models in that it serves both as a general ecosystem model (e.g. for resolving various biogeochemical processes) and also as a species- or group-specific model (e.g.

resolving the behaviour of various phytoplankton species) (Herzfeld and Hamilton 2000).

Thus DYRESM forms the hydrodynamic driver that is coupled to CAEDYM so that DYRESM-CAEDYM is potentially capable of integrating both biological and physical aspects of lakes, such as trophic interactions between upper level consumers (e.g. piscivores and planktivores), alongside physical processes, such as the significance of wind and turbulence on disturbing sediments and loading the water column with nutrients. The coupled model potentially uses all of the following components as inputs: meteorological conditions, lake morphometry (and changes in water level), the volume and composition of inflows and outflows, resuspension and settling rates of sediment components, the community composition of phytoplankton and their responses to changes in light and nutrient regimes, zooplankton and fish communities along with their feeding and digestion rates and dietary preferences. (The latest information about DYRESM-CAEDYM is available on <http://www.cwr.uwa.edu.au/>).

Visualisation of the simulation data is by a graphical user interface named MODELLER, which has also been developed by the CWR. The type of visualisation is extensive. A vertical transect depicting the average variation with depth of an extensive array of variables is possible. These 'slices' may be viewed over a selected time period from days to years. However, since neither Lake Crescent nor Sorell stratify, the detailed outputs from MODELLER have been used only sparingly in this thesis.

DYRESM-CAEDYM is useful for examining fine and broad scale trends of water quality within a lake. Simulations over short time scales (days to months) may be run to look specifically at event-based phenomena. Alternatively, simulations may be run using longer time steps over long time periods (years to decades) to examine long-term lake-wide averages and the characteristics of a lake's 'evolution'. This form of modelling can potentially assess the importance of trophic interactions, fluctuating lake levels and the influence of major inflow events from streams.

## **Chapter 2 What are the “drivers” of water quality and water clarity in lakes Crescent and Sorell?**

### **2.1 Introduction**

In the late 1990s, there was an unprecedented and obvious degradation in the clarity of water in lakes Crescent and Sorell. This, coupled with a decline in the trout fishery and the broad-scale loss of aquatic macrophytes in Lake Sorell necessitated an investigation to determine the factors responsible for these detrimental changes. Special attention was given to the change in the light climate of both lakes, as the viability of phytoplankton and aquatic macrophytes is largely governed by the amount of light available for photosynthesis, as growth is only viable if photosynthesis outweighs respiration (Spence 1982), and there was concern that Lake Sorell, especially, may be about to ‘tip’ into an undesirable, phytoplankton-dominated stable state.

A switch from a clear-water to a turbid state may stem from a major physical or biological disturbance, outside the moderate disturbances that a lake may be subjected to without compromising a given state. This major disturbance erodes the stabilising positive feed back loops maintaining the lake in a stable equilibrium (Scheffer 1990) and causes the whole ecosystem to become unstable (Kristensen, Sondergaard et al. 1992). In these two lakes, the major increases in suspended particulate material (SPM) in the late 1990s, may indicate incipient changes in their previously stable states, and, as outlined in Chapter 1, the potential contributors to increased SPM in these lakes are algal blooms, wind-driven resuspension of organic and inorganic sediments, and the persistence of resuspended colloids.

The potential major disturbances in these lakes that may degrade water clarity and increase both living and non-living SPM include: increased nutrient loading, changes to the trophic structure (e.g. increased recruitment of a large piscivore), or, in the case of extreme low lake levels, increased predominance of sediment resuspension (Blindow, Hargeby et al. 1998; Bachmann, Hoyer et al. 1999; Coops, Beklioglu et al. 2003). This chapter focuses on documenting the changes in water clarity and water

quality variables associated with nutrients and algal biomass. Trophic considerations are deferred to Chapter 5.

Increased eutrophication commonly is associated with increased algal standing crop and more frequent algal blooms (Harris 1986). Assessing the trends and understanding the sources of nutrients is important in gauging the likelihood that changes in nutrient levels influence algal productivity.

Nutrients can be derived from both internal sources (e.g. release from sediments under anoxic conditions (Wetzel 2001) or entrainment of sediments by wind driven waves (Hamilton and Mitchell 1988)) and external sources (e.g. tributary streams that transport nutrients from the surrounding catchment (Holtan, Kamp-Nielsen et al. 1988)).

Land use in the catchments of lakes Crescent and Sorell include both forestry (selective logging) and farming (grazing cattle and sheep, with some improved pasture) and have the potential to significantly change nutrient and sediment values entering the lakes. Quantifying the magnitude of nutrient loadings entering the lakes from the sub catchments and comparing these with similar studies allow an assessment to be made about the likely importance of these inputs.

Algal productivity may also be governed by internal lake processes that are largely independent of increased nutrient imports, such as entrainment of meroplankton (Schelske, Carrick et al. 1995), or increased opportunity for growth of phytoplankton due to a reduction in the occurrence of aquatic macrophytes, macrophytes that when abundant, provide various feedback mechanisms that limit phytoplankton growth (Scheffer 1998).

Finally, having examined the likely factors influencing primary productivity, it is necessary to determine the degree to which increased algal standing crop affects water clarity and light attenuation. This allows an assessment of the significance of changes in algal productivity in reducing water clarity and limiting light availability.

To understand the relative roles of these components, it was necessary, first, to collate and evaluate the historical record of changes in water quality in these two lakes to examine whether the “limnological paradox” noted by Cheng & Tyler (1973, 1976) still persisted and to determine if there were any indications of changes to key nutrient concentrations and the biomass of algae. Given the patchy nature of the historical record, however, it was also necessary to conduct a more consistent descriptive survey describing both the light climate and water quality of these two lakes over the study period 2000 – 2002.

There was anecdotal and some quantitative evidence that the concentrations of phosphorus and nitrogen had increased in lakes Sorell and Crescent in recent years, along with the obvious periods of elevated turbidity and low water levels during the 1990s. This warranted a detailed investigation of nutrient and sediment loads entering the lakes from their sub-catchments. The aim was to determine if loadings differed between subcatchments and thus assess whether differences in land management could be contributing to the recent declines in water quality in these lakes. Information on sub-catchment nutrient loadings are also beneficial in determining potential long-term trends and impacts on internal nutrient and sediment stores.

Determining the likely factors responsible for the degradation can be difficult and requires a multi-faceted approach that takes into account both internal and external processes that may influence nutrient values, primary productivity and SPM. Establishing the magnitude to which key variables influence water clarity is difficult and conclusions may be drawn that require significantly different management strategies as highlighted by the Bachmann/Lowe debate with lake Apopka (Bachmann, Hoyer et al. 2001; Lowe, Battoe et al. 2001) where by both authors have identified plausible reasons for the observed degradation in the light climate, increasing productivity and loss of aquatic macrophytes, but have divergent ideas in regards to the suitability of proposed management initiatives to address the problem.

## **2.2 Methods and data sources**

To explore trends in key water quality variables for these two lakes, all relevant, reliable water quality data were collated from historical sources. These data were then combined with an extensive data set collected from more sites and more frequently in each lake over the period of 2000 – 2002. The details of each of these data sets are now described in turn, followed by a description of the analytical methods used.

### ***2.2.1 Collation of historic water quality data***

A thorough literature review was conducted to collate as much data on lakes Crescent and Sorell as possible. Relevant sources of information are summarised in Table 1 along with a brief description of the type and quality of information and data used. Quantitative information on turbidity and nutrients was only available from the work carried out by the IFS identified in the last three rows in Table 1. The data recorded by Cheng and Tyler was useful for establishing that the lakes are well mixed, therefore justifying the depth integrated sampling procedures used in this study. By contrast the other data sources identified in rows 1, 3 and 4 were only useful for qualitative comparisons of phytoplankton and zooplankton community characteristics.

Table 1 Information and data sources accessed during the current project along with brief descriptions of the type and quality of the available information.

Reference / Source	Data / information used, time period	Type of Data	Level utilised
(Burrows 1968)	Zooplankton heterogeneity - 1967	Qualitative/quantitative	Low
(Cheng and Tyler 1973a; Cheng and Tyler 1973b; Tyler 1974; Cheng and Tyler 1976a; Cheng and Tyler 1976b)	Water column mixing, phytoplankton community composition, Secchi depth and lake level information: 1967 – 1971.	Quantitative	Medium
(Chilcott 1986)	Lake level information, catchment land use	Qualitative - literature review – no additional sampling	Low
(Cutler, Kinrade et al. 1990)	Phytoplankton community composition - April 1987	Quantitative – limited - 1 sample from each lake – detailed species composition	Low
Work carried out by the IFS at a time of high water and good eco-system health - unpublished data	Lake level, turbidity and Secchi measurements – ~ fortnightly September 1992 – September 1993	Quantitative – coincided with a period of high water levels and good water quality	Medium
Work carried out by the Hydro Electric Corporation during 1991/92, 1994/95 and 1997/98.	Total and dissolved nutrient values, turbidity, Secchi depth, chlorophyll <i>a</i> , qualitative phytoplankton and zooplankton samples	Quantitative – bi monthly sampling over 12 months giving 6 samples for each time period in question	High
Detailed work carried out by the IFS from early 1996 to the start of 2000.	Lake levels, total and filtered turbidity, Secchi depth, chl <i>a</i> , qualitative zooplankton and phytoplankton sampling, total and dissolved nutrients, phys-chem data.	Quantitative – fortnightly to monthly sampling, 4 sites within each lake for turbidity and Secchi depth; single ‘mid’ site for phyto/zoo and nutrients	High

## 2.2.2 Water quality sampling 2000 - 2002

### 2.2.2.1 Site selection

To gain an estimate of spatial variation and to give a good indication of lake averages, 8 sites were selected for each lake (Figure 1 and Figure 2). Sample locations were



recorded with GPS to allow for wind fetch and shear stress calculations. Four sites were marked with buoys to allow for repeated sampling at the same location, and they coincided with approximate sampling locations used by the IFS for water sampling that was carried out between 1996 and early 2000. An additional 4 sites were included in the sampling program to improve estimates of lake wide turbidity, suspended solids and algal biomass. Table 2 summarises the morphometric details of the two lakes according to the bathymetric maps shown in Figure 1 and Figure 2. Hypsographic curves of both lakes are presented in Appendix 1.

Figure 1 Bathymetric map of Lake Sorell, including approximate locations of water sampling sample stations and the wind station. Main isopleths are at 1m intervals. Arrow denotes true North.

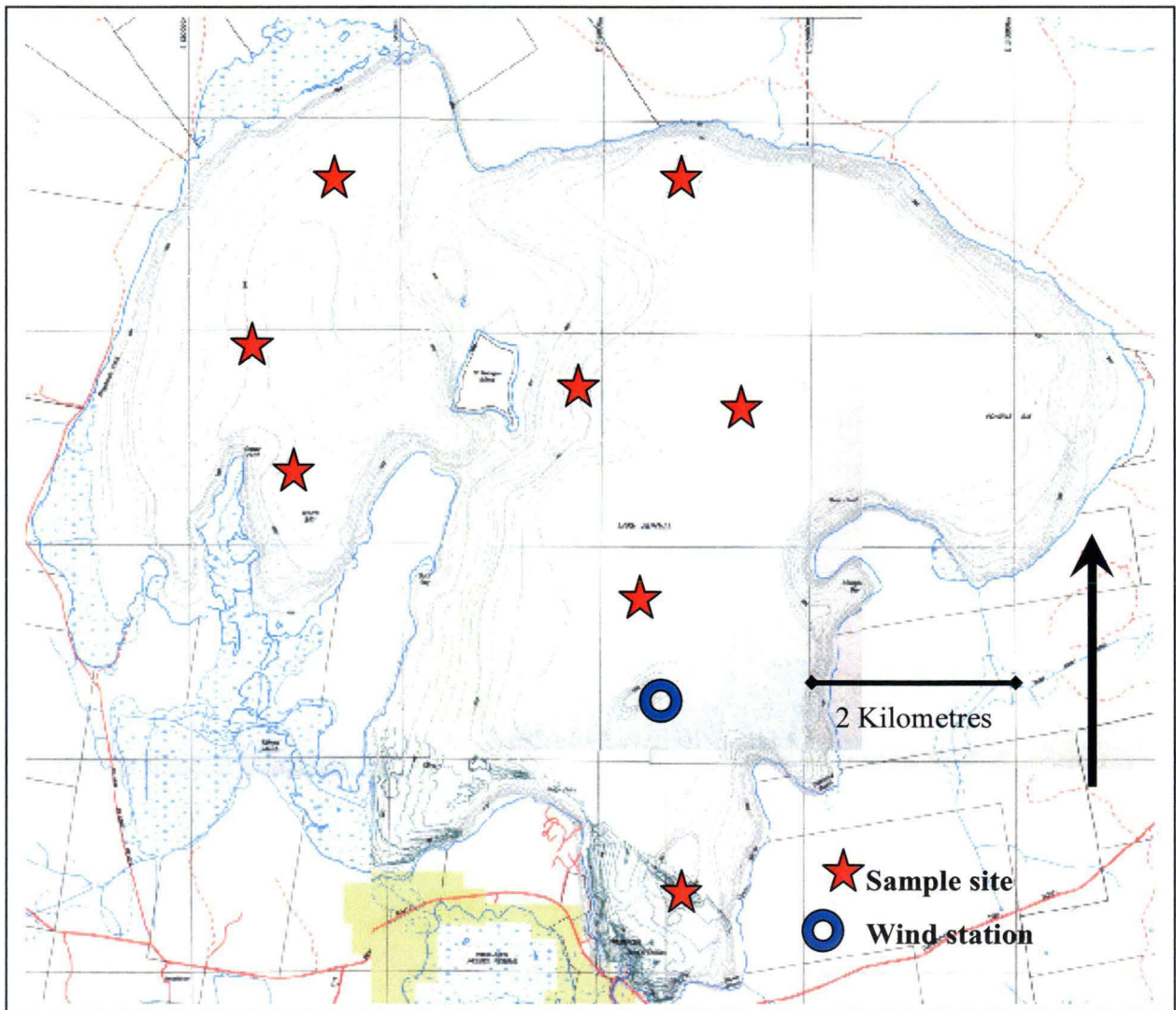
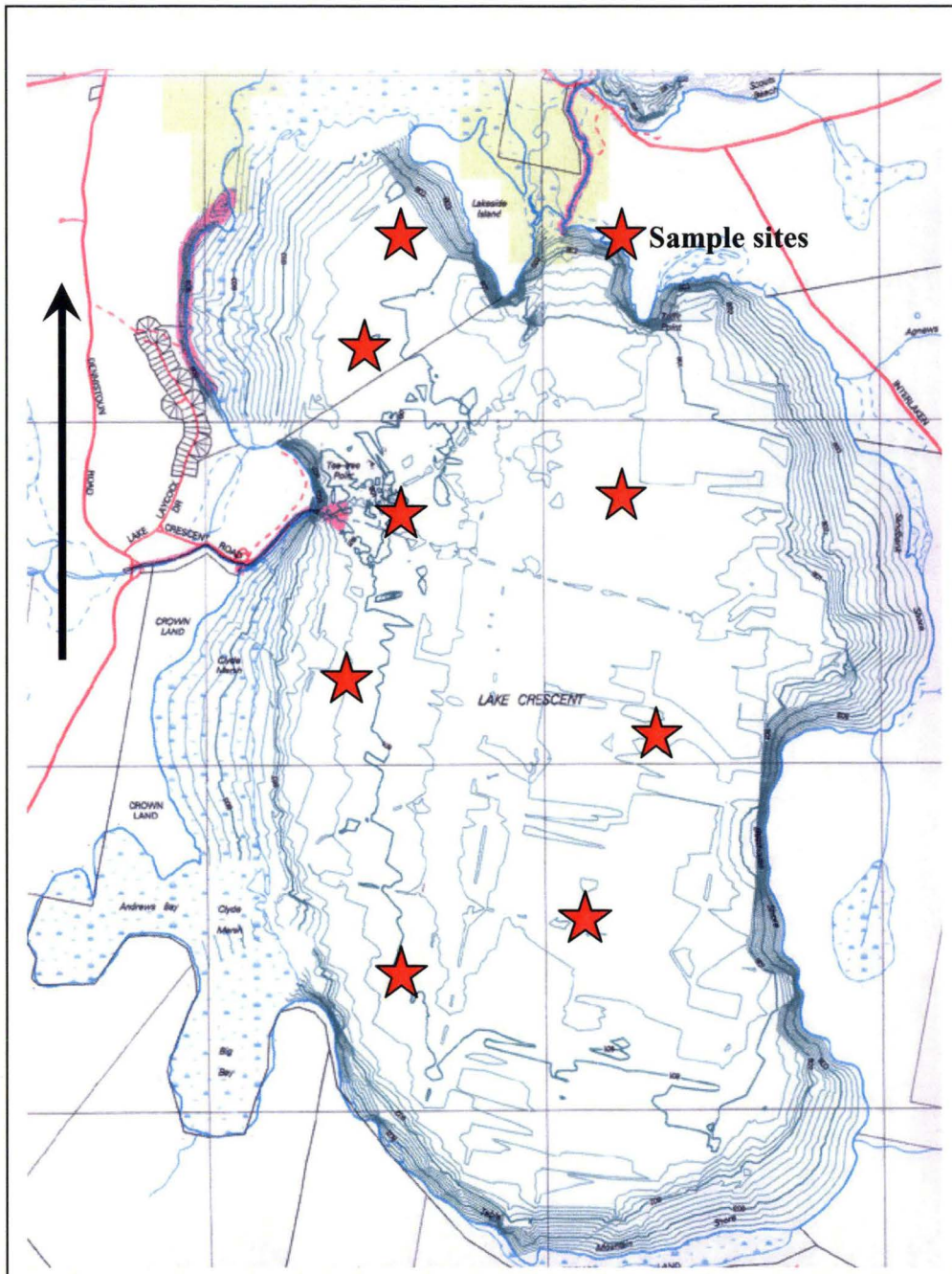




Figure 2 Bathymetric map of Lake Crescent, including approximate locations of water sampling sample stations. Main isopleths are at 1m intervals. Arrow denotes true North.



The bathymetric map of Lake Crescent was produced by the Survey and Geographic Information Department (1996) of the then Hydro Electric Corporation. The bathymetry of Lake Sorell was produced by the Rivers and Water Supply (1902) (Peterson and Missen 1979) and validated by the Survey and Geographic Information Department at the time Lake Crescent was surveyed in 1996. It was concluded that the bathymetric map of Lake Sorell created in 1902 described the lake bed satisfactorily.

Table 2 Morphometric details of lakes Crescent and Sorell when the lakes are at full supply.

	Lake Crescent 803.8mASL	Lake Sorell 804.36mASL
Surface Area (km <sup>2</sup> )	23.1	51.6
Catchment area (km <sup>2</sup> )	32.8	98.4
Volume (megalitres)	49386	156517
Mean depth (m)	2.30	3.07
Max depth (m)	3.80	4.30
Shore Line (m)	29990	52520
Shore Line Development	1.76	2.06

The morphometric data (Table 2) and the hypsographic data (Appendix 1) was determined by digitising the bathymetric maps of both lakes Sorell and Crescent (Figure 1 and Figure 2) and importing the images into the GIS software package MapInfo V6 for analysis.

#### 2.2.2.2 Water sampling

Routine sampling was carried out at fortnightly to three weekly intervals from April 2000 to August 2002. The frequency of sampling was set by logistical constraints on the sampling program and was carried out as regularly as possible. At each site, in-situ measurements of turbidity, Secchi depth and light attenuation were made using the methods and instruments described in Section 2.2.3. A 1-litre depth-integrated sample was taken for the measurement of suspended solids (inorganic and organic), algal biomass (as non-phaeophytin corrected chlorophyll-*a*), dissolved colour and colloidal turbidity. Depth integrated samples were satisfactory due to the well-mixed, predominantly unstratified nature of the water column (Cheng and Tyler 1973a) and

are consistent with similar studies in shallow lakes that have found little stratification with a generally well mixed water column (Hamilton 1990).

Samples were refrigerated at 4 °C and processed immediately on return to the laboratory, which was always within 24 hours of collection. An additional 1 L depth-integrated water sample was taken at the central site of each lake for the measurement of total and dissolved nutrients. This sampling regime for nutrients was consistent with previous sampling regimes and has proven to be sufficient to detect major changes in the concentrations of some of these nutrients. Nutrient samples were transported on ice and frozen on return to the laboratory prior to filtration and analysis by the State Forensic Laboratory (SFL), an Australian National Authority for Accreditation (NATA)-registered analytical chemical laboratory.

### **2.2.3 Analysis of water samples**

#### **2.2.3.1 Light climate: turbidity, water clarity and light attenuation**

Turbidity was measured in-situ with an Analite NEP160 portable turbidity meter (McVan Instruments Pty. Ltd., Mulgrave, Victoria, Australia). The in-situ measurement was corrected to be comparable to the Hach 2100-P turbidity meter (Hach Company, Loveland, Colorado, USA) that had been used to measure turbidity prior to 2000 (Appendix 2). Filtered turbidity was determined in the laboratory with a Hach 2100-P turbidity meter on filtrate passed through a Whatman GD filter paper (1 µm nominal pore size) (Whatman International Ltd., Brentford, Middlesex, UK). Measurements of filtered turbidity were made to estimate the contribution that fine colloidal suspended particulate material made to measurements of total turbidity.

Secchi depth was measured with a Secchi disk on the shaded side of the boat. An average was taken between where the disk is visible and where it disappears from view after raising and lowering through the water column at least 5 times.

Light attenuation was measured with a LI-COR LI-192 Underwater Quantum Sensor (LI-COR Biosciences, Lincoln, Nebraska, USA). Triplicate readings at several depths were taken which agreed to within  $\pm 5\%$ . Kirk's (1994) relationship was used to calculate light attenuation and euphotic depth:

$$K_d = \frac{1}{z_2 - z_1} \ln \left( \frac{E_d(z_1)}{E_d(z_2)} \right) \quad 2.1$$

where  $K_d$  is the vertical light attenuation coefficient,  $z_1$  and  $z_2$  are water depths (m) from the water surface, and  $E_d$  is the measured downward irradiance at depth  $z$ .

The euphotic depth has been taken as the depth to which 1% of surface PAR reaches and approximates the lower limit of photosynthesis (Kirk 1994). This is similar to a number of other studies that have found strong relationships between turbidity derived from suspended sediment and increased light attenuation (Schagerl and Oduor 2003; James, Best et al. 2004).

### 2.2.3.2 Laboratory determination of suspended solids, dissolved colour and colloidal material

Inorganic and organic suspended solids were determined gravimetrically according to Standard Methods (A.P.H.A. 1992). Dissolved colour (or gilvin as g440, i.e. absorbance at 440 nm) was measured on filtrate passed through a Millipore 0.45  $\mu\text{m}$  membrane filter paper (Millipore Corporation, Billerica, Mass., USA) (Kirk 1976). The contribution of colloidal material was determined by reading spectral absorbance at 750 nm (turbidity correction as per Standard Methods) (A.P.H.A. 1992). Absorbance readings were initially measured on a Shimadzu UV-120-02 spectrophotometer (Shimadzu Corporation Ltd., Nakagyo-ku, Kyoto, Japan) in a 4 cm cuvette. Later, g440 was measured on a Varian Cary 50 CONC spectrophotometer (Varian Inc., Palo Alto, California, USA) in a 10 cm cuvette. Absorbances measured on the Varian were corrected so as to be comparable to measurements made on the Shimadzu (Appendix 3).

For the purpose of this study, 'colloidal' was defined, following Whitten and Brooks (1972), as being non-filtrable material of extremely small size ( $< 1 \mu\text{m}$ ) that forms a dispersion of extremely fine particles suspended in the water column in the lakes. This material, due to its ultra small size, allows the supporting forces to exceed the gravitational forces that promote settling such that the material remains in suspension indefinitely. This definition accords with that of Whitten and Brooks. The molecular

composition and origin of this material was not studied in detail, but the very fine, non-filtrable residue likely originates from fine dispersive clays derived from the dolerite soils typical of eastern Tasmania (Crooks 1982).

Measurements of colloidal suspended solids were not possible using standard methods of determination; consequently, the following method was devised. The mass of colloidal material (less than 1  $\mu\text{m}$ ) was quantified by freezing water samples and measuring the total mass of suspended solids of the thawed sample. Freezing resulted in almost complete flocculation of the colloidal material which then was easily retained on a Whatman GFC filter paper. Refer Appendix 4 for an overview of the method and an example of the analysis. The material retained was processed gravimetrically for suspended solids as per Standard Methods (A.P.H.A. 1992). Differences in mass between non-frozen and frozen samples were then used to estimate the mass of colloidal material in suspension.

#### **2.2.3.3 Algal biomass**

Chlorophyll samples were filtered through Whatman GF/F filters within 1-2 hours of collection, frozen in liquid nitrogen and later extracted in 90% acetone. Chlorophyll and phaeopigment analysis was carried out spectrophotometrically according to Standard Methods (A.P.H.A. 1992). Chlorophyll-*a* measurements used for dry weight algal biomass estimates were not adjusted for phaeopigments due to inherent problems in phaeopigment corrections in spectrophotometric techniques (Mantoura, Jeffrey et al. 1997). These problems have the potential to grossly overestimate phaeopigment (Pha) concentration and underestimate chlorophyll-*a* in Pha corrected samples (Webb, Burnison et al. 1992). Algal biomass was estimated from chlorophyll-*a* concentrations assuming a dry-weight/chlorophyll ratio of 70 (Scheffer 1998).

Additionally, spectrophotometrically determined non-Pha corrected chlorophyll-*a* were consistent with techniques employed from previous surveys on lakes Sorell and Crescent (Table 1).

#### **2.2.3.4 Nutrient samples**

The total and filtered nutrients measured were soluble reactive phosphorus, total phosphorus, nitrite, nitrate, ammonia, and total nitrogen. The components chosen were comparable to the species measured in previous surveys undertaken by the IFS and adequately represented the most significant nutrient fractions of interest. Samples were processed by the State Forensic Laboratories according to Standard Methods (A.P.H.A. 1992) (refer Section 2.2.2.2).

#### **2.2.4 Statistical analyses of water quality samples**

Because of the varying sample sizes of the water quality data for the period 1991 – 2002, box-and-whisker plots were initially inspected to determine appropriate transformation of water quality variables and to display any coarse trends. Analysis of variance (ANOVA) was then conducted for each variable in each lake separately and the existence of linear or quadratic trends was tested using a priori contrasts. The assumptions of ANOVA were checked by plotting residuals versus estimates and examining normal probability plots for untransformed or transformed variables as appropriate (Hillman and Quinn 2002).

The relationship between turbidity, suspended solids and euphotic depth (Zeu) was investigated using non-linear regression. A power function was most appropriate for these relationships, and the adequacy of the regressions assessed using the usual methods of inspecting plots of residuals and leverage and influence diagnostics (Harrell 2001; Hillman and Quinn 2002) .

#### **2.2.5 Methods for estimating nutrient budgets**

##### **2.2.5.1 Subcatchments**

The main tributary to the lakes is Mountain Creek to the north of Lake Sorell. There also flows west to east several significant creeks in the vicinity of Silver Plains to the west of Lake Sorell. The dominant tributary of Lake Crescent (excluding Interlaken Canal flowing from Lake Sorell) is Agnews Creek, which flows into the eastern side of Lake Crescent. Overall, the ratio of catchment area to lake surface area is low, with the total catchment area being 1.75 times the combined surface area of both lakes.

Table 3 outlines the surface areas of each of the main tributaries.

Table 3 Subcatchment surface areas (SA) for each of the tributaries entering lakes Crescent and Sorell.

Catchment	SA (km <sup>2</sup> )
Mountain Creek (Lake Sorell)	27.03
North Silver Plains Creek (Lake Sorell)	6.75
Silver Plains Creek (Lake Sorell)	9.69
Kemps Marsh drain (Lake Sorell)	12.97
Dogs Head Creek (Lake Sorell)	8.60
Agnews Creek (Lake Crescent)	7.91
Total catchment area (including lake surface)	205.9

The Mountain Creek catchment to the north of Lake Sorell is dominated by native bushland with some selective logging occurring in the lower end of the catchment in the vicinity of the lake. The majority of the Mountain Creek catchment is relatively pristine. The Silver Plains Creek catchment which includes a small tributary to the north of Silver Plains Creek along with a drain discharging from Kemps Marsh is a mixture of native bushland with some sown pasture, native grasslands, grassy woodlands and wetland areas used for grazing. Agnews Creek had a significant wetland area in the upper catchment which has since been drained to improve grazing and is now utilized for limited sheep and cattle grazing.

Farming has been carried out in the immediate vicinity of the lakes since the 1820s, but some areas previously used for agriculture have reverted to native forest. Sown pasture, native grasslands, grassy woodlands and wetlands are favoured for grazing, with bush runs also being used. Currently approximately 18% of the catchment is used for cattle and sheep grazing, all on private land with the exception of a pastoral lease on Crown land at Robertsons Bay (Gudde 2004).

Due largely to the nature of the terrain, forestry activities are limited to selective logging and thinning operations with no clear-felling. Some logging has been carried out in the past on the State forest on the northern shore of Lake Sorell. Recent forestry operations have been located to the south, southeast and northwest of Lake Crescent and to the east, west and northwest of Lake Sorell (Gudde 2004).

### 2.2.5.2 Stream discharge

Major creeks flowing into lakes Crescent and Sorell were identified. For each inflow, a suitable weir pool was located as close to the lake as possible (Table 4). A suitable weir pool was determined as a pool with as uniform bottom contour as possible with a channel contour that was unlikely to overfill at high water levels. In each, a stage board was securely fastened to give a reference height that extended below the water level under low or base flow conditions. Alongside each stage board, a Dataflow 392 data recorder (Dataflow Services, Trintech Group Plc., Addison, Texas, USA) coupled with a Dataflow capacitance depth probe was installed. Each logger was housed in a PVC stilling-well to reduce the influence of turbulence and waves on depth measurements. Resolution of the probes was approximately 1 mm. Depth loggers were calibrated against the stage boards so that recorded heights from loggers could be compared directly to the stage height. Weir pool height was logged at half-hourly intervals from July 2000 to August 2002.

Table 4      Coordinate locations for inflow and outflow gauging sites for lakes Crescent and Sorell. Co-ordinates as UTM/UPS in Australian Geodetic 1966 (AGD66) projection.

Location	Easting	Northing
Mountain Creek	515 790	5 341 890
Unnamed Creek adjacent to Silver Plains Creek	508 810	5 339 140
Silver Plains Creek	508 490	5 338 010
Drain Above Kemps Marsh	509 460	5 334 990
Drain Below Kemps Marsh	511 680	5 335 570
Lake Sorell Outflow	514 290	5 333 940
Agnews Creek	515 970	5 332 300
Lake Crescent Outflow	512 200	5 331 170

Approximately fortnightly field sampling trips were made during times of significant stream discharge with additional 'event-based' trips made when high flows occurred. During each trip instantaneous discharge was measured with the use of an Ott-C2 (OTT Hydrometry, Kempen, Germany) 50 mm diameter propeller flow meter. Discharge ratings were developed for each site (Appendix 5) by relating measured instantaneous discharge to stage height using linear and non-linear regressions. Estimates of total stream discharge were then made from the continuous record of



stage height obtained by the loggers following standard procedures (Gordon, McMahon et al. 1992).

Loggers were installed at sites in 4 tributaries entering Lake Sorell: Mountain Creek, Silver Plains Creek, a small creek approximately 500 m north of Silver Plains Creek, and the drain flowing out of Kemps Marsh. Agnews Creek flowing into Lake Crescent was sampled regularly, but due to security reasons and site suitability, was not fitted with a logger. Correlation and regression analysis of instantaneous flow with the other gauged creeks was used to estimate total discharge for Agnews Creek (Appendix 5).

### **2.2.5.3 Water sample collection and analysis**

Regular fortnightly sampling trips were undertaken during times of stream flow and additional trips were also undertaken at times of high flow. On each sampling occasion, water samples were taken to measure total suspended solids, total phosphorus and total nitrogen. Samples were treated and analysed the same way as lake water samples.

Four high flow events were sampled using two Sigma 900 automatic water samplers (Hach Company, Loveland, Colorado, USA) deployed on Mountain Creek and Silver Plains Creek. Each sampler had a 24 by 1 litre sample capacity and were fitted with a float switch that tripped at a predetermined stage height. The time that samples were taken was related to recorded stage height. Water samples from the sampler were analysed for suspended solids, total phosphorus and total nitrogen. The data from these event-based samples were used to derive precise loadings for each of these events to compare with the routine water sampling so that errors from the routine sampling could be estimated.

### **2.2.5.4 Loading calculations**

For streams with continuous flow records, instantaneous load ( $L$ ) was calculated as  $L = QC$ , where  $Q$  is the instantaneous discharge in litres per day and  $C$  is the concentration of the species in question in grams per litre, thus giving instantaneous loads in grams per day.

Loading calculations were carried out in two ways. If a significant relationship was found between instantaneous discharge and concentration (Appendix 6) then stream discharge was used to estimate concentration. The continuous record of stream concentration was combined with flow to give a continuous record of stream load. Alternatively, if an insignificant relationship was found between stream flow and concentration, it was assumed that nutrient and sediment concentrations changed linearly between sample dates (Buckney 1979; Cosser 1989; Jeppesen, Jensen et al. 1999).

To determine total load for a time period, instantaneous load was plotted against real time and the area under the curve integrated using the trapezoidal rule, thus estimating the total load for the time period in question (Culley and Bolton 1983).

#### **2.2.5.5 Catchment loading comparison**

A comparison of exports from sub-catchments was made by comparing both the instantaneous nutrient export coefficients and flow-weighted concentrations of each sub-catchment (Buckney 1979). Nutrient export coefficients differ from nutrient loads in that they take into account catchment surface area and are therefore an areal export rate. Areal export rates were then compared with published values from other studies to determine the importance and magnitude of sub-catchment loads.

#### **2.2.5.6 Lake exports**

The volume of water released from Lake Sorell into Lake Crescent was monitored during both the 2000 and 2001 summer periods. For water released from Lake Sorell, discharge measurements were made by the Inland Fisheries Service Tasmania (IFS), using the methods described above.

Detailed information of the volume of water released from Lake Crescent down the Clyde River for the 2000 and 2001 seasons was obtained from the Water Resources Division of the Department of Primary Industries, Water and Environment Tasmania (DPIWE). DPIWE manage a permanent gauging station on the Clyde River approximately 1km downstream from the Lake Crescent outflow (Table 4). The gauging station has a continuous record of discharge from Lake Crescent and is managed independently from the IFS.

An automatic water sampler was deployed at the outflow of each lake to monitor the concentrations of exported suspended sediment, nitrogen and phosphorus. Comparisons were made between export estimates made from automatic water samples taken on two separate occasions at 24 and 48 hour intervals to export estimates made using measured mid-lake concentrations from routine lake sampling trips. Good agreement was found between export estimates made from lake concentration and export estimates made from the automatic water samplers, with Lake Sorell having an average error discrepancy of 15% and Lake Crescent having an average error discrepancy of 20% for the components of interest.

For time periods that the automatic water samplers were not installed, measured mid-lake concentrations from routine lake sampling trips were used to approximate the concentrations of the species in outflows. Linear changes in concentration were assumed between sample dates (Cosser 1989) and export loadings calculated using the methods for tributary stream loadings. This information was used to approximate sediment and nutrient exports from both lakes Crescent and Sorell.

#### **2.2.5.7 Internal fluxes**

Annual internal fluxes of suspended sediment and nutrients were estimated from measurements taken during routine sampling trips. Lake-wide average suspended solids concentration and colloidal solids concentration (approximated from the relationship of colloidal solids and filtered turbidity (Appendix 4)) and mid-lake nutrient concentrations were multiplied by the lake volume for each sample trip to determine the total mass of each species in suspension. A range of water column 'loadings' were determined for the time period in question. The difference between the minimum and maximum (range) was used as an indication of the 'maximum' flux or internal change in the mass of material in suspension. This flux was used as an approximate estimate of the internal loading on the water column. Istvanovics et al. (2004) recognised the importance of sediment resuspension in determining nutrient dynamics, particularly phosphorus dynamics, and stated conventional studies that focussed on phosphorus mobilisation and release from the sediments to be less relevant than studies on dynamics of sediment resuspension. It was therefore assumed

that changes in nutrient and suspended sediment concentrations within lakes Crescent and Sorell due to sediment entrainment would give an estimate of internal loading.

#### **2.2.5.8 Estimates of errors in loading calculations**

Errors in estimates of loads from inflows and outflows were calculated by comparing loads computed from events sampled by the automatic water samplers to loads approximated via interpolation between routine water samples for the time period in question. On average, an error of  $\pm 20\%$  was found.

## **2.3 Results**

### **2.3.1 Long-term trends in water quality**

Figure 3 through to Figure 10 summarise changes in annual average values of the major water quality variables from 1991 in both lakes. An increasing trend from 1991 to 2000 is evident in most cases, with 1998 being the year that values began to increase rapidly. There is a statistically significant increase in turbidity (total and colloidal), chlorophyll-*a* and total phosphorus and nitrogen (all linear contrasts  $P < 0.0001$ , except for total phosphorus in Lake Crescent where  $P = 0.002$ ) for both lakes from the early 1990's through to 2000/2001, at which time values peaked. There was some evidence of non-linear trends for chlorophyll-*a* in both lakes and total phosphorus in Lake Sorell with significant quadratic and/or cubic terms; non-linear behaviour was most marked for turbidity and colloidal turbidity for both lakes with highly significant cubic terms ( $P < 0.0001$ ) (Figure 9 and Figure 10). Interestingly, soluble reactive phosphorus (Figure 5), ammonia (Figure 3), and nitrate (Figure 4) did not change dramatically throughout the period, except for nitrate concentrations in Lake Sorell which increased considerably during 2000 and peaked in 2001 resulting in significant linear ( $P = 0.005$ ) and quadratic ( $P = 0.02$ ) trends (Figure 4).

This increase in nitrate in Lake Sorell is ten to twenty times the average concentrations present from 1991 to 1999. The trend of increased concentrations of total nitrogen (Figure 6) and total phosphorus (Figure 7) were similar for both lakes, with concentrations peaking during 2000/2001 with values substantially higher than those recorded during the 1990's. The mechanism for these increases is unknown, as external inputs of these nutrients were negligible (Section 2.3.4) and little has changed

in the catchment that would have increased inputs. Presumably these increases arose from internal sources such as resuspension.

Increasing trends of chlorophyll-*a* concentrations in both lakes are quite distinct, with strong linear (both  $P < 0.0001$ ) and weaker quadratic (both  $P < 0.05$ ) components. Concentrations in 2000 and 2001 increased up to four times on average in Lake Crescent and three times on average in Lake Sorell from the early 1990s (Figure 8). Interestingly, even in the face of increased concentrations in Lake Sorell, the contrast between Lake Crescent and Lake Sorell in algal biomass still held, with Lake Crescent having approximately ten times the algal biomass of Lake Sorell.

While there were significant and comparable increases in total phosphorus (Figure 7) concentration in both lakes between 1991 and 2002, Lake Sorell did not show the same magnitude of increase in algal biomass as Lake Crescent (Figure 8), with a highly significant difference in the pattern of annual average concentrations of chlorophyll-*a* existing between the two lakes (ANOVA; lake  $\times$  year interaction  $F_{9,215}=3.193$ ,  $P = 0.0001$ ). This further highlights the dissimilarities between the two lakes, especially considering the extremely high values of algal biomass attained in Lake Crescent (Figure 8) despite the reduced light availability (Figure 9) from 2000 on.

The dramatic increases in turbidity (Figure 9) and filtered turbidity (Figure 10), especially in Lake Sorell, are clear. Historically, both lakes had average turbidities  $\sim 10$ -30 NTU, with Lake Crescent usually being more turbid than Lake Sorell; this was attributed to the greater algal biomass in Lake Crescent (Cheng and Tyler 1973a). During 2000 turbidity was up to  $10 \times$  higher in Lake Crescent and up to  $20 \times$  higher than average historic values in Lake Sorell. Over 2002, turbidity in Lake Sorell declined from the extreme values recorded in 2000/2001 (reflected in a significant cubic trend over time,  $P = 0.02$ , which was not present in Lake Crescent,  $P = 0.12$ ), although turbidity values for 2002 were still high compared to the values typical of Lake Sorell prior to the mid 1990s. The record for filtered or colloidal turbidity is shorter, but the patterns are broadly similar to total turbidity, although values appear to remain elevated from 2000 on. High colloidal turbidity is important and poses a

significant management problem, as colloids remain in suspension for long periods which could maintain high background turbidity even with significant reductions in both particulate turbidity and algal biomass.

### **2.3.2 *Light climate***

The increase in turbidity from the early 1990's to the present is extreme (Figure 9). This increase, coupled with persistently low water levels, has resulted in catastrophic changes in the ecology of both lakes. Figure 11 summarises the highly significant inverse power relationship between turbidity and euphotic depth in both lakes. The data used for euphotic depth estimates were determined from measurements of light attenuation made from April 2000 to August 2002 during the current project (Section 2.2.3).

At the high, sustained turbidity values characteristic of both lakes over the period 1999 to 2002, the lowest level of light attenuation reached resulted in a maximum euphotic depth of 30 cm. This effectively prevents the entire lake bed of lakes Crescent and Sorell from obtaining adequate solar radiation to sustain photosynthesis and support plant growth at the sediment surface (Spence 1982).

Figure 3      Box and whisker plots of ‘central lake’ ammonia concentrations in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year. No records were collected in 1993 or 1996 from either lakes Crescent or Sorell.

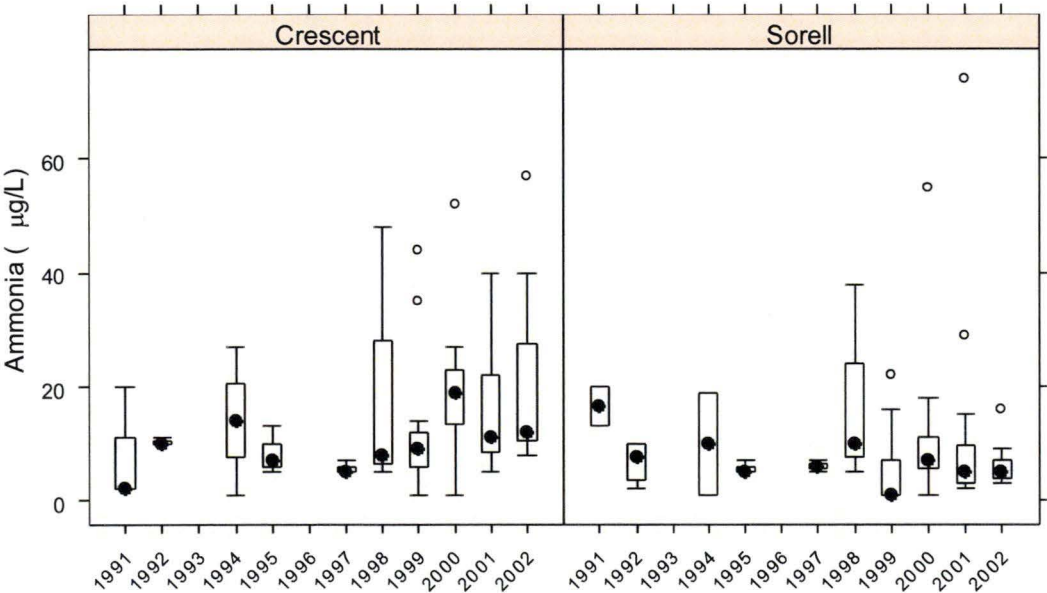


Figure 4 Box and whisker plots of ‘central lake’ nitrate concentrations in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year. No records were collected in 1993 or 1996 from either lakes Crescent or Sorell.

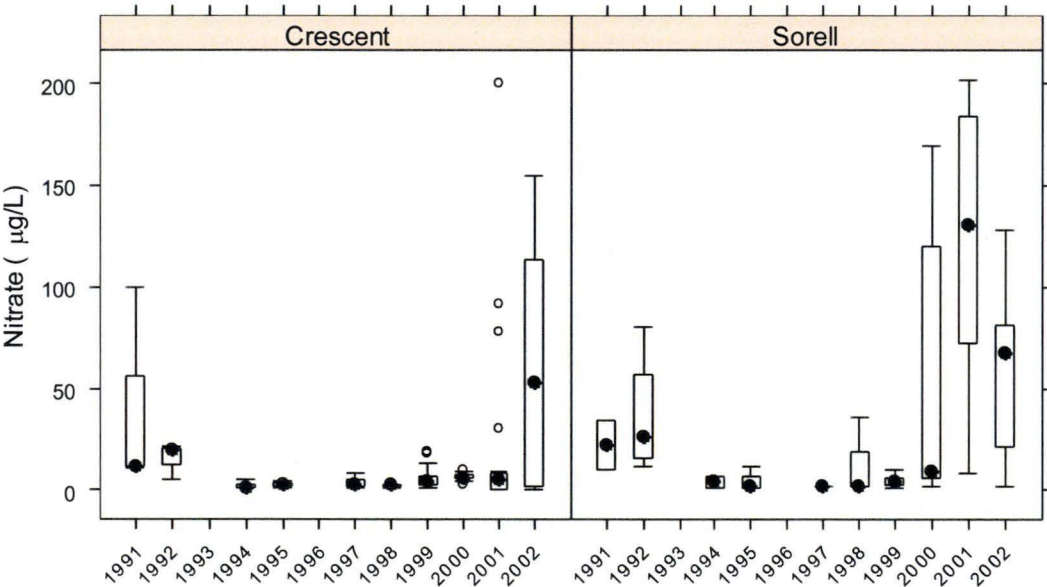




Figure 5 Box and whisker plots of 'central lake' soluble reactive phosphorus concentrations in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year. No records were collected in 1993 or 1996 from either lakes Crescent or Sorell.

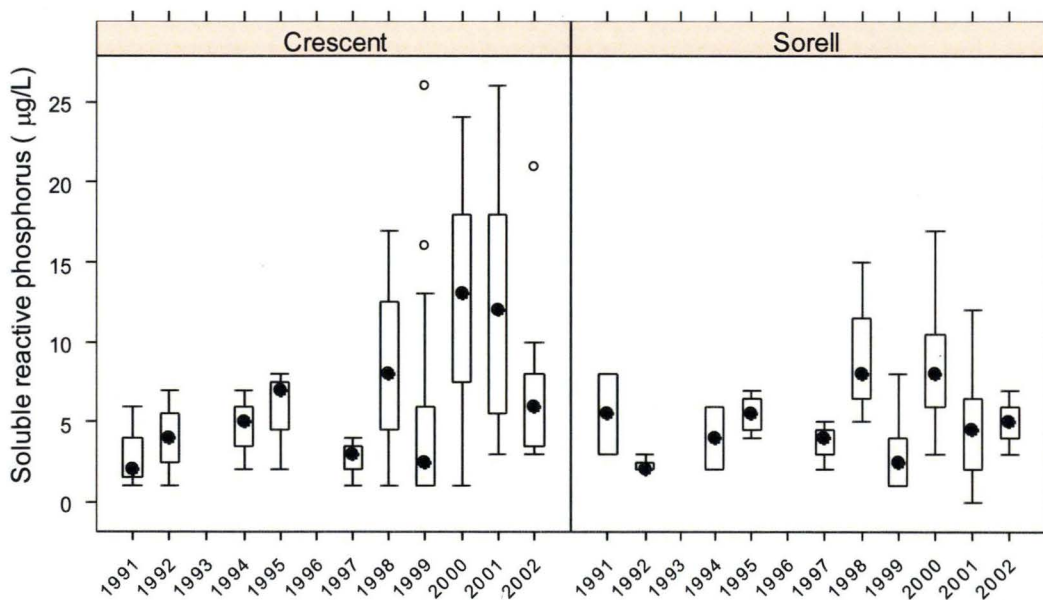


Figure 6 Box and whisker plots of 'central lake' total nitrogen concentrations in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year. Note the logarithmic scale on the ordinate. No records of total nitrogen were collected in 1993 or 1996 from either lakes Crescent or Sorell.

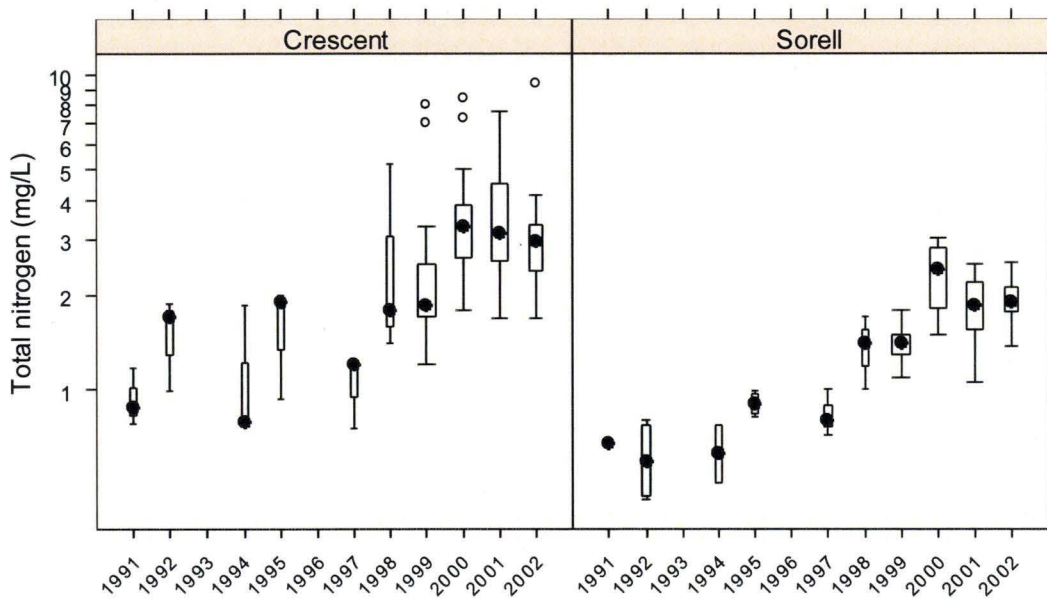


Figure 7 Box and whisker plots of ‘central lake’ total phosphorus concentrations in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year. Note the logarithmic scale on the ordinate. No records of total phosphorus were collected in 1993 or 1996 from either lakes Crescent or Sorell.

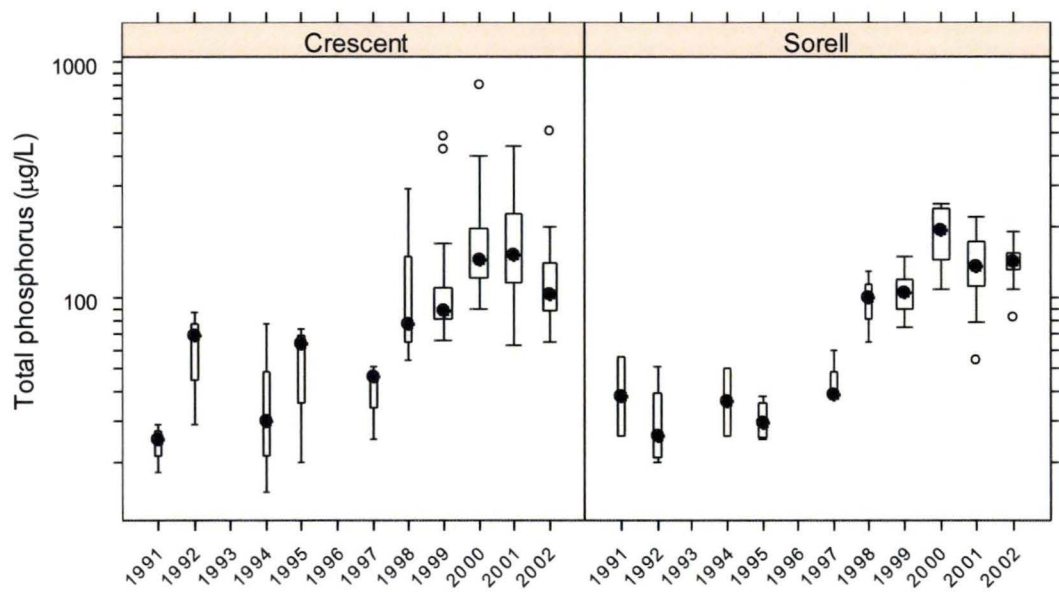


Figure 8 Box and whisker plots of lake-wide chlorophyll *a* concentrations in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year. Note the logarithmic scale on the ordinate. No records of chlorophyll-*a* were collected in 1993 from either lakes Crescent or Sorell, nor in 1998 from Lake Crescent.

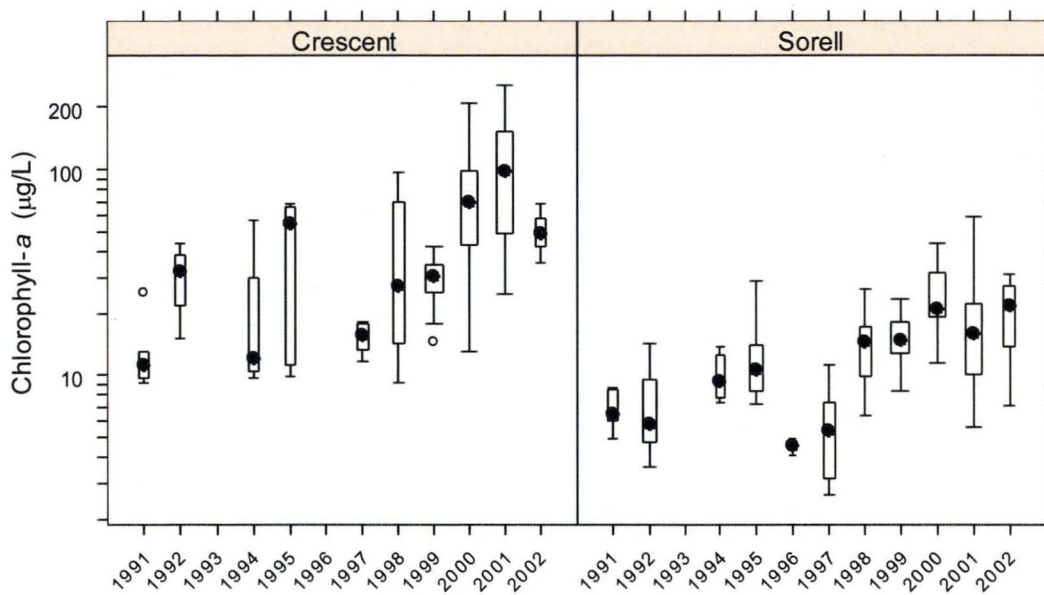


Figure 9 Box and whisker plots of lake-wide turbidity values in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year.

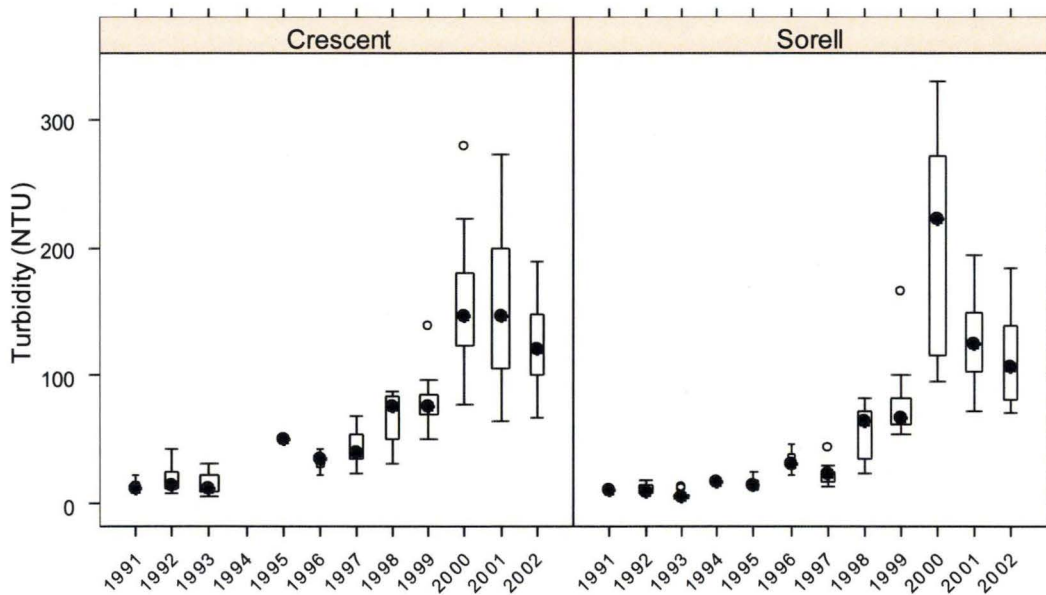


Figure 10      Box and whisker plots of lake-wide filtered turbidity (colloidal turbidity) values in lakes Crescent and Sorell, 1991 to 2002. Dots are the median concentration, and the hinges of the boxes are the first and third quartiles of the data for each year. Width of the boxes are proportional to the sample size in each year. No records of colloidal turbidity were collected prior to 1996 from either lakes Crescent or Sorell.

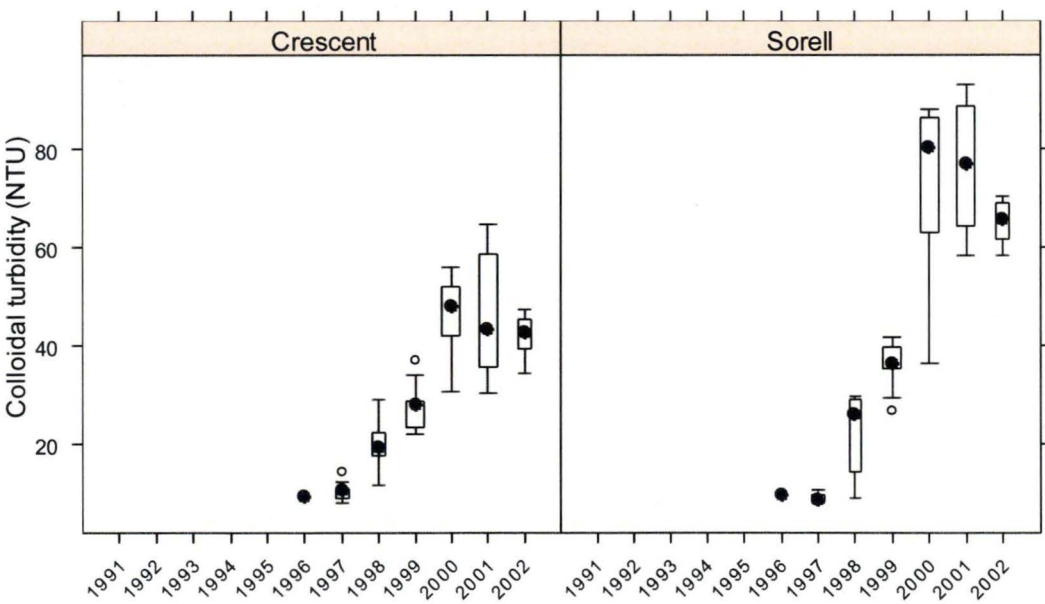
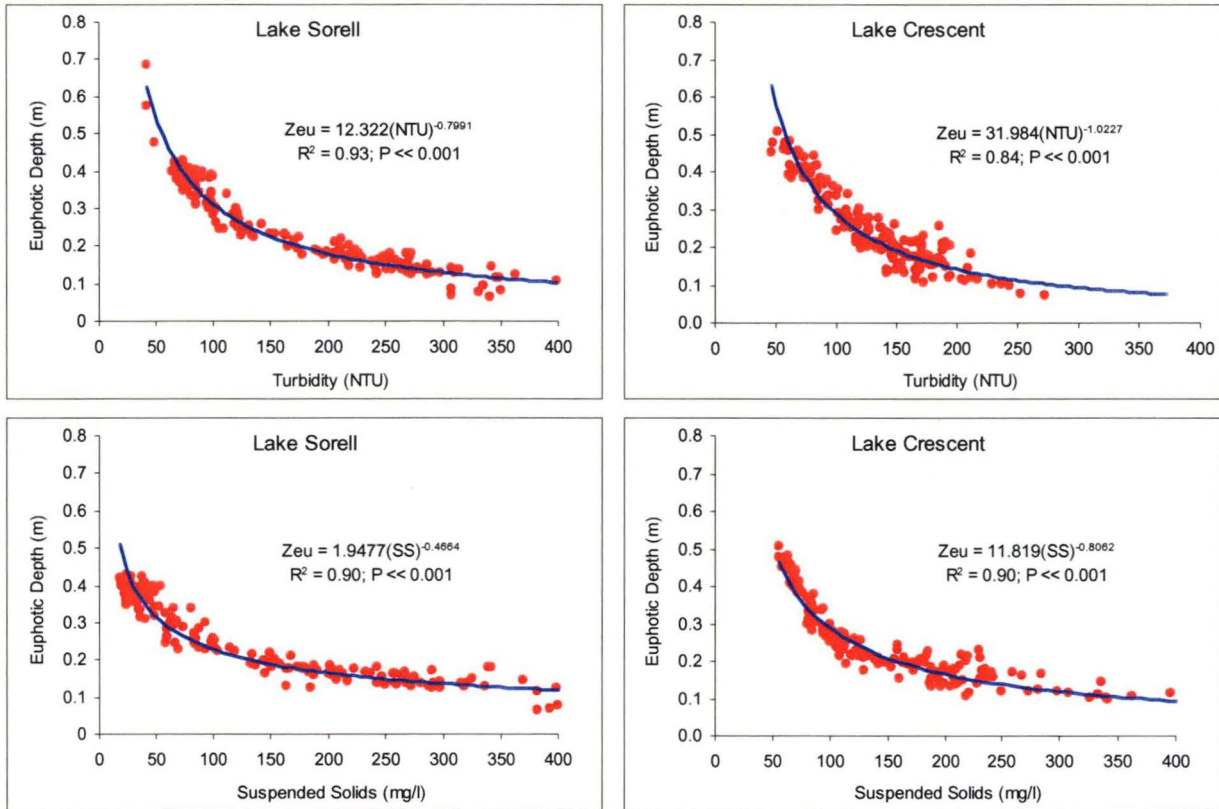




Figure 11 The relationship between euphotic depth (Zeu) in metres, versus turbidity (NTU) and suspended solids (SS) (mg L<sup>-1</sup>) for both lakes.



### 2.3.3 Relationships between turbidity and nutrients, suspended solids and algal biomass

The relationships between turbidity and total nutrients were investigated using all available data collected by the IFS summarised in Table 1. Total nitrogen and total phosphorus are strongly linearly related to turbidity in both lakes (Table 5, Figure 12, Figure 13) with over 80% of variation in total nutrients explained by turbidity.

Moreover, turbidity is strongly related to the concentration of suspended sediment in the water column (Lake Sorell:  $r^2 = 0.94$ ,  $P < 0.001$ ,  $n = 275$ ; Lake Crescent:  $r^2 = 0.94$ ,  $P < 0.001$ ,  $n = 280$ ; Appendix 7). However, Lake Crescent differs from Lake Sorell in that the slopes of the regressions are an order of magnitude greater for both nutrients (Table 5).

Table 5      Linear regression results describing the relationships between total nitrogen, total phosphorus and turbidity for lakes Crescent and Sorell. N is the total number of observations, *P* is the p-value of the regression.

Lake	Dependent Variable	Coefficient (NTU)	Intercept	<i>r</i> <sup>2</sup>	N	<i>P</i>
Lake Crescent	Total N	0.0227	0.3006	<b>0.82</b>	85	<0.001
Lake Crescent	Total P	0.0012	-0.0038	<b>0.82</b>	83	<0.001
Lake Sorell	Total N	0.0075	0.8270	<b>0.81</b>	86	<0.001
Lake Sorell	Total P	0.0007	0.0436	<b>0.82</b>	85	<0.001

Figure 12      Total phosphorus and total nitrogen concentrations versus turbidity for Lake Crescent.

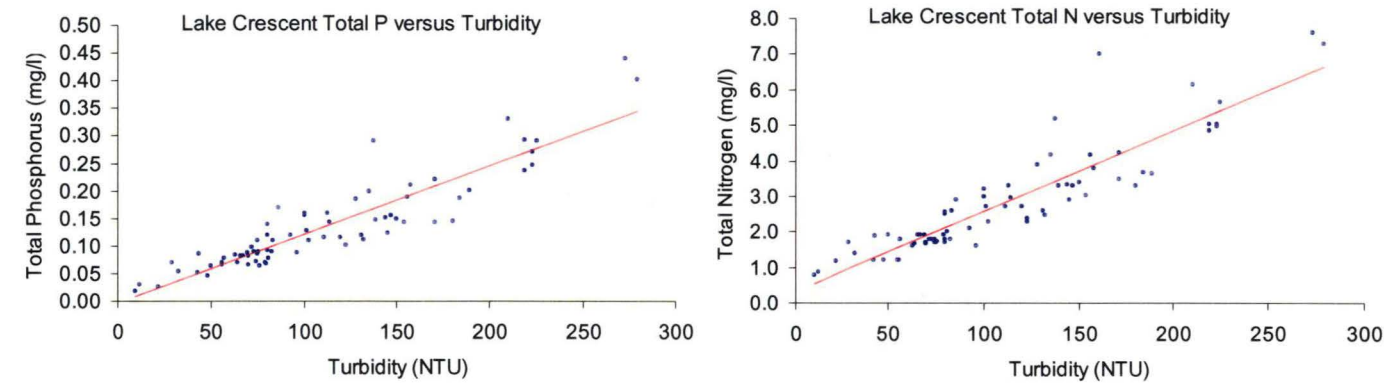
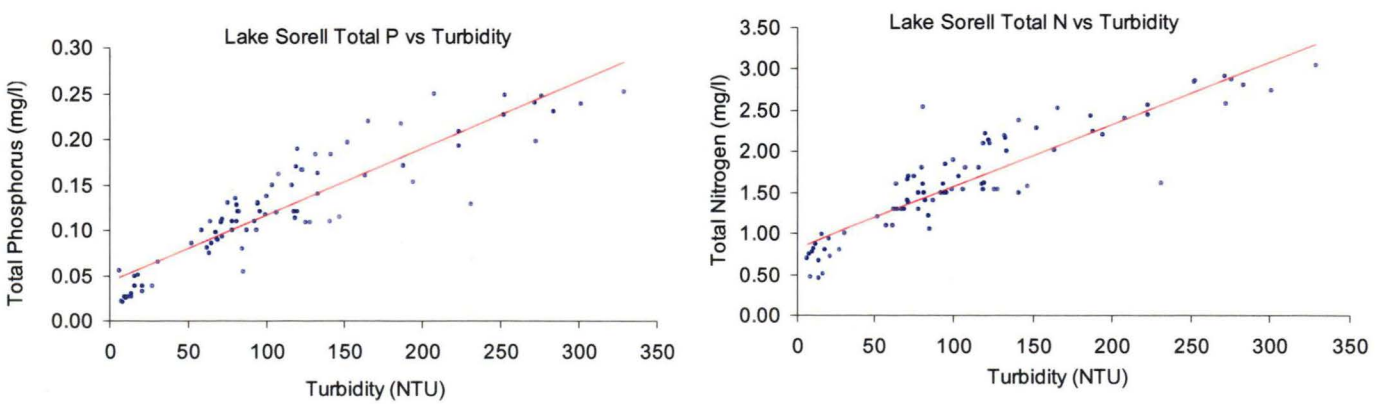


Figure 13      Total phosphorus and total nitrogen concentrations versus turbidity for Lake Sorell.



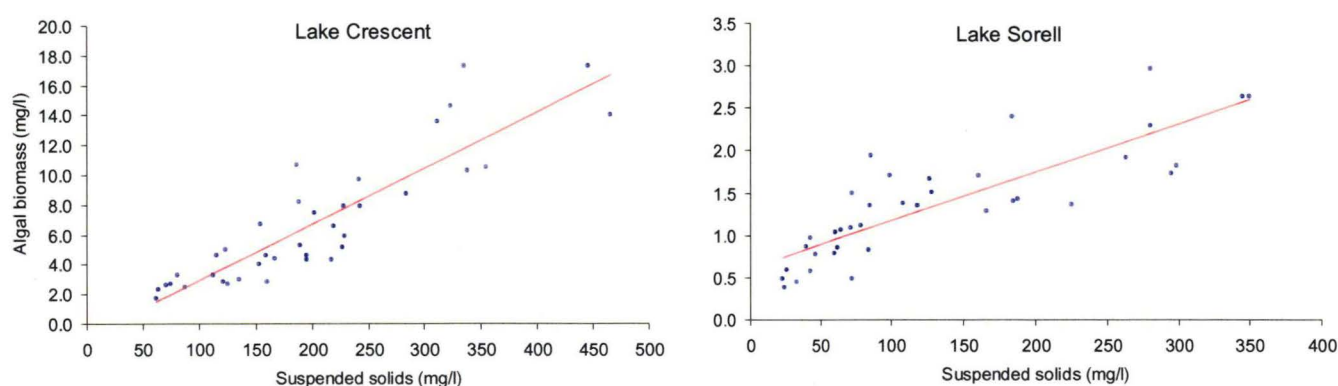


To investigate relationships between phytoplankton biomass and suspended solids, information collected during the course of the current project was used (Section 2.2.2). For each water sample analysed the percent contribution of total algal biomass (as dry weight) was calculated as a percentage of total suspended solids and summarised in Table 6. Phytoplankton biomass is closely related to suspended solids (Table 6, Figure 14), but the percentage contribution of algal biomass (as dry weight) to suspended sediment is generally small, especially in Lake Sorell (Table 6). Note also that the larger regression coefficient for Lake Crescent means that an order of magnitude greater increase in algal biomass results from a comparable increase in suspended sediment compared with Lake Sorell (Table 6, Figure 14).

Table 6 Regression coefficients and the relative contribution of algal biomass to suspended solids for lakes Crescent and Sorell.

Lake	Coefficient	Intercept	% Algal contribution to SS			$r^2$	$N$	$P$
			Min	Average	Max			
Crescent	0.0376	-0.8434	1.7	6.7	17.3	<b>0.79</b>	38	<0.01
Sorell	0.0057	0.6122	0.6	1.3	2.3	<b>0.71</b>	37	<0.01

Figure 14 Relationship between algal biomass (dry weight) and suspended solids for lakes Crescent and Sorell.



### 2.3.4 Nutrient budgets

The results are summarised in Table 7 to Table 11. Table 7 outlines the volume of water entering the lakes from each of the sub-catchments. Evident is the increased water volume for 2001, due largely to increased early autumn and late spring rainfall.

Appendix 5 contain hydrographs for each of the sub-catchments for 2000 and 2001. Mountain Creek has the greatest discharge per km<sup>2</sup>. This may be a result of both greater rainfall in the north of the catchment and greater runoff due to catchment characteristics. Catchment characteristics can have a marked influence on runoff generation by changing key hydraulic aspects of a catchment. Runoff is generated within a catchment through infiltration excess, subsurface storm flow and saturation excess overland flow, all of which are influenced by catchment characteristics and land use and determine the potential discharge capacity of a catchment (Pfister, Kwadijk et al. 2004).

Stream loading is usually the most critical water quality characteristic (Barrett and Loh 1982), especially when the impact on receiving waters is of concern. However, despite their wide acceptance, the use of nutrient export coefficients (NECs) to compare catchment activities rarely take differences in runoff characteristics into consideration (Beaulac and Reckhow 1982; Birch 1982; Culley and Bolton 1983). NECs are used extensively to compare nutrient and sediment loadings between areas of differing land use and also to compare catchment loadings within a localised area (Erskine and Saynor 1995). They also allow a trophic status to be assigned to the catchment (Buckney 1979). The NECs for each tributary were calculated by dividing the total annual load by catchment area, to give export rates as kg km<sup>-2</sup> yr<sup>-1</sup>. The range of export coefficients was substantial, ranging from 250 to 1100 kg km<sup>-2</sup> yr<sup>-1</sup> for suspended sediment (Table 7); 30 to 600 kg km<sup>-2</sup> yr<sup>-1</sup> for total nitrogen (Table 8); and 1.5 to 6 kg km<sup>-2</sup> yr<sup>-1</sup> for total phosphorus (Table 9).

To be able to use nutrient loads to compare the impact of catchment activities, it is necessary to take differing catchment attributes into consideration (Cosser 1989), as sub-catchments may vary in their ability and efficiency to catch and transport water. For example, Mountain Creek (Table 7) has a higher discharge per unit catchment area than the other tributaries. Although not obvious, the increased discharge alone may lead to a disproportionate sediment and nutrient export from this subcatchment. In order to compare the impact of catchment activities on values of sediment and nutrients present within the catchment, it is necessary to take the difference in discharge into account (Cosser 1989). Flow-weighted concentrations achieve this by dividing the total catchment load by the total discharge volume, giving an average

‘flow-weighted’ concentration for the time period in question (in  $\text{mg L}^{-1}$ ). Comparison of flow-weighted concentrations between catchments leads to a better understanding of differences in proportional contributions between the catchments.

Table 7 Total discharge (megalitres) from each of the gauged sub-catchments of lakes Crescent and Sorell. (Refer Appendix 5 for individual hydrographs).

Sub-catchment	SA ( $\text{km}^2$ )	Discharge 2000	Discharge 2001	Discharge $\text{km}^{-2}$ 2000	Discharge $\text{km}^{-2}$ 2001
Mountain Creek	27.03	<b>9924</b>	<b>13609</b>	367	503
Silver Plains Creek	9.69	<b>1990</b>	<b>4180</b>	205	431
Nth Silver Plains Creek	6.75	<b>994</b>	<b>2417</b>	147	358
Kemps Marsh drain	12.97	<b>616</b>	<b>2473</b>	48	191
Agnews Creek	7.91	<b>950</b>	<b>2010</b>	120	254
Dogs Head Creek	8.60	NA	NA	NA	NA

The increased discharge of 2001 resulted in higher total sediment and nutrient loads than in 2000 (Table 8 to Table 10). An exception was Mountain Creek, which experienced a higher total sediment load during 2000 than 2001, even with the increased discharge of 2001 (Table 8). This is likely due to significant earth works undertaken in the lower reaches of Mountain Creek on erosion control during 1999, which may have mobilised a limited amount of sediment that was entrained and discharged during 2000. The level of sediment entering Lake Sorell from Mountain Creek is still extremely low for both 2000 and 2001.

Table 8 shows sediment loadings from tributaries to be very low for both 2000 and 2001. This is further highlighted with comparison of flow-weighted concentrations which fall below  $5 \text{ mg L}^{-1}$ , with the exception of suspended solids draining from Kemps Marsh for 2000. The sediment export coefficients (NECs) for the subcatchments were all  $< 1000 \text{ kg km}^{-2} \text{ yr}^{-1}$  with the exception of the small tributary to the north of Silver Plains Creek, which had an estimated load marginally greater than  $1000 \text{ kg km}^{-2} \text{ yr}^{-1}$  for 2001. Sediment NECs were very low with an average of  $425 \text{ kg km}^{-2} \text{ yr}^{-1}$  for 2000 and  $690 \text{ kg km}^{-2} \text{ yr}^{-1}$  for 2001. The increased load for 2001 can be attributed to the increased discharge from each of the subcatchments for 2001 (Table 7). This is further supported by the flow weighted concentrations which are all lower for 2001 than 2000 (Table 8).

The sediment NECs are all extremely low when compared with export coefficients from other studies from mainland Australia and Tasmania, with Erskine and Saynor (1995) finding an average sediment load of  $10\,000\text{ kg.km}^{-2}.\text{yr}^{-1}$  for a number of Australian rivers with grazing as the predominant land use. Loughran (1977) estimated average annual sediment yields from a small rural catchment in the Hunter Valley, NSW to be  $28\,000\text{ kg km}^{-2}\text{ yr}^{-1}$ ; and Olive (1982) lists sediment yields from 16 subcatchments of the Murray-Darling basin ranging from  $1\,500\text{ kg km}^{-2}\text{ yr}^{-1}$  to  $67\,000\text{ kg km}^{-2}\text{ yr}^{-1}$ , with an average of  $12\,000\text{ kg km}^{-2}\text{ yr}^{-1}$ . Within Tasmania, Olive (1975) found sediment NECs for three catchments in the south-east to range from  $8\,700\text{ kg km}^{-2}\text{ yr}^{-1}$  to  $10\,000\text{ kg km}^{-2}\text{ yr}^{-1}$  with native forest with limited forest harvesting being the dominant land use for the areas studied.

Interestingly there is substantial fluctuation in sediment NECs between years and between catchments. The increased NEC of  $1\,090\text{ kg km}^{-2}\text{ yr}^{-1}$  calculated for North Silver Plains Creek for 2001 may result from this stream being channelised for the lower few kilometres before it enters Lake Sorell. Whilst sampling this small stream, a significant amount of frost heave was observed during 2001 along the banks along with a large amount fine clay sediment that was deposited along the creek bed. This material was easily dispersed when disturbed. This contrasts sharply with Silver Plains Creek, located approximately 1 km south, that had a creek bed dominated by gravel and pebbles with little evidence of channelisation or disturbance. The increased sediment loading from North Silver Plains Creek may be due to a combination of increased flows for 2001 and erosion and entrainment of material from the creek bed and banks.

Kemps Marsh drain and, to a lesser extent, Agnews Creek also had comparably high sediment export coefficients compared to the other catchments, although as mentioned previously, the level of sediment load is extremely low compared to other mainland Australia and Tasmanian studies. The increased sediment load from Kemps Marsh drain is also reflected in the high flow weighted concentrations, especially in the case of 2000. This may be due to Kemps Marsh being exposed for several years preceding 2000 (Heffer 2003), at which time significant rains resulted in discharge from the marsh. It is possible that export of sediment from the marsh increased due to

inundation following a prolonged period of desiccation (Baldwin and Mitchell 2000). This phenomena may also explain the marginally higher flow weighted concentration of nitrogen and phosphorus measured in the Kemps Marsh drain for 2000 and 2001 (Table 9 and Table 10) (Baldwin and Mitchell 2000).

Table 8 Total annual sediment loadings (in kg), area-specific loadings (NEC's) ( $\text{kg km}^{-2}$ ) and flow weighted concentrations ( $\text{mg L}^{-1}$ ) from each of the gauged sub-catchments of lakes Crescent and Sorell.

Sub-catchment loads	Total load		Area-specific NEC		Flow-weighted	
	2000 (kg)	2001 (kg)	2000 ( $\text{kg km}^{-2}$ )	2001 ( $\text{kg km}^{-2}$ )	2000 ( $\text{mg L}^{-1}$ )	2001 ( $\text{mg L}^{-1}$ )
Mountain Creek	14720	6666	545	247	1.48	0.49
Silver Plains Creek	2560	4800	260	500	1.29	1.15
Nth Silver Plains Creek	3120	7390	460	1090	3.14	3.06
Kemps Marsh drain	5535	11079	427	854	8.99	4.48
Agnews Creek	3414	6036	431	763	3.59	3.00
<b>Total Sediment input (kg)</b>	<b>29350</b>	<b>35970</b>				

Nitrogen loadings show an interesting pattern (Table 9) with nitrogen loadings and flow weighted concentrations being low for all tributaries except Agnews Creek. Using the trophic classification system of Buckney (1979) all tributaries may be termed oligotrophic, as they have loads  $< 500 \text{ kg km}^{-2} \text{ yr}^{-1}$ . The higher load from Agnews Creek degrades the classification for this tributary to being mildly 'mesotrophic'. Comparisons of flow weighted concentrations further underline the contrast, with nitrogen values in Agnews Creek being significantly higher than the Australian and New Zealand Environment and Conservation Council (A.N.Z.E.C.C. 1992) guideline maximum for rivers and streams of  $0.75 \text{ mg L}^{-1}$ . The higher loading of nitrogen from Agnews Creek has the potential to influence internal nitrogen stores in Lake Crescent, as a net gain in nitrogen was evident in Lake Crescent during 2001 (Table 11). Elimination of Agnews Creek as a significant source of nitrogen would lead to a net loss of nitrogen from Lake Crescent under current conditions. The drain discharging from Kemps Marsh also has marginally elevated flow weighted concentrations in comparison to the other catchments, although the areal loadings for this tributary are relatively low.

The water quality for all inputs as measured by NECs can be deemed to be of high quality, with the exception of nitrogen loads entering Lake Crescent from Agnews Creek (Table 9). Average NECs of nitrogen (Table 9) compare favourably with mainland Australian studies. For example, nitrogen export coefficients calculated by Buckney (1979) for six small subcatchments of the Onkaparinga River, South Australia were between  $475 \text{ kg km}^{-2} \text{ yr}^{-1}$  and  $55\,385 \text{ kg km}^{-2} \text{ yr}^{-1}$ . If the extreme outlier is excluded then the range is between  $475 \text{ kg km}^{-2} \text{ yr}^{-1}$  and  $3\,650 \text{ kg km}^{-2} \text{ yr}^{-1}$  with an average of  $2\,120 \text{ kg km}^{-2} \text{ yr}^{-1}$ . Although Buckney's (1979) work was carried out on heavily cultivated soils. Nitrogen export coefficients listed by Beaulac and Reckhow (1982) for forested catchments in the USA are comparable with the nitrogen loads of tributaries entering lakes Crescent and Sorell. A study by Quinn and Stroud (2002) in New Zealand investigated nutrient exports from areas of contrasting land use and lists nitrogen export coefficients of  $83 \text{ kg km}^{-2} \text{ yr}^{-1}$  and  $152 \text{ kg km}^{-2} \text{ yr}^{-1}$  for two catchments dominated by native forest compared to  $565 \text{ kg km}^{-2} \text{ yr}^{-1}$  and  $1\,395 \text{ kg km}^{-2} \text{ yr}^{-1}$  for two catchments dominated by grazing pasture. The nitrogen loads entering lakes Crescent and Sorell all compare favourably with Quinn and Stroud's (2002) forested New Zealand catchments.

The exception is Agnews Creek, with over an order of magnitude increase in nitrogen loading in some instances compared to the other tributaries entering Lake Sorell. The magnitude of the nitrogen load from Agnews Creek compares closely to impacted catchments from mainland Australia and overseas. The level of nitrogen entering Lake Crescent from Agnews Creek is of concern as it has the capacity to increase nitrogen stores in Lake Crescent (Table 11).

Table 9 Annual nitrogen loadings and flow weighted concentrations from each of the gauged sub-catchments of lakes Crescent and Sorell.

Sub-catchment loads	Total load		Area-specific NEC		Flow-weighted	
	2000 (kg)	2001 (kg)	2000 (kg km <sup>-2</sup> )	2001 (kg km <sup>-2</sup> )	2000 (mg L <sup>-1</sup> )	2001 (mg L <sup>-1</sup> )
Mountain Creek	767	1332	28	50	0.08	0.10
Silver Plains Creek	410	820	40	85	0.21	0.20
Nth Silver Plains Creek	275	660	40	100	0.28	0.27
Kemps Marsh drain	1112	2001	86	154	1.81	0.81
Agnews Creek	4763	4543	602	574	5.01	2.26
<b>Total Nitrogen input (kg)</b>	<b>7330</b>	<b>9360</b>				

The range of phosphorus export coefficients for subcatchments was relatively low ranging from 1.5 to 3.7 kg km<sup>-2</sup> yr<sup>-1</sup> for 2000 to 3.2 to 6.1 kg km<sup>-2</sup> yr<sup>-1</sup> for 2001 (Table 10). The increased load for 2001, as observed with both sediment and nitrogen loading, may be attributed to the increased discharge of 2001. Cullen and O'Loughlin (1982) recognised discharge to be the predominant determinant of phosphorus load and to be more important than concentration. Reliable discharge data is therefore essential for determining phosphorus exports accurately.

Estimated phosphorus export coefficients for the tributaries entering lakes Crescent and Sorell were extremely low when compared to mainland Australian studies. For example, Buckney (1979) calculated phosphorus NEC's for six subcatchments of the Onkaparinga River, South Australia, all of which were extensively cultivated and reported values ranging from 36 kg km<sup>-2</sup> yr<sup>-1</sup> to 144 kg km<sup>-2</sup> yr<sup>-1</sup>, while Cullen and O'Loughlin (1982) listed 35 values of NECs for total phosphorus transport from mainland Australian catchments which ranged from 2 kg km<sup>-2</sup> yr<sup>-1</sup> to 106 kg km<sup>-2</sup> yr<sup>-1</sup>, with an average of 39 kg km<sup>-2</sup> yr<sup>-1</sup>. Table 10 shows that all tributary NEC's for total phosphorus entering lakes Crescent and Sorell fall at the bottom end of this range.

Cullen and O'Loughlin (1982) concluded that forested areas export between 1.0 kg km<sup>-2</sup> yr<sup>-1</sup> and 20 kg km<sup>-2</sup> yr<sup>-1</sup>. Native forest and grazing are the dominant land uses within the lakes Crescent and Sorell catchment, and the measured phosphorus export coefficients are similar to those typical of forested and low impacted

catchments in southern Australian. Similarly, Campbell (1978) estimated a phosphorus export of  $2.4 \text{ kg km}^{-2} \text{ yr}^{-1}$  for a relatively pristine, forested catchment in Victoria, an export rate comparable to the rates estimated for the current study.

Table 10 shows that phosphorus loadings are generally low, and would be termed oligotrophic using Buckney's (1979) criterion because all loads  $<< 19.7 \text{ kg km}^{-2} \text{ yr}^{-1}$ . The flow weighted concentrations are also very low. Using the nutrient status classification system from the State of the Environment Report 1988 (Commissioner for the Environment, 1989), Mountain Creek would be classed as of 'excellent' quality, with the remaining tributaries having 'good' water quality. The exception is Kemps Marsh drain during 2000 and Agnews Creek for 2000 and 2001, which have 'moderate' phosphorus concentrations according to this classification.

The slightly greater flow weighted phosphorus concentrations for these two subcatchments may be due to both draining wetlands which, after a period of desiccation, may lead to increased levels of phosphorus in water draining these areas (Baldwin and Mitchell 2000). Additionally, both Kemps Marsh and the marsh area at the headwaters of Agnews Creek are used for grazing over the drier months of October through to April, which would lead to an increase in animal faeces coupled with mechanical damage of the area that may further lead to increased transport of phosphorus when the wetland areas become inundated sufficiently for overland runoff to occur.

Table 10 Annual phosphorus loadings and flow weighted concentrations from each of the gauged sub-catchments of lakes Crescent and Sorell.

Sub-catchment loads	Total load		Area-specific NEC		Flow-weighted	
	2000 (kg)	2001 (kg)	2000 ( $\text{kg km}^{-2}$ )	2001 ( $\text{kg km}^{-2}$ )	2000 ( $\text{mg L}^{-1}$ )	2001 ( $\text{mg L}^{-1}$ )
Mountain Creek	76	67	2.8	2.5	0.008	0.005
Silver Plains Creek	21.4	43	2.2	4.4	0.011	0.010
Nth Silver Plains Creek	14.4	34.3	2.1	5.1	0.014	0.014
Kemps Marsh drain	20	41	1.5	3.2	0.032	0.017
Agnews Creek	29	48	3.7	6.1	0.031	0.024
<b>Total Phosphorus input (kg)</b>	<b>161</b>	<b>233</b>				



The data collected on nutrient loading from each of the subcatchments was combined with information collected on internal fluxes in sediment and nutrient values and used to develop lake-wide nutrient and sediment budgets for both lakes for 2000 and 2001 (Table 11). A detailed breakdown of each subcatchments inputs into Lake Sorell and Lake Crescent, and exports from both lakes for each of the components listed in Table 11 is given in Appendix 8.

Table 11, shows that internal loading driven by sediment resuspension, estimated as the flux in minimum and maximum lake-average concentrations for the period in question (Section 2.2.5), is the most significant mechanism influencing water quality in these lakes. For example, inputs of suspended sediment into Lake Sorell amounted to 0.08% and 0.17% of the total flux in internal suspended sediment concentration for 2000 and 2001 respectively, 2% and 5% of the total internal flux in nitrogen for 2000 and 2001, and 1% and 1.5% of the total internal flux in phosphorus. For Lake Crescent, the contrast was reduced due to increased inputs of material entering Lake Crescent from Lake Sorell Appendix 8. The change in mass of internal sediment and nutrient concentrations still outweighs the mass of inputs significantly. For example, inputs of suspended sediment into Lake Crescent amounted to 11% and 20% of the total flux in internal suspended sediment concentration for 2000 and 2001 respectively, 10% and 21% of the total internal flux in nitrogen for 2000 and 2001, and 7% and 8% of the total internal flux in phosphorus. It is therefore likely that the increased concentrations of suspended sediment, nitrogen and phosphorus in both lakes is not due to high loadings from subcatchments and is driven by internal processes present in the lakes.

The low level of material entering the lakes from the tributaries (with the exception of water released from Lake Sorell into Lake Crescent), coupled with high values of internal loading, resulted in a significant loss of nitrogen, phosphorus and suspended sediment from both lakes. As an example, the sum of particulate and colloidal sediment (total sediment) exported from Lake Sorell was 30 and 50  $\times$  the mass of material imported for 2000 and 2001 respectively. Nitrogen exports were 1.8 and 2.8  $\times$  the amount imported for 2000 and 2001, and phosphorus exports 2.7 and 1.8  $\times$  the mass imported for 2000 and 2001 respectively. Under current conditions of

increased suspended sediment and nitrogen and phosphorus levels in Lake Sorell, there is a net loss of material from the lake through water release.

For Lake Crescent, a net loss of material for 2000 and 2001 was evident with the exception of nitrogen and total suspended sediment for 2001. The magnitude of loss from Lake Crescent was less than from Lake Sorell due to high sediment and nutrient loadings entering Lake Crescent from Sorell. For example, the sum of particulate and colloidal sediment (i.e. total sediment) exported from Lake Crescent was twice the mass of material imported for 2000 compared to ~ 50% retention of inputs for 2001. Comparably, nitrogen exports were  $1.8 \times$  the input for 2000 compared to a 30% retention of nitrogen in Lake Crescent for 2001.

In the case of phosphorus, a net loss for both 2000 and 2001 was estimated with exports being  $2.2$  and  $1.4 \times$  the mass imported for 2000 and 2001 respectively. The degradation in water quality in Lake Sorell results in significant sediment and nutrient loads entering Lake Crescent from Lake Sorell. Under such conditions, there is the potential for a net gain in material in Lake Crescent, particularly in years that irrigation release from Lake Crescent is largely supplemented by water release from Lake Sorell, as was the case in 2001. This is particularly the case with nitrogen, as the high load entering Lake Crescent from Agnews Creek exacerbates the significant load of material entering from Lake Sorell.

Table 11 Summary of total sediment and nutrient inputs and exports for Lake Sorell and Lake Crescent for 2000 and 2001. For a more detailed overview refer Appendix 8.

	Lake (year)	Inputs (tonnes)	Internal Change (tonnes) (+/-)	Exports (tonnes)	Net Change (tonnes)
Nitrogen	Sorell (2000)	2.6	135.0	4.7	-2.1
	Crescent (2000)	9.4	94.0	16.8	-7.4
	Sorell (2001)	4.8	94.0	12.6	-7.8
	Crescent (2001)	17.1	82.0	12.3	<b>+4.8</b>
Phosphorus	Sorell (2000)	0.13	13.95	0.35	-0.22
	Crescent (2000)	0.38	5.65	0.82	-0.44
	Sorell (2001)	0.19	12.65	0.35	-0.16
	Crescent (2001)	0.40	5.00	0.54	-0.14
Particulate Sediment	Sorell (2000)	26	23150	242	-217
	Crescent (2000)	246	6350	1062	-816
	Sorell (2001)	30	12250	280	-250
	Crescent (2001)	286	5100	441	-155
Colloidal Sediment	Sorell (2000)	0	9950	564	-564
	Crescent (2000)	564	1100	483	<b>+81</b>
	Sorell (2001)	0	5400	1180	-1180
	Crescent (2001)	1180	1350	360	<b>+820</b>
Total Sediment	Sorell (2000)	26	33100	800	-774
	Crescent (2000)	803	7450	1545	-742
	Sorell (2001)	30	17650	1461	-1431
	Crescent (2001)	1467	7450	802	<b>+666</b>

## 2.4 Discussion

### 2.4.1 *Implications of the changed light climate*

The extreme light attenuation in Lake Sorell is likely a key factor in the disappearance of aquatic macrophytes as similar levels of light attenuation has been implicated in the disappearance and limitation of aquatic macrophytes elsewhere (Carter, Rybicki et al. 1996), with the underwater light climate determining the distribution, abundance and primary production of aquatic macrophytes (Van Duin, Blom et al. 2001) and the degree of light penetration limiting the maximum depth to which aquatic macrophytes can grow (Best, Buzzelli et al. 2001). Light limitation by increased absorption and scattering by suspended particulate material and dissolved colour therefore has the potential to reduce the area of a lake colonisable by aquatic macrophytes.

For example, in Lake Apopka, Florida, increased light attenuation resulting from increased sediment resuspension and algal biomass is believed to be responsible for the wide scale loss of aquatic macrophytes, with Lowe et al. (2001) using measurements of PAR and a compensation depth of 1% of surface radiation to conclude light limitation of macrophytes to occur over 95% of the lake bottom.

Similarly, Chambers and Kalff (1985) found a highly significant correlation between Secchi disc depth and the maximum depth to which angiosperms colonised for 90 lakes located in Quebec and elsewhere the world.

In the case of lakes Crescent and Sorell, the increase in turbidity that occurred in 1998 and 1999 (Figure 9) reduced the euphotic depth to < 0.4 m (Figure 11), the continued increase in turbidity during 2000 resulted in a further reduction of the euphotic depth to < 0.2 m. The acute increase in light attenuation experienced by both lakes Crescent and Sorell, coupled with increased physical disturbance at the sediment surface from increased wave action that resulted from reduced lake levels reached during 1998, 1999, 2000 and 2001 (Chapter 3, Section 3.3.3) (Jupp and Spence 1977), would be sufficient to explain the loss and continued absence of aquatic macrophytes from the lakes. The unprecedented levels of sediment erosion and the increase in areas of both

lakes Crescent and Sorell in contact with the wave base or wave mixing zone (Chapter 3, Section 3.3.3, Figure 31 and Figure 32) would limit the minimum depth to which macrophytes could colonise by physically uprooting plants, displacing seedlings and transporting propagules (Chambers and Kalff 1985). This was illustrated by Chambers (1985) who found a strong and highly significant correlation between the minimum depth that macrophytes colonise and the depth of surface wave mixing in four lakes in southern Quebec.

Taking into account the current level of light attenuation, re-establishment of aquatic macrophytes would be highly dependent on a major improvement in the light climate of both lakes. The strong exponential relationship of euphotic depth with turbidity and suspended solids reflects the significant influence a small change in turbidity and suspended solids has on limiting light availability. Reducing turbidity to values well below 50 NTU will be necessary before a significant improvement in light availability results (Figure 11).

#### ***2.4.2 The relationship between turbidity, algal biomass and nutrients***

The processes leading to increased turbidity and the mechanisms controlling nutrients are closely related in these two lakes (Section 2.3.3). Therefore, as the concentration of suspended sediment increases, resulting in increased turbidity, the concentrations of both total phosphorus and total nitrogen also increase. This suggests that the mechanism responsible for increased SPM also results in increased nutrient loading of the water column. This is not surprising for phosphorus, as its transport is closely related to the transport of suspended sediment (Ahl 1988) because of its strong affinity for binding to the surface of inorganic suspensoids (Holtan, Kamp-Nielsen et al. 1988).

Similarly, Hamilton and Mitchell (1988) found increased water column phosphorus concentrations to be highly correlated with sediment resuspension from wind driven waves in Lake Waipori, New Zealand. This, coupled with Sondergaard et al.'s (2003) generalisation that the pool of phosphorus in lake sediments is often more than  $100 \times$  that of the water column concentration, highlights the significance of sediment-water interactions in influencing nutrient concentrations in shallow lakes.

The increase in total phosphorus and nitrogen concentrations represents a significant increase in the trophic status of both systems. With phosphorus being identified as the main limiting nutrient in both lakes (Cheng and Tyler 1976a), its increased concentration may be stimulating algal growth. However, it is difficult to be certain of this link here because biologically available phosphorous may remain strongly sorbed to the sediment surface and unavailable for biological uptake (Laenen and LeTourneau 1996). This is especially so in systems with a significant colloidal contribution to the SPM, as is the case with lakes Crescent and Sorell. The charged nature of the colloidal material (Drever 1997; Langmuir 1997) makes it an effective phosphorus scavenger and colloids are therefore closely linked with the transport of nutrients (Broberg and Persson 1988). For example, Oliver and Hart (1993) found phosphorus bound up in the colloidal fraction (3nm – 1µm) to account for 36% to 40% of total phosphorus. This is directly comparable to the contribution of colloidal phosphorus and nitrogen to the total nutrient pool in lakes Crescent and Sorell (Appendix 9).

The low concentrations of measured soluble reactive phosphorus (Figure 5) compared to the high concentrations of measured total P (Figure 7) support Laenen's view. Alternatively, it is possible that released bio-available phosphorus (SRP) is scavenged immediately by algal cells or sorbed back on to inorganic suspensoids so that measured values remain relatively low, despite a considerable sink of SRP being available (Bostrom, Persson et al. 1988; Nolan, Lawrance et al. 1995).

Understanding the relevance of increased phosphorus concentrations on stimulating algal growth in both lakes Crescent and Sorell would be beneficial in discerning the factors responsible for observed increases in algal biomass. Further research would be needed to determine the amount of bio-available phosphorus measured in the total phosphorus pool and to conclude if increased nutrient levels have reduced nutrient limitation of the algae.

Whatever the precise relationships between SRP and algal biomass, it is clear that in these lakes phytoplankton biomass is closely linked to fluctuations in suspended solids (Table 6, Figure 14). It might be argued that the increased algal biomass may be

responsible for the observed increase in suspended solids and turbidity and hence reduction in aquatic macrophytes in the lakes (e.g. Lowe, Battoe et al. 2001), and that this increased algal biomass results from the increased concentrations of limiting nutrients in the lakes (Section 2.3.1) (e.g. Lowe, Battoe et al. 2001). However, this mechanism is unlikely given that algal biomass never contributed more than 18% of total suspended solids and averaged less than 7% in both lakes.

In the case of lakes Crescent and Sorell, despite a strong correlation between algal biomass and suspended solids (Figure 14), the contribution of algae to SPM was low. Algae in lakes Crescent and Sorell would therefore contribute little to the extreme values of light attenuation routinely measured in both lakes between 2000 and 2002. This is supported by Van Duin et al. (2001) who states that the largest contributor to light attenuation over a short time scale to be suspended solids, especially in the case of wind exposed shallow lakes.

Thus I conclude that the documented increase in phytoplankton biomass in both lakes Crescent and Sorell is more likely to be associated with the extreme increases in values of SPM rather than the agent responsible for increased SPM and the observed reduction in the euphotic depth. This is similar to the conclusions of Bachmann et al. (1999) in Lake Apopka, Florida, who argued that the high concentration of phytoplankton evident in the lake since a switch from a clear to turbid state during the 1940s was not responsible for the high values of turbidity and the reduction in the euphotic depth as only about 10% of suspended particles represent living algal cells. Bachman et al. (2001) concluded a significant reduction in phytoplankton through proposed phosphorus removal in Lake Apopka would fail to significantly effect water transparency as phytoplankton made up such a small proportion of the SPM.

This was contrary to Lowe et al. (2001) who used relationships between algae and the light attenuation coefficient in Lake Apopka to predict increases in light availability based on reductions in algal standing crop alone. However, their approach failed to account for the contribution that inorganic suspensoids, detritus and dissolved colour make to light attenuation (Van Duin, Blom et al. 2001).

A strong relationship between algal biomass and the light attenuation coefficient may occur if the process leading to increased algal standing crop also leads to increased SPM, such as may be the case with sediment resuspension and entrainment of meroplankton. Other researchers have reached similar conclusions that phytoplankton biomass increases as a result of suspension of sediments in the water column (Demers, Therriault et al. 1987; Carrick, Aldridge et al. 1993; Ogilvie and Mitchell 1998) with Schelske et al. (1995) observing a doubling of chlorophyll-*a* concentration during a sediment resuspension event along with demonstrating a significant correlation between chlorophyll-*a* concentration and wind speed.

Establishing causation between increased algal biomass and the interplay between suspended sediment and nutrient loading is important in recognising appropriate management initiatives to improve the light climate of both lakes. As discussed, turbidity appears little influenced by seston in the face of high values of suspended sediment loading (Table 6) despite a significant relationship between suspended solids and algal biomass (Figure 14). A reduction in tripton values would improve PAR availability and, coupled with the loss of aquatic macrophytes, may stimulate primary productivity and result in a significant increase in algal standing crop. This may occur independent of the mechanisms that currently give rise to high algal biomass and may lead to a sustained increase in algal standing crop further limiting light availability and coupled with the loss of aquatic macrophytes result in a turbid, algal dominated system (Scheffer 1990).

Comparing changes in total phosphorus (Figure 7), chlorophyll-*a* concentration (Figure 8), turbidity (Figure 9) and light attenuation (Figure 11) over time in lakes Crescent and Sorell highlights the dissimilarities between the two lakes in regards to controlling processes stimulating and limiting algal productivity. Both lakes have comparable increases in total phosphorus, turbidity and light attenuation contrasting the extreme increase in algal biomass in Lake Crescent compared to a significant yet much lower increases in algal biomass in Lake Sorell. Intuitively, if light and nutrients alone were responsible for limiting algal productivity in both systems then comparable increases in algal biomass would likely result. Therefore, the processes responsible for limiting and stimulating algal productivity in the two lakes are significantly different.



### **2.4.3 Nutrient budgets**

For 2000 and 2001, low nutrient export coefficients for subcatchments draining into lakes Sorell and Crescent resulted in a net loss of nitrogen, phosphorus and suspended solids from each lake. The exception was nitrogen in Lake Crescent, where inputs from Lake Sorell and Agnews Creek combined exceeded exports from the lake in water released down the Clyde River.

The high nitrogen loading of Agnews Creek entering Lake Crescent is compounded by the release of water from Lake Sorell that, for the period studied, had increased nitrogen levels. The strong correlation between SPM and total nitrogen values in Lake Sorell (Figure 13), and the aim of future management of the lakes to reduce SPM values and improve water clarity will reduce nitrogen concentration in water released from Lake Sorell lowering nitrogen loads entering Lake Crescent.

In the case of Agnews Creek, the highest recorded nitrogen levels coincide with low flows, possibly reflecting a dominance of nitrogen carried in groundwater as opposed to dissolved and particulate forms of nitrogen entering the creek by overland flow. This would be similar to the mechanism proposed by Grieve and Gilvear (1994) who found the principal source of nitrate to be groundwater. It is possible that land use practices in the top of the catchment increase nitrogen levels in the groundwater which then enters Agnews Creek. Unlike phosphorus (which will not move substantially with subsurface flow due to a strong affinity for binding to the clay fraction within soils or precipitation as iron phosphate), nitrogen may travel in either dissolved or particulate form, and can be transported by subsurface flow and groundwater (Cullen 1983).

It is possible that the increased nitrogen loading of Agnews Creek enters via subsurface flow and groundwater intrusion, and this is supported by the low nitrogen concentrations measured at increased discharge compared with high nitrogen concentrations measured during periods of low flow (Appendix 6). This likely reflects the source of nitrogen to be limited in the catchment with nitrogen transport being less dependent on overland flow as the reverse would be expected (Buckney 1979), as observed with the other tributaries measured in this study that show a significant

positive correlation of increasing nitrogen concentration with increased discharge (Appendix 6).

Alternatively, a significant point source input of nitrogen within the catchment would result in a similar phenomena as the nitrogen is limited and is therefore diluted with increasing discharge (Cullen and O'Loughlin 1982). With no significant development within the Agnews Creek catchment, the likelihood of a point-source input high in nitrogen is low. At the headwaters of Agnews Creek there exists an extensive marsh area which has been channelised and drained in the past to improve grazing. Complete desiccation of sediments that would occur over summer in the wetland area at the top of the Agnews Creek catchment may promote nitrogen release from the sediments when the soils become rewetted and discharge from the area commences (Baldwin and Mitchell 2000; Scholz, Gawne et al. 2002). The channelised wetland area may therefore significantly influence nitrogen export from Agnews Creek. Promoting water retention in the wetland area at the top of the catchment to limit the degree of desiccation of sediments and provide partial drying of wet sediments over summer months may provide a zone of coupled nitrification-denitrification and lead to a reduction of nitrogen through promoting denitrification and loss of nitrogen gas to the atmosphere (Baldwin and Mitchell 2000).

The source of nitrogen is difficult to determine and warrants further investigation. Fractionation of the nitrogen component entering Lake Crescent from Agnews Creek would help in determining the key hydrological pathways for the transfer of nitrogen into Lake Crescent (Heathwaite and Jones 1996) and provide insight into the source of high nitrogen in Agnews Creek.

It may be concluded that the water quality of inflows, with the exception of nitrogen in Agnews Creek, is of high quality, with most inflows being relatively 'pristine'. From the 2 year monitoring program detailing nutrient and suspended solids loadings entering lakes Crescent and Sorell from surrounding subcatchments, coupled with knowledge of current land-use practices in the area and comparisons to similar studies from mainland Australia and overseas, it is possible to conclude that at current values, local agricultural and forestry practices are not having a measurable detrimental impact on the water quality of tributaries nor on the receiving waters of lakes Crescent

and Sorell. Further work on Agnews Creek is warranted to identify the source of increased nitrogen, as Agnews Creek reflects a mild case of increased eutrophication and is significantly influencing nitrogen loading in Lake Crescent.

From routine monitoring work carried out on both lakes, it is possible to conclude that the highly significant increase in nitrogen, phosphorus and suspended sediment measured between 1998 and 2002 in both lakes Crescent and Sorell is due to internal loading via sediment resuspension, and this process reflects the most significant source of sediment and nutrients in both lakes.

With the highly degraded state of water quality experienced by lakes Crescent and Sorell during 2000 and 2001, with high values of nitrogen, phosphorus and suspended sediment, coupled with low external loadings of these fractions entering the lakes from the subcatchments, there was evident a net loss of nitrogen, phosphorus and sediment from both Lake Sorell and Lake Crescent, with the exception of nitrogen in Lake Crescent for 2001.

#### **2.4.4 Conclusion**

It may be concluded that observed increases in values of nutrients and suspended sediment in both lakes Crescent and Sorell over recent years is due primarily to internal processes and is not influenced by external inputs.

Quantifying the contribution each independent variable, such as seston, inorganic suspensoids, detritus and dissolved colour, makes to limiting light availability will provide a basis for understanding the processes responsible for the decline in water quality. Additionally, understanding sediment resuspension dynamics in both lakes is of considerable importance as it appears that sediment entrainment is a key factor driving SPM values and degrading the light climate, particularly with increased turbidity and SPM concentrations coinciding with successive dry years and low lake levels.

In conclusion, increasing suspended sediment is strongly correlated with a substantial increase in algal biomass in both lakes. Given the morphological and physical similarities between the lakes (Table 2), it is likely that the processes responsible for

this are similar, and internal loading of nutrients from sediment resuspension is a more likely mechanism driving increased algal biomass rather than changes to nutrient inputs from the surrounding catchment. However, the lakes behave quite differently as to the magnitude of change in primary productivity which questions the reasons for such a contrast.

The extreme contrast in algal productivity for two systems that adjoin and share such physical, chemical and morphological similarities is of considerable interest and warrants further investigation. Qualifying and quantifying both top-down and bottom-up processes that influence primary productivity will be necessary to understand the dynamic behaviour of the two lakes and to determine possible management strategies to improve water clarity and promote the desired macrophyte-dominated clear water state characteristic of Lake Sorell in past years.

## **Chapter 3 The role of colloids and resuspension**

### **3.1 Introduction**

Chapter 2 presented evidence that during the 1990s the water quality of both lakes Crescent and Sorell had deteriorated with increases in turbidity and nutrients which coincided with low water levels during this period. Estimates of nutrient loadings suggest that most of the increase in nutrients is from “internal loading” from resuspended sediments. Sustained and even increased demands for irrigation water supply combined with a management imperative to restrict access of introduced carp to the littoral wetlands of these lakes could mean that lake levels may stay low for some years. Apart from the obvious detraction from the visual and aesthetic appeal of these lakes during these drier conditions, such a regime could have serious consequences for their long-term conservation status, especially if repeated, prolonged episodes of resuspension of nutrients “flips” these lakes into an alternative algal “cloudy water” state. Conversely, decreasing nutrient and algal levels, along with improving the light climate, is likely to be closely linked to managing turbidity. Given the low overall concentrations of inputs (Chapter 2), and high internal loading, manipulation of water levels appears to be the only feasible management option for these lakes.

From Chapter 2, it was clear that colloidal material was likely to be important in the light climate of these two lakes. Colloidal turbidity is caused by small clay particles ( $< 1\ \mu\text{m}$ ) being suspended in the water column, and these particles often have electrostatic, negative surface charges (Langmuir 1997). When two particles approach each other, the surface charge of one particle is repelled by the other (Drever 1997), and this repulsion between particles counteracts potential aggregation by the van der Waals force (Drever 1997). Thus, charged colloids, combined with their extremely slow settling velocities (Stumm and Morgan 1996), result in particles that remain almost indefinitely in suspension. For example, settling rates for uncharged spheres of diameter  $1\ \mu\text{m}$  to  $0.1\ \mu\text{m}$  and a density of  $2\ \text{g cm}^{-3}$  in water according to Stokes Law (Equation 1.3) would be between  $20\ \text{cm day}^{-1}$  and  $2\ \text{mm day}^{-1}$  (Shaw 1983). The presence of colloids in freshwater systems therefore contributes significantly to ‘background turbidity’.

Background turbidity is caused by material that is present in the water column under quiescent conditions, and is independent of material that has been resuspended from the lake bed. This material largely consists of buoyant algal cells and colloidal clay particles (Hamilton 1990). The issue of increased colloidal loading in both lakes Sorell and Crescent is extremely important as this material is contributing significantly to degraded water clarity.

The relationships between water depth, fetch and shear stress (Equations 1.9 -1.18) coupled with the linear relationship between shear stress and sediment entrainment (Hamilton and Mitchell 1996; Hamilton and Mitchell 1997) show that low water levels reached in both lakes in recent years (Appendix 10) would increase the degree of sediment entrainment in both high and low energy areas, where low energy areas represent zones of sediment accumulation where fine sediments with grain sizes less than 6µm are focussed and settle out continuously (Hakanson and Jansson 1983).

Additionally, the low water levels reached after 1998 would increase sediment entrainment beyond typical levels in areas of frequent sediment disturbance leading to exposure of new, previously undisturbed material. The newly exposed, well consolidated material would likely contain more clay colloids than the upper layer of sediment top layer, as the fine particulate fraction would have had little opportunity to be entrained and transported out of the system. The sediment top layer (Blom, Van Duin et al. 1992) represents the layer of sediment frequently resuspended and sedimented in moderate to high energy transportation/erosion areas (where transportation areas are zones of discontinuous deposition of fine materials where both sediment deposition and entrainment periodically occur, and erosion areas are zones of relatively continuous sediment disturbance with no deposition of fine materials (Hakanson 1982)). Therefore, the large increase in suspended colloids observed in lakes Sorell and Crescent likely reflects increased disturbance of well consolidated sediments in the lower sediment layer below the less consolidated sediment top layer (Blom, Van Duin et al. 1992), along with increased disturbance of sediments in sheltered low-energy accumulation areas.

These considerations prompted this phase of the project, and there were two aims. First, the intensive record of measurements from 2000 - 2002 were plotted as time-series, and trends and correlations assessed. The data were further analysed using descriptive multiple regression models in an attempt to determine which components of the suspended materials in the water of each lake were most important in explaining measures of turbidity and water clarity. This method has been commonly employed in studies of shallow lakes, and has been used to set priorities for management (cf. Scheffer 1998). The second aim of this chapter was to investigate the relationships between turbidity and other measures of water clarity, and lake level in order to better understand how components of the suspended material behave and develop some simple descriptive models to assess the risks of resuspension events as lake levels are lowered.

## **3.2 Methods**

### **3.2.1 Temporal trends**

Details of the methods and data sources are given in to Chapter 2, Section 2.2. Initially time-series plots were inspected to assess any broad-scale trends and changes in major water quality variables for the period from April 2000 to August 2002. This allowed comparisons to be made between key components that influence turbidity over time and also allowed comparisons to be made between lakes.

To determine which components were responsible for the reduction in water clarity, multiple linear regression was used to model the contribution of algae, detritus, resuspended inorganic material and dissolved colour and colloids to turbidity, Secchi disk transparency and light attenuation (Scheffer 1998). These five water quality components are responsible for almost all light scattering and absorption in freshwater systems (Kirk 1985). The size of the regression coefficients do not necessarily reflect the importance of each variable by themselves (e.g. Quinn and Keough 2002). Instead, previous researchers have sought to quantify the relative importance of these variables by multiplying the average concentration of each variable by the corresponding coefficient and the results are plotted to graphically summarise the total and relative contribution of each component (Gerbeaux and Ward 1991; Blom, van Duin et al. 1994). For this study, I computed the average concentration of each

variable in each of lakes Crescent and Sorell separately for each sample date prior to multiplying through by the relevant coefficients from each regression model. The resulting plots sought to depict the changes in the relative importance of algae, detritus, resuspended inorganic material and dissolved colour and colloids to turbidity, Secchi depth and light attenuation (i.e. separate regression models were computed for each of the three response variables).

Thus, for Lake Crescent, data were collected from 8 sites on 39 sample trips starting on 19 April 2000 until 27 February 2002, and a subset of 4 of these sites were sampled on four further occasions on 13 June, 10 and 25 July and 22 August 2002. This yielded 328 records, with a small number of missing values for most of the variables as tabulated in Table 12, except for light attenuation because of a lack of instrument availability. For Lake Sorell, 8 sites were visited over the same period and a subset of 4 sites were visited on a further 7 dates yielding 341 records, again with some missing values (Table 12). Since the pattern of missing values was random, and the total number of records with a missing value was < 5% (except for light attenuation) for each lake, incomplete cases for a given regression model were simply omitted (Harrell 2001 p.49).

Values of each of the response variables (i.e. turbidity, Secchi depth and light attenuation), were regressed on the four predictor variables (i.e. inorganic solids, detritus, algal biomass and colour & colloids). Regression assumptions were checked by plots of residuals versus estimates, normal probability plots, and influence and leverage diagnostics (Quinn & Keough 2002). For some models one or two potentially influential points were identified, and regressions were repeated omitting these values. However, these points made little difference to the regression coefficients and hence little difference to the resulting plots.



Table 12 Number of data records for which there was a missing value for each variable in lakes Crescent and Sorell.

Variable	Crescent	Sorell
turbidity	1	0
Secchi depth	10	1
light attenuation	137	141
inorganic solids	5	9
detritus	8	21
algal biomass	4	20
colour & colloids	5	7

### 3.2.2 *Modelling turbidity responses to water levels*

#### 3.2.2.1 Exposure of bed to wave action

To determine if the low lake levels experienced by both lakes after 1998 increased the area of lake susceptible to wind driven wave disturbance, equations 1.15 to 1.18 were used to model wavelengths across the lake surface from information on wind fetch, water depth and wind speed for a maximum of 842 points in Lake Sorell at full supply level (FSL) and 325 points in Lake Crescent at FSL.

The energy parameter with the greatest influence on bottom dynamics is wavelength which controls the depth that the orbital motion from wind driven waves influences the sediments (Hakanson 1977). Using the relationship described by Carper and Bachmann (1984) that assumes the point at which the wave base contacts the sediment surface to be at a water depth of  $\frac{1}{2}$  the wavelength, a graph was derived depicting the changes in lake bottom area influenced by wind driven waves for lake levels typical of those before and after 1998 (Appendix 10).

Hakanson (1982) derived a formula coined the ‘energy-topography factor’ which approximates the erosion/transportation area of a lake based on basic morphometric information on lake area and average depth. The energy-topography factor may be written as:

$$a_{E+T} = 25 \times (\sqrt{a} / \bar{D}) \times 41^{0.0061 \times (\bar{D} / \sqrt{a})} \quad 3.1$$

where  $a_{E+T}$  = the percent area of the lake bed subject to erosion and transportation,  $\bar{D}$  is the average lake depth (m) and  $a$ , the surface area of the lake in km<sup>2</sup>. Equation 3.1 illustrates the importance of lake depth in determining erosion/transportation areas and in turn influencing areas of sediment accumulation. A significant decrease in average lake depth would therefore increase the area of lake bottom influenced by wind driven waves and reduce the size and existence of low energy areas of sediment accumulation, areas that represent significant sinks (and therefore sources) of fine particulate material (Hakanson and Jansson 1983).

### 3.2.2.2 Lake volumes, water residence times and flushing

Analyses were carried out that compared average historical and observed lake volumes for the two lakes coupled with water inflow, evaporation and outflow volumes to determine theoretical water residence times for both lakes. This was undertaken since water exchange may reduce conservative substances such as colloids by influencing the retention (and hence the outflow) of suspended material in lakes (Hakanson 1995). Moreover, flushing has been identified as a potential management tool providing water is available in sufficient quality and quantity (Hosper 1998).

The large volume difference between the lakes would likely influence the rate of water exchange, in turn influencing the export of fine particulate material. Although a simplistic analogy, taking a lake volume typical of the lakes for the period from January 1970 to December 1997, Lake Sorell, at an average level of 804.09 mASL has a volume of 142 000 ML, compared to Lake Crescent with average lake volume of 34 000 ML (at an average level of 803.30 mASL) for the same period. If an equivalent water volume entering Lake Sorell was to be released through Lake Crescent and down the Clyde River, the exchange rate of Lake Crescent would be

approximately 4 times that of Lake Sorell. Obviously this fails to take into account evaporative loss from the lakes where Lake Sorell would lose almost twice the volume of Lake Crescent due to its increased surface area.

Inflow volumes entering the lakes estimated for July 2000 to June 2001 and July 2001 to June 2002 were used to determine theoretical water residence times according to Hakanson and Jansson (1983) which, under the assumption of steady state (i.e. the inflow volume  $Q_{in}$  equals the outflow volume  $Q_{out}$ ), the water residence time ( $T_w$ ) is given by:

$$T_w = V/Q \quad 3.2$$

Average lake volumes ( $V$ ), for the period running January 1970 to December 1997 were used to estimate of 'typical' lake volume, as opposed to using full supply volume as the lakes are typically held at lower volumes than the full supply volume.

Additional to inflow volume, estimates of rainfall and evaporative loss were made to determine 'net' inflow volume entering the lakes, as rainfall and evaporative loss play a significant role in the total amount of water entering and exiting the system. Water residence time for both lakes were approximated by dividing 'average' lake volume by net inflow volume for the period in question.

The annual average evaporative loss was estimated from the Bureau of Meteorology's Bushy Park weather monitoring station after comparison with daily evaporative loss rates from an evaporation pan located on the shores of Lake Sorell, and maintained by staff of the Inland Fisheries Service over 2002 and 2003

To further assess the role of the water budget in mediating colloidal turbidity, expected decreases and increases in colloidal turbidity in Lake Sorell were calculated assuming dilution and concentration to be the dominant processes controlling colloids at high lake levels. Assuming colloids to behave as a conservative substance, a basic dilution/concentration formulation was applied to approximate changes in colloidal turbidity as determined from changes in lake volume. The equation was of the form  $C(i) \times V(i) = C(f) \times V(f)$  where  $C(i)$  is the initial colloidal turbidity (NTU),  $V(i)$  the

initial volume of Lake Sorell (ML), and  $C(f)$  being the final 'expected' colloidal turbidity with a change in lake volume of  $V(f)$ . Changes in lake volume for three separate periods were used to determine expected colloidal turbidity levels.

### **3.3 Results**

#### ***3.3.1 Temporal trends in light climate and its determinants***

Over the period April 2000 to August 2002, the peak in average lake-wide turbidity for both lakes was recorded on the 17<sup>th</sup> July 2000 with an average of 280 NTU for Lake Crescent (Figure 15) and 330 NTU for Lake Sorell.

Concentrations of inorganic material and detrital material correlated closely with turbidity for both lakes Crescent and Sorell suggesting a dependence of turbidity and water clarity on tripton. For both lakes there was a sharp increase in all components, with the exception of colour and colloids, from early to mid 2000, at which point levels peaked and plateaued in Lake Sorell (Figure 16). In Lake Crescent, there was substantial variation between sampling dates from April 2000 to June 2000 with a sharp increase for the on 17<sup>th</sup> July 2000 at which point inorganics and detritus peaked with the corresponding peak in turbidity in both lakes (Figure 15).

Turbidity rose sharply in Lake Sorell from April 2000 with average levels remaining above 200 NTU from late May 2000 to mid October 2000, at which point lake-wide average levels of turbidity fell below 200 NTU. Average levels of turbidity in Lake Sorell remained below 200 NTU until October 2001, at which point lake-wide average levels of turbidity fell once again, and remained below 100 NTU for the period running October 2001 to May 2002. The sampling trips of June, July and August of 2002 for Lake Sorell recorded a marked rise in lake-wide average turbidity levels to peak at 143 NTU on the 22<sup>nd</sup> of August 2002.

Turbidity in Lake Crescent showed a marked contrast in behaviour to Lake Sorell. Despite the maximum being recorded on the same sampling trip as Lake Sorell, turbidity levels in Lake Crescent varied substantially between sampling trips and remained high through 2000 and 2001, with average levels exceeding 200 NTU on

numerous sampling occasions until the 10<sup>th</sup> of August 2001, at which point lake-wide average turbidity levels began to trend downwards to fall below 100 NTU on the 22<sup>nd</sup> of November, 2001, where they remained through to June 2002.

Algal biomass and dissolved colour and colloids behaved differently between the two lakes. Algal biomass in Lake Crescent remained, on average, an order of magnitude higher than Lake Sorell for the period running April 2000 to August 2002. For both lakes Crescent and Sorell, trends in algal biomass appear to follow trends in turbidity, with a positive correlation between the two. For both lakes, peaks in algal biomass closely coincided with increases and peaks in turbidity. In the case of Lake Crescent, maxima in algal biomass follow maxima in turbidity, with the magnitude of variation between samples increasing with increasing lake-wide average values of algal biomass (Figure 15). This suggests that the algal standing crop in Lake Crescent is closely aligned with the process leading to increases in turbidity.

Overall, the amount of dissolved colour and colloids in both lakes has increased dramatically. Estimates from the early to mid 1990's of dissolved colour averaged  $2.8 \text{ m}^{-1}$  (standard deviation,  $s = 0.62$ ;  $n = 13$ ) for Lake Sorell and  $2.8 \text{ m}^{-1}$  ( $s = 0.95$ ;  $n = 13$ ) for Lake Crescent (Inland Fisheries Service, unpublished data). This has increased to an average of  $16 \text{ m}^{-1}$  ( $s = 2.8$ ;  $n = 337$ ) for Lake Sorell and  $7 \text{ m}^{-1}$  ( $s = 1.9$ ;  $n = 339$ ) for Lake Crescent over April 2000 to August 2002. Since dissolved colour measures the amount of humic acids, which often result from the decomposition of plant material (Kirk 1994), these increases are consistent with the loss and erosion of littoral aquatic macrophyte beds since the late 1990s.

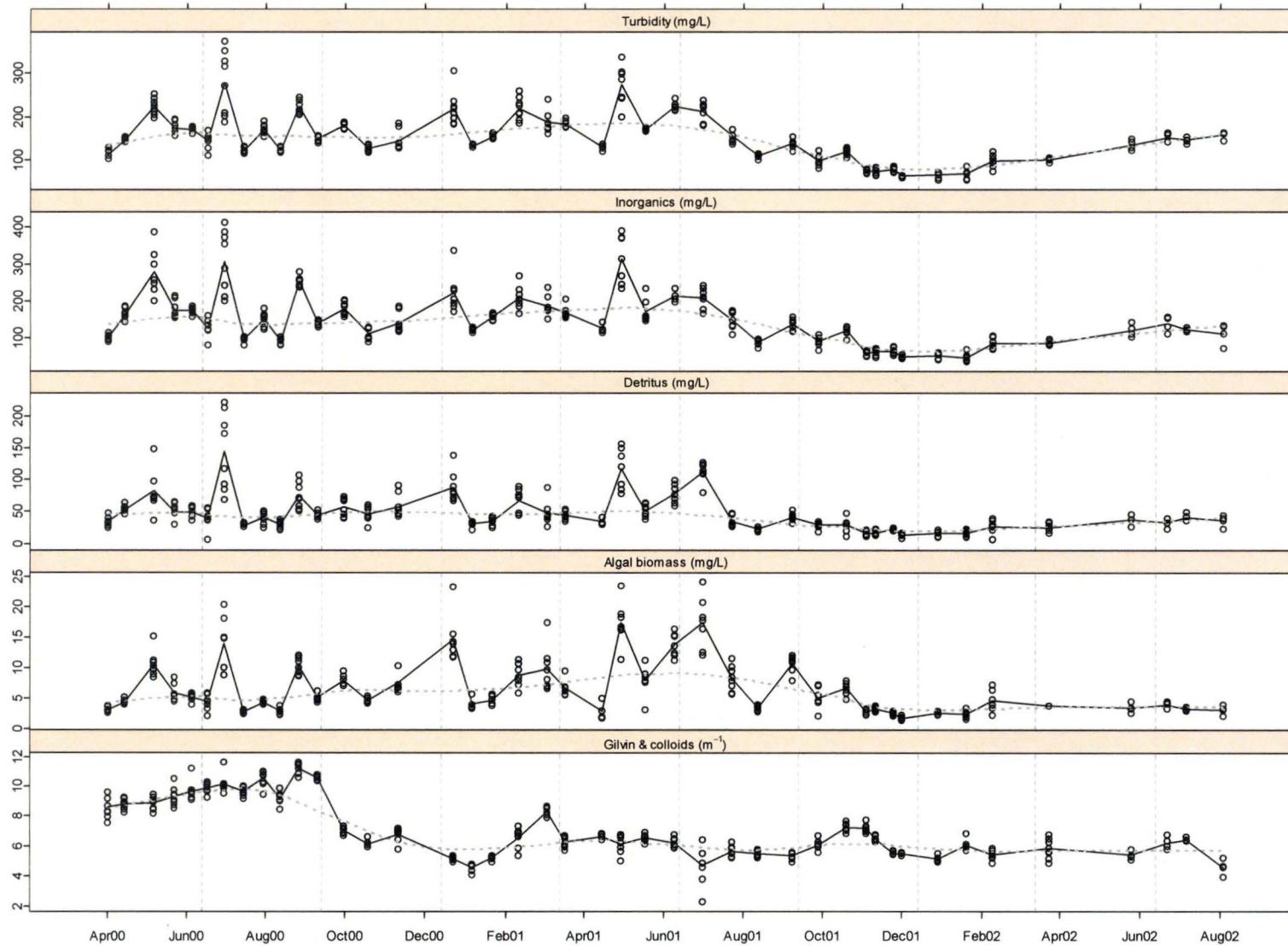


Figure 15 Time-series plots of turbidity, and concentrations of suspended inorganic matter, detritus, algal biomass and colour and colloids for the period April 2000 to August 2002 in Lake Crescent. Symbols are measures from each sampling station, and the thin line connects the mean values on each sample date. The dotted line is a loess-smoother to accentuate longer term trends.

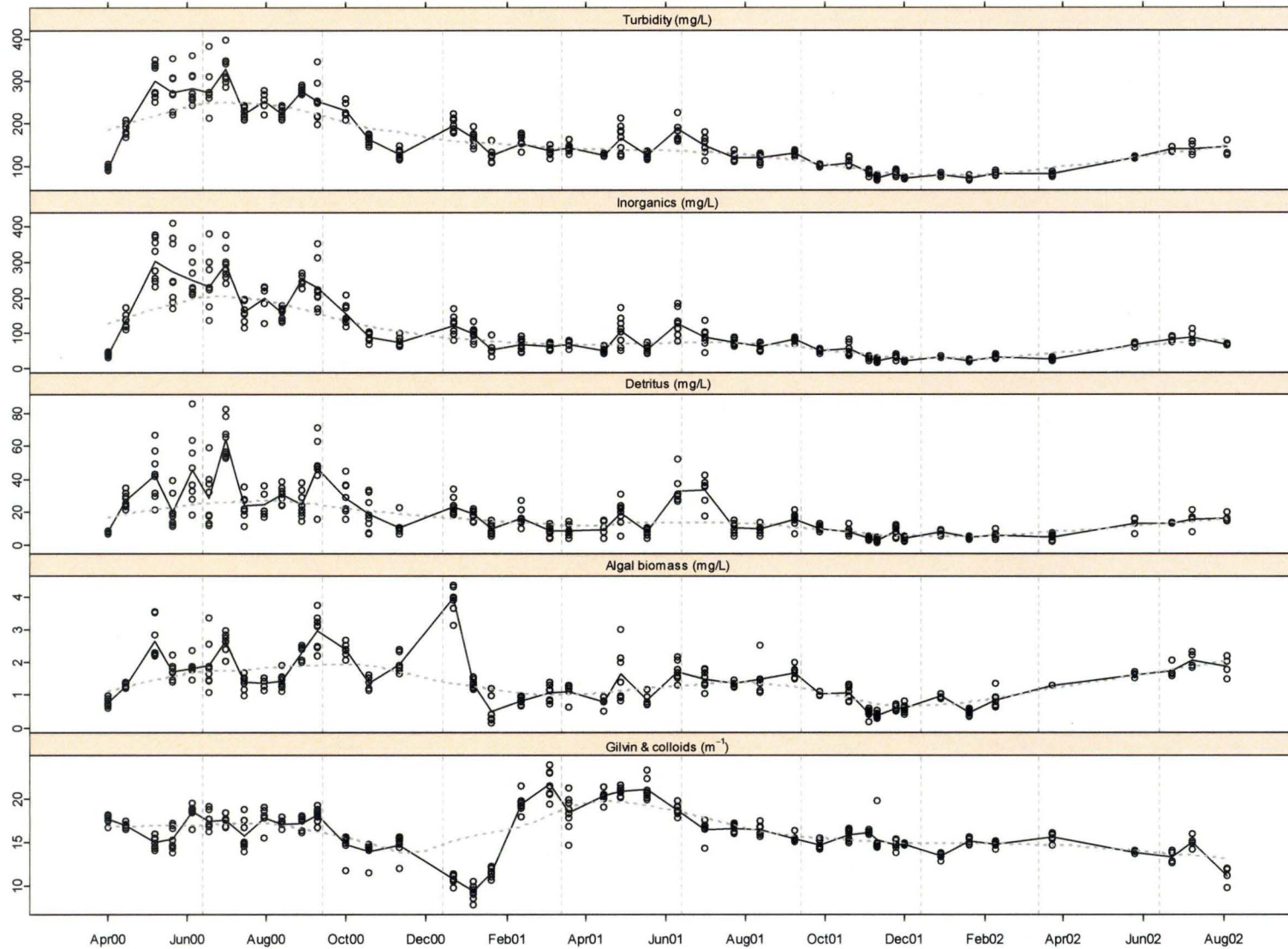


Figure 16 Time-series plots of turbidity, and concentrations of suspended inorganic matter, detritus, algal biomass and colour and colloids for the period April 2000 to August 2002 in Lake Sorell. Symbols are measures from each sampling station, and the thin line connects the mean values on each sample date. The dotted line is a loess-smoother to accentuate longer term trends.

The regression models fitted very well (all  $P < 0.001$ , all  $R^2 > 0.8$ , except for 1/Secchi depth in Lake Crescent; see Table 13 and Table 14), and indicate that inorganic suspensoids and dissolved colour and colloids collectively account for much of the turbidity, inverse Secchi depth and light attenuation (Figure 17 to Figure 22).

Dissolved colour and colloids were more important in Lake Sorell than in Lake Crescent, while in Sorell the relative contribution of colour and colloids increased at the expense of the contribution by inorganic solids in late 2001 through to early 2002, during which period the water levels in the lakes rose substantially. Overall, the amount of dissolved colour and colloids in both lakes has increased dramatically.

Prior to 1998 colloidal turbidity was stable and rarely rose above 10-15 NTU (Chapter 2, Figure 10; Figure 26 and Figure 27), while over the period 2000-2002 it has increased to  $> 60$  NTU in Lake Crescent and  $> 90$  NTU in Lake Sorell (Chapter 2, Figure 10; Figure 26 and Figure 27; discussed in detail in the following section).

Table 13 Multiple regression coefficients for Lake Sorell. The coefficients for the regressions of each dependent variable (with standard errors in parentheses) are given along with the sample size,  $n$ , and coefficient of multiple determination,  $R^2$ .

Dependent variable	Intercept	Inorganics	Detritus	Algae	Colour and colloids	$R^2$	$N$
Light attenuation	1.650 (1.286)	0.412 (0.024)	0.023 (0.020)	0.055 (0.027)	0.258 (0.099)	<b>0.939</b>	168
1/Secchi depth	4.306 (0.512)	0.032 (0.002)	0.024 (0.009)	0.092 (0.141)	0.046 (0.029)	<b>0.841</b>	314
Turbidity	45.078 (6.477)	0.689 (0.020)	0.686 (0.108)	6.019 (1.761)	1.242 (0.367)	<b>0.947</b>	318

Table 14 Multiple regression coefficients for Lake Crescent.

Dependent variable	Intercept	Inorganics	Detritus	Algae	Colour and colloids	$R^2$	$N$
Light attenuation	0.185 (1.254)	0.101 (0.014)	0.096 (0.029)	0.145 (0.196)	0.768 (0.212)	<b>0.862</b>	171
1/Secchi depth	3.762 (0.337)	$1.674 \times 10^{-2}$ ( $2.749 \times 10^{-3}$ )	$2.429 \times 10^{-2}$ ( $5.759 \times 10^{-3}$ )	0.142 (0.041)	0.148 (0.049)	<b>0.765</b>	304
Turbidity	35.25 (3.04)	0.656 (0.252)	0.214 (0.052)	0.755 (0.368)	0.905 (0.445)	<b>0.950</b>	312



Figure 17 The contribution (total and relative) of inorganics, detritus, algae and colour (including colloids) to total turbidity levels in Lake Sorell from April 2000 to July 2002.

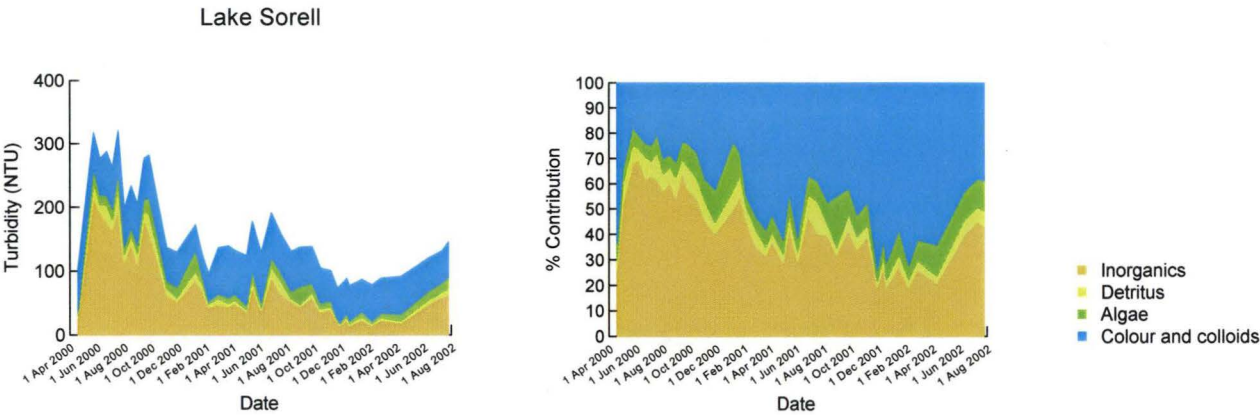


Figure 18 The contribution (total and relative) of inorganics, detritus, algae and colour (including colloids) to total turbidity levels in Lake Crescent from April 2000 to July 2002.

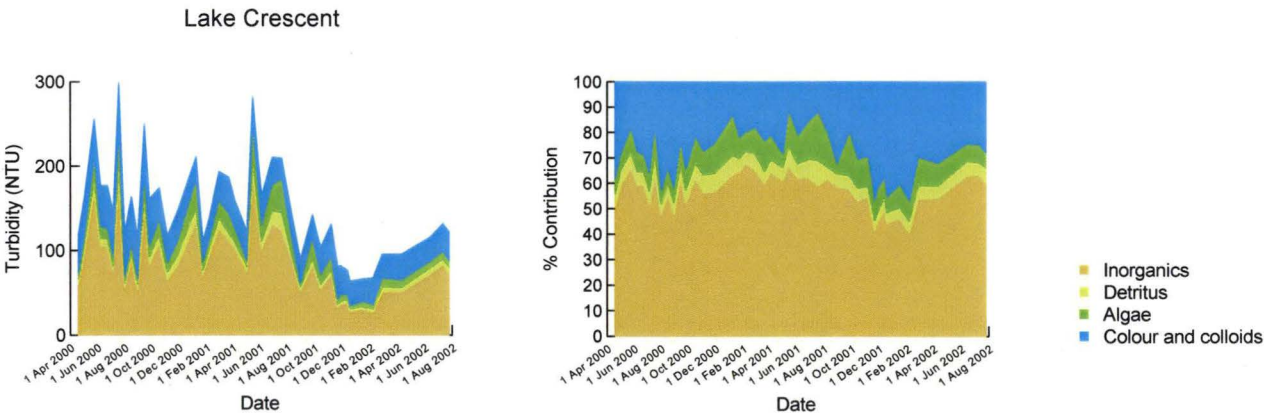


Figure 19 The contribution (total and relative) of inorganics, detritus, algae and colour (including colloids) to inverse Secchi levels in Lake Sorell from April 2000 to July 2002.

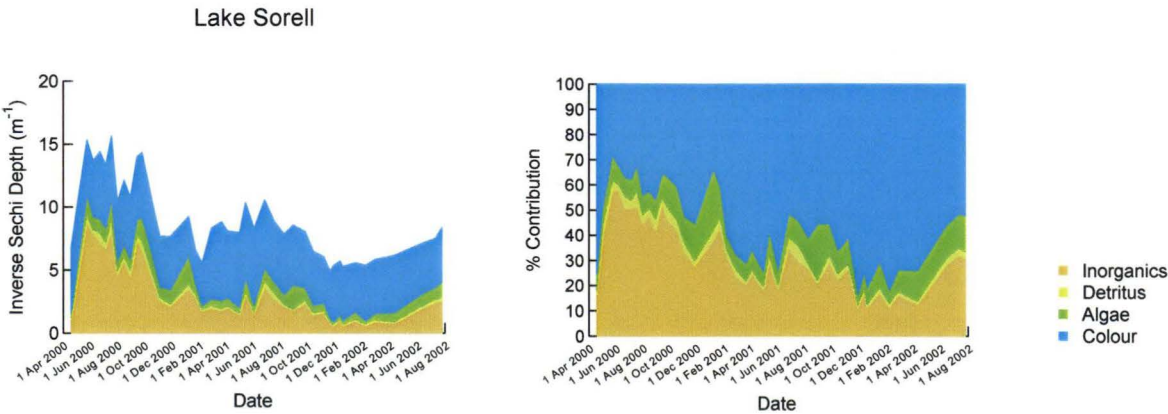


Figure 20 The contribution (total and relative) of inorganics, detritus, algae and colour (including colloids) to inverse Secchi levels in Lake Crescent from April 2000 to July 2002.

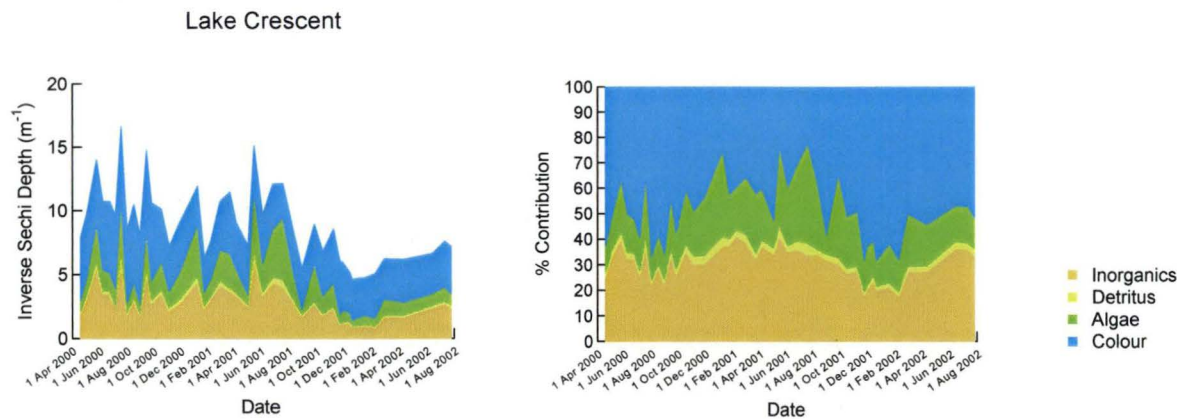


Figure 21 The contribution (total and relative) of inorganics, detritus, algae and colour (including colloids) to light attenuation in Lake Sorell from April 2000 to July 2002.

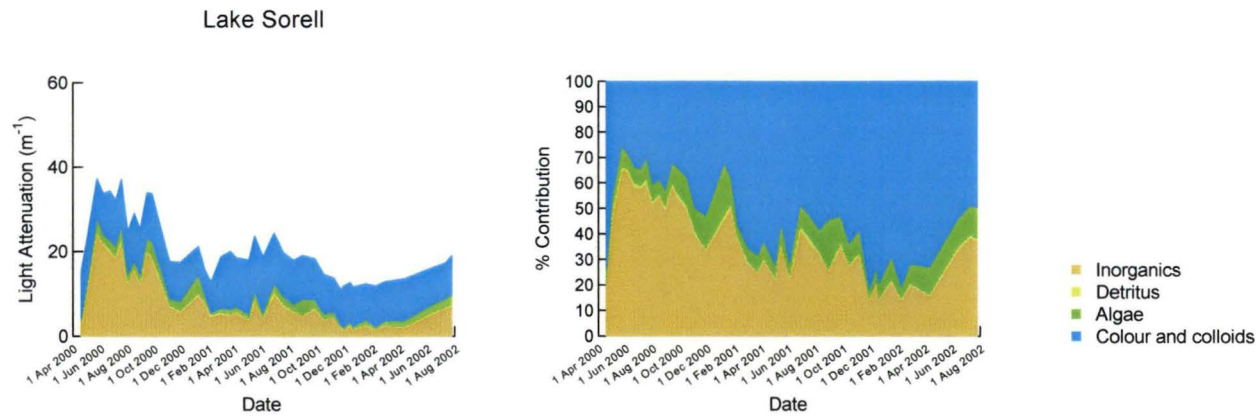
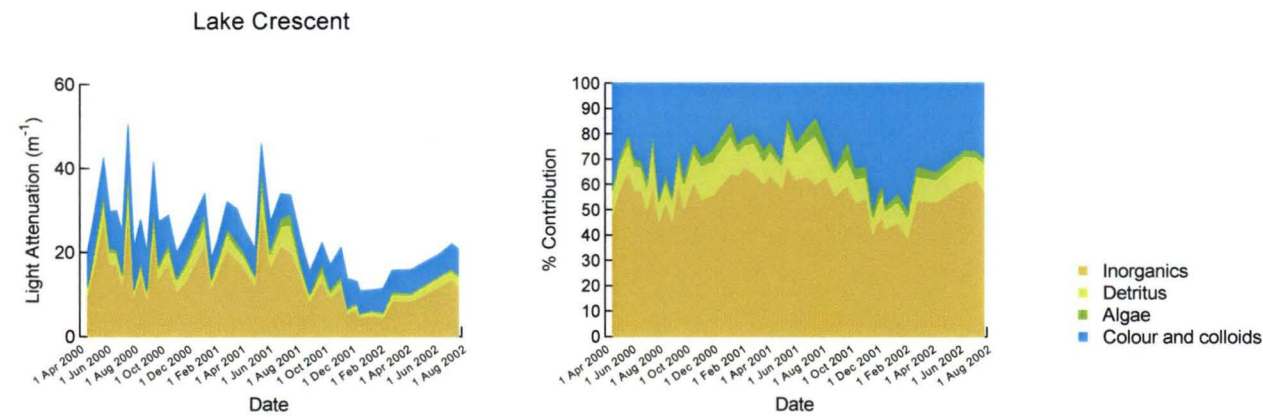
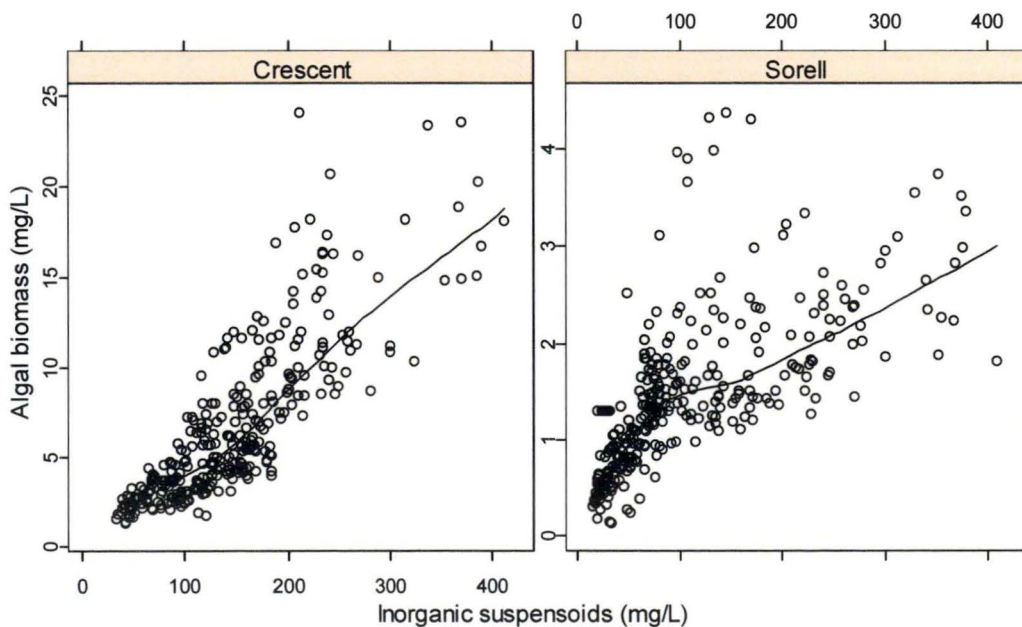


Figure 22 The contribution (total and relative) of inorganics, detritus, algae and colour (including colloids) to light attenuation in Lake Crescent from April 2000 to July 2002.



Algal biomass appeared to contribute little to the overall light climate, although it is strongly correlated with inorganic suspensoids in both lakes (see figure below), which may mask its influence in Figure 17 to Figure 22 above. Nevertheless, there is some suggestion that algal biomass may be more important at some times, and this is borne out by the percentage contribution by algal biomass to total suspended solids documented in Table 6 in the previous chapter. Apart from the very large difference in overall algal biomass between the two lakes (Figure 23), the relationship between algal biomass and inorganics differs. In Lake Crescent algal biomass continues to increase roughly linearly as the concentration of inorganics increases, while Lake Sorell shows a decrease in the slope of the relationship once inorganics exceed 100 mg L<sup>-1</sup>. This further suggests that algal standing crop in Lake Crescent may depend more on sediment resuspension than in Lake Sorell, and the source of algae in Lake Crescent may be closely aligned with the sediments as has been documented in lakes with a significant meroplanktonic community (Carrick, Aldridge et al. 1993; Schelske, Carrick et al. 1995; Schelske 2002).

Figure 23 Relationship between algal biomass and the concentration of inorganic suspensoids recorded at each sampling station in lakes Crescent and Sorell from April 2000 to August 2002. The line is a loess smoother; note the different scales for algal biomass for each lake.





### **3.3.2 Turbidity and lake levels**

The association of decreasing water level with increasing total turbidity is obvious (Figure 24, Figure 25). The autumn of 1998 showed a substantial increase in turbidity in Lake Sorell, coinciding with the minimum water level reached for the year. From this point, turbidity in both lakes has increased in marked 'steps', coinciding with minimum lake levels reached annually from 1998 thus suggesting a strong interaction between low water levels and increased sediment entrainment. The highest recorded turbidity in both lakes coincides with the lowest recorded water levels (Figure 24, Figure 25; Appendix 10).

Further evidence of the prevalence of wind resuspension is the increased hysteresis in turbidity as lake levels decrease. At high lake levels, fluctuations in turbidity are less marked and variation in lake-wide average turbidity values decreases significantly reflecting a relatively homogenous, clear water (Figure 15 and Figure 16). At this stage, turbidity values are likely influenced to a greater extent by primary productivity and background turbidity. When water levels are low, the influence of wind events are more likely to be important resulting in variable and more frequent high values of turbidity, as occurred during the very low lake levels reached in 2000 and 2001. In Lake Sorell, the high water level reached at the end of 2001 did not reduce turbidity to that recorded in 1997 at a similar lake level, which strongly implicates colloids as a likely cause of this persistently high turbidity at the end of 2001.

Figure 24 Changes in turbidity and lake level in Lake Sorell from mid 1996 to December 2002.

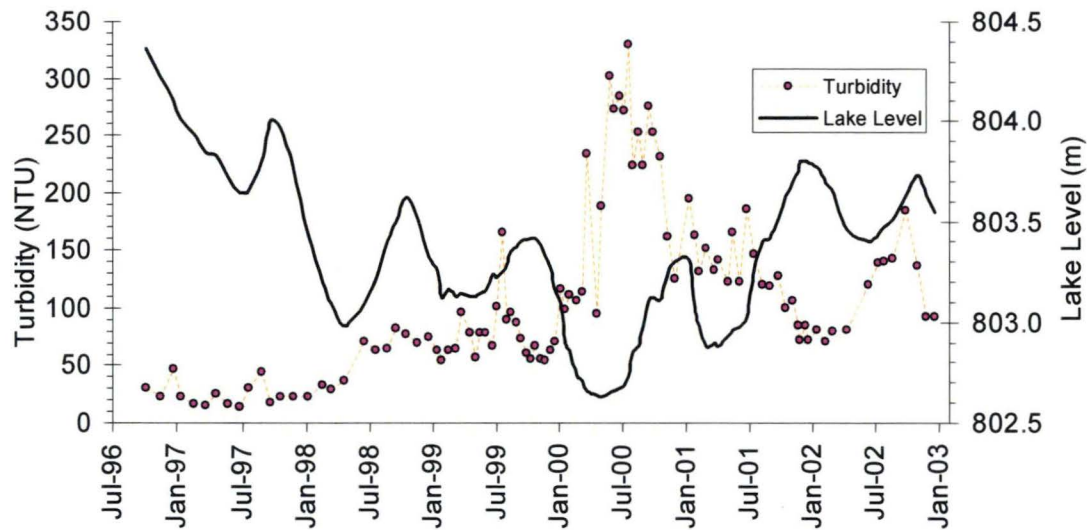
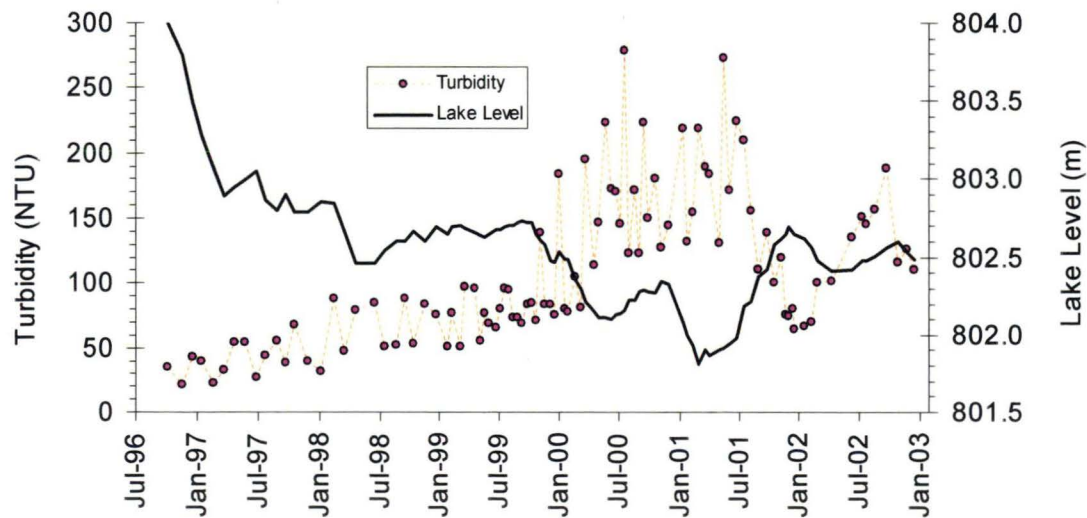


Figure 25 Changes in turbidity and lake level in Lake Crescent from mid 1996 to December 2002.



Colloid levels in both lakes Sorell and Crescent have increased dramatically. In Lake Sorell, colloidal turbidity levels during 1996 were around 10 NTU; in 2001, levels peaked at 93 NTU (Figure 26). Due to the increased significance of colloidal turbidity in Lake Sorell, total turbidity levels have remained high, even with the significant rise in water level seen at the end of 2001. At this point, colloids contributed up to 90% of total turbidity recorded in Lake Sorell. During this time, if colloidal turbidity was at the historic level of 10 NTU, turbidity in Lake Sorell would have been comparable to historic levels of around 20 to 30 NTU.

The role of colloidal turbidity is implicated in Figure 26 and Figure 27 with step-like increases coinciding with lake levels falling below 803.2m (AHD) in Lake Sorell and 802.7 m in Lake Crescent. This suggests that when the lakes drop below these levels, there is increased erosion of consolidated lake bed sediments.

This process seems most marked in Lake Sorell, where, over the long term, there is a marked hysteresis in the relationship between colloidal turbidity and lake level (Figure 28). Colloidal turbidity was  $4 \times$  greater at water levels around 803.5 in early 2002 than in 1996. This pattern is not replicated in Lake Crescent and it seems that a rise in water level usually results in a substantial drop in colloidal turbidity (Figure 29).

It appears that colloidal turbidity has increased in the lakes due to unprecedented levels of bed erosion driven by low lake levels reached during 2000 and 2001, with the highest recorded turbidity coinciding with the lowest water levels on record.

Figure 26 Colloidal turbidity and lake levels for Lake Sorell from July 1996 to December 2002.

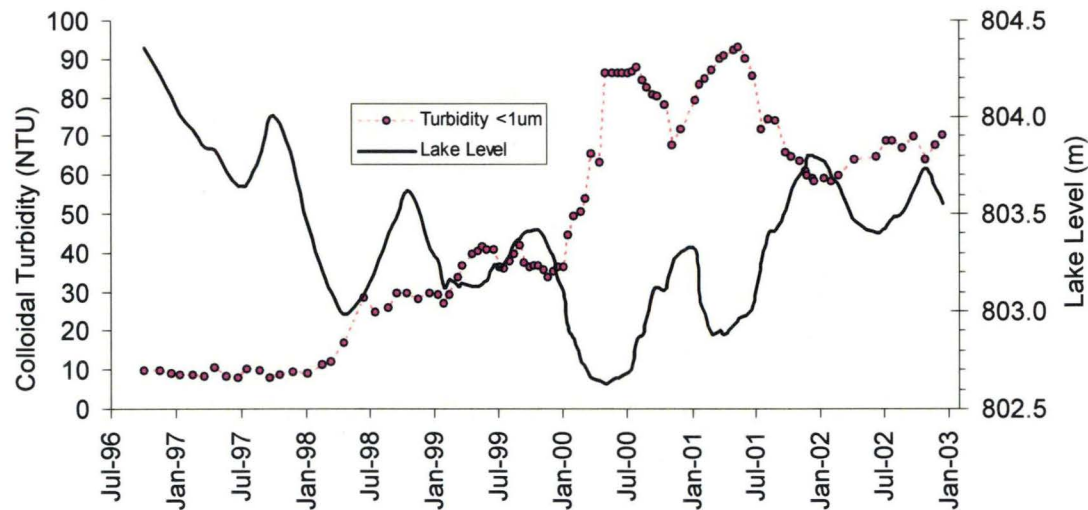


Figure 27 Colloidal turbidity and lake levels for Lake Crescent from July 1996 to December 2002.

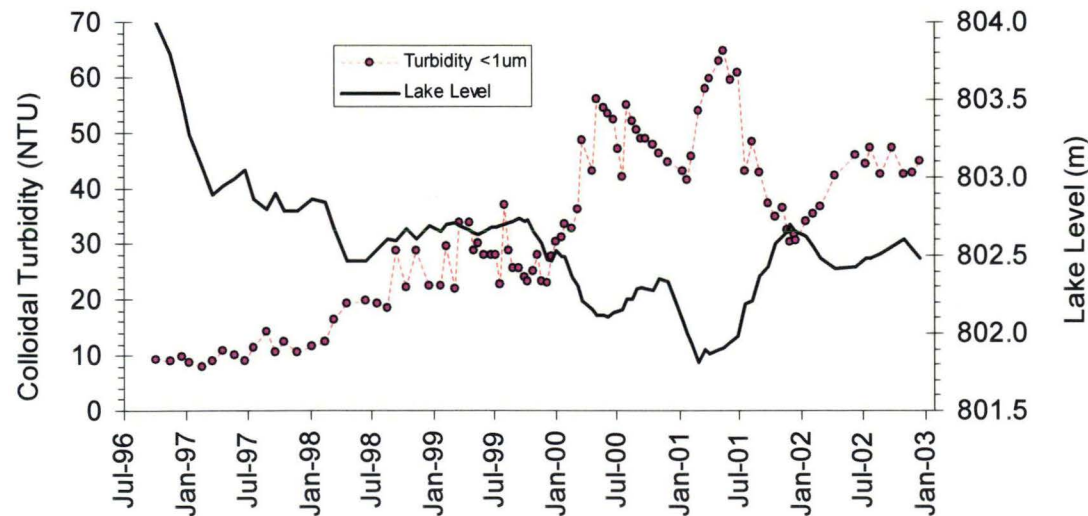


Figure 28 Relationship between lake level and colloidal turbidity in Lake Sorell. The line trace starts in 1996 at the far bottom right of the graph and finishes in August 2002 with the end of the line trace in the top middle of the graph. Open symbols denote data collected after the peak in colloidal turbidity in 2001, and have been excluded from the fitted regression trend.

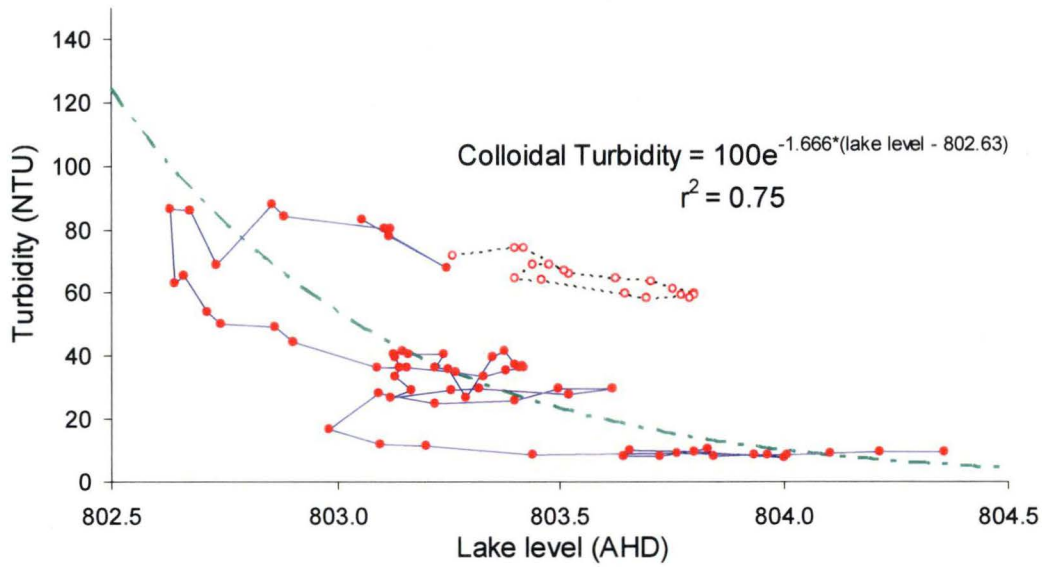
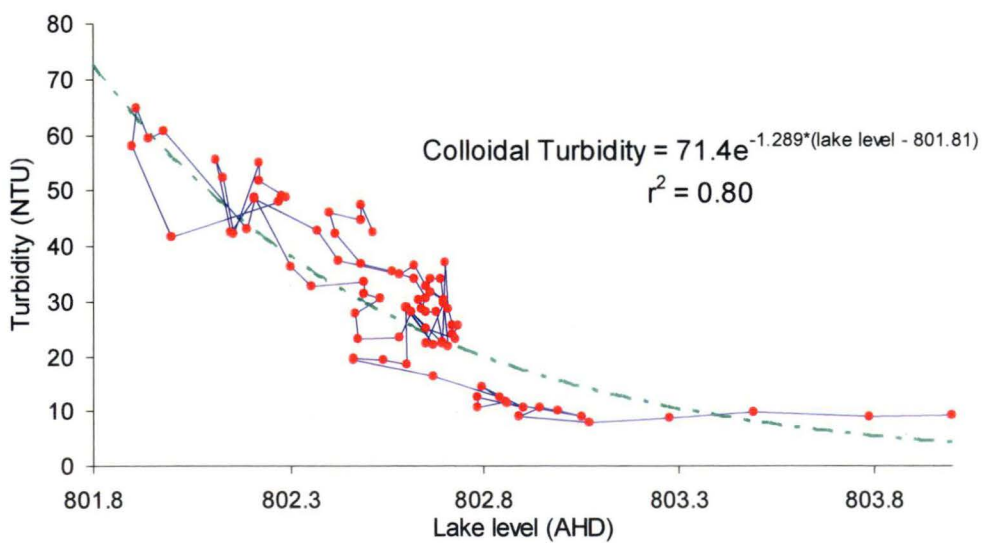


Figure 29 Relationship between lake level and colloidal turbidity in Lake Crescent.





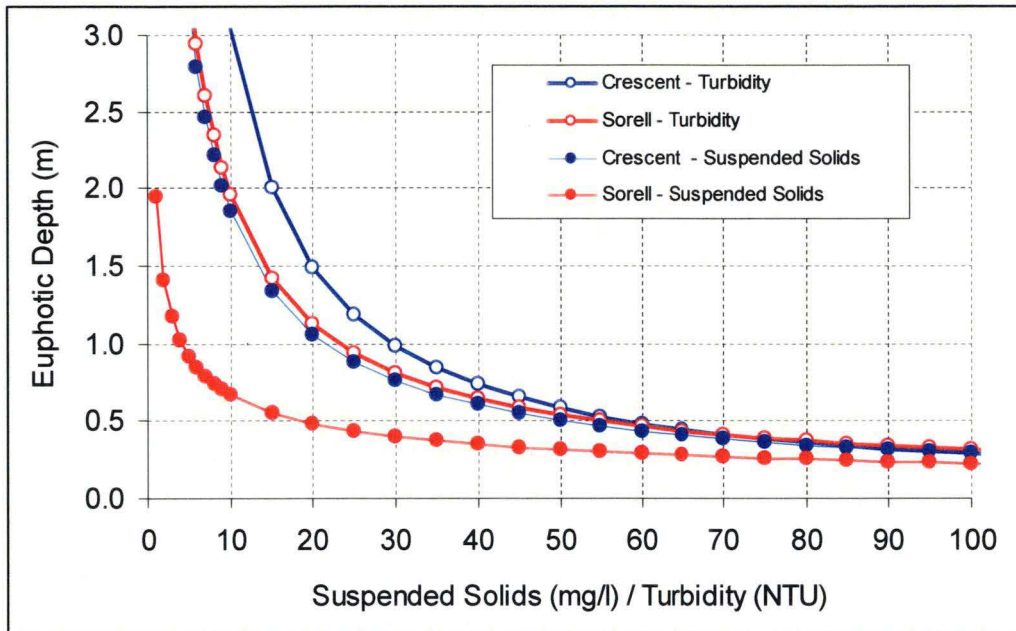
The important contribution colloids make to limiting light availability in Lake Sorell is obvious with reference to Figure 30. Figure 30 is an extrapolation of the relationships summarised in Chapter 2, Figure 11. The data used for the euphotic depth modelling presented in Chapter 2, Figure 11 utilises information collected during the course of this research project only. The entire data set is therefore influenced by the presence of colloids.

In regards to Figure 30, suspended solids as represented on the graph does not include colloidal solids as this material passes through the filter paper during the filtration step in the gravimetric estimation of suspended solids. Comparatively, colloidal particulates contribute significantly to light scattering in the water column and are adequately accounted for in the routine measurement of turbidity.

From Figure 30, it is seen that even in the face of a significant reduction in suspended solids in Lake Sorell, the influence of colloids in limiting light availability and reducing the euphotic depth remains highly significant. For example, a significant reduction in suspended solids in Lake Sorell to around 10 mg/L results in a euphotic depth of around 60 cm. A similar reduction in suspended solids in Lake Crescent returns a euphotic depth of approximately 1.9m. Comparatively, a reduction in turbidity to around 10 NTU in Lake Sorell, a level reached frequently in the past, increases the euphotic depth to around 2m, a depth that exposes a large area of the lake bed to solar radiation levels capable of sustaining plant growth.

The presence of high amounts of colloidal material in Lake Sorell is a critical issue when addressing the degraded state of the light climate. It is of particular relevance as the colloids show little evidence of settling out, even with the return of relatively high water levels and calm conditions experienced by the lakes during 2002 that resulted in a significant reduction in measured 'particulate' suspended solids.

Figure 30 Comparisons of euphotic depth versus turbidity and suspended solids in lakes Sorell and Crescent. Curves have been extrapolated from the relationships outlined in Chapter 2, Figure 11.



### 3.3.3 Modelling turbidity responses to water levels

The graphs derived from Carper and Bachmann's (1984) methods are presented in Figure 31. A significant increase in lake bottom area influenced by wind driven waves after 1998 is evident. The increase would result in sediment disturbance and entrainment in low energy accumulation areas, along with increasing the magnitude of disturbance in transportation and erosion areas of the lakes that were typically susceptible to sediment entrainment in the past.

The ability of sediments to withstand a limited degree of exposure is best illustrated by Lake Sorell in Figure 28. The lake bed appears to have an inherent ability to resist exposure at low lake levels as colloidal turbidity remains at background levels for an extended period of time as lake levels fall below a threshold level. At the threshold level, a lake level well below a point typically reached in past times, the well consolidated sediment second layer would be increasingly exposed and over time, as shear stress increases and the cohesive forces binding the sediments together is exceeded, entrainment of the sediment second layer results with the observed release of fine colloidal material into suspension. The critical shear stress depends on the

properties of the sediments. Newly deposited material typical of the sediment primary layer is more easily entrained than material which has been left undisturbed and compacted over a long period of time (Bengtsson, Hellstrom et al. 1990) such as is typical of the sediment secondary layer.

Even with lake volume increased well above a level at which colloids have been at low 'background' levels in the past, they seem to remain in suspension for an indefinite period. This contrasts with the observed lag in increased entrainment and release of colloids at low lake levels below a threshold level. The cycle continues as lake levels once again fall below the threshold with the sediments exhibiting a similar resistance to further disturbance and little resettling or sedimentation once the colloids have been suspended.

Lake Crescent, although showing a marked increase in colloids in recent years, has not experienced the same level of increase as Lake Sorell. It is possible that this is due to Lake Crescent being more turbulent than Lake Sorell, a point that, although not quantified, has been mooted in previous studies (Cheng and Tyler 1976b). This would increase the intensity and frequency of sediment entrainment in Lake Crescent increasing the opportunity for mobilisation and removal of fine particulates.

Figure 32 demonstrates that Lake Crescent has, on average, a more energetic environment than Lake Sorell. It depicts areas of the lake bed influenced by wind driven waves at three different westerly wind speeds for lake levels at the first quartile, median and 1 standard deviation below the mean level for the period January 1970 to December 1997 (lake level data summarised in Appendix 10). Lake Crescent clearly has a greater proportion of its bed in contact with the wave base than Lake Sorell, especially at higher wind speeds (~ 25 knots) meaning that Lake Crescent would be subject to stronger sediment disturbance at the lake levels experienced before the mid 1990s.

Figure 31 Lake level changes prior to and after 1998 and the increase in exposed lake bed for three wind speeds for commonly occurring westerly winds for both Lake Sorell and Lake Crescent.

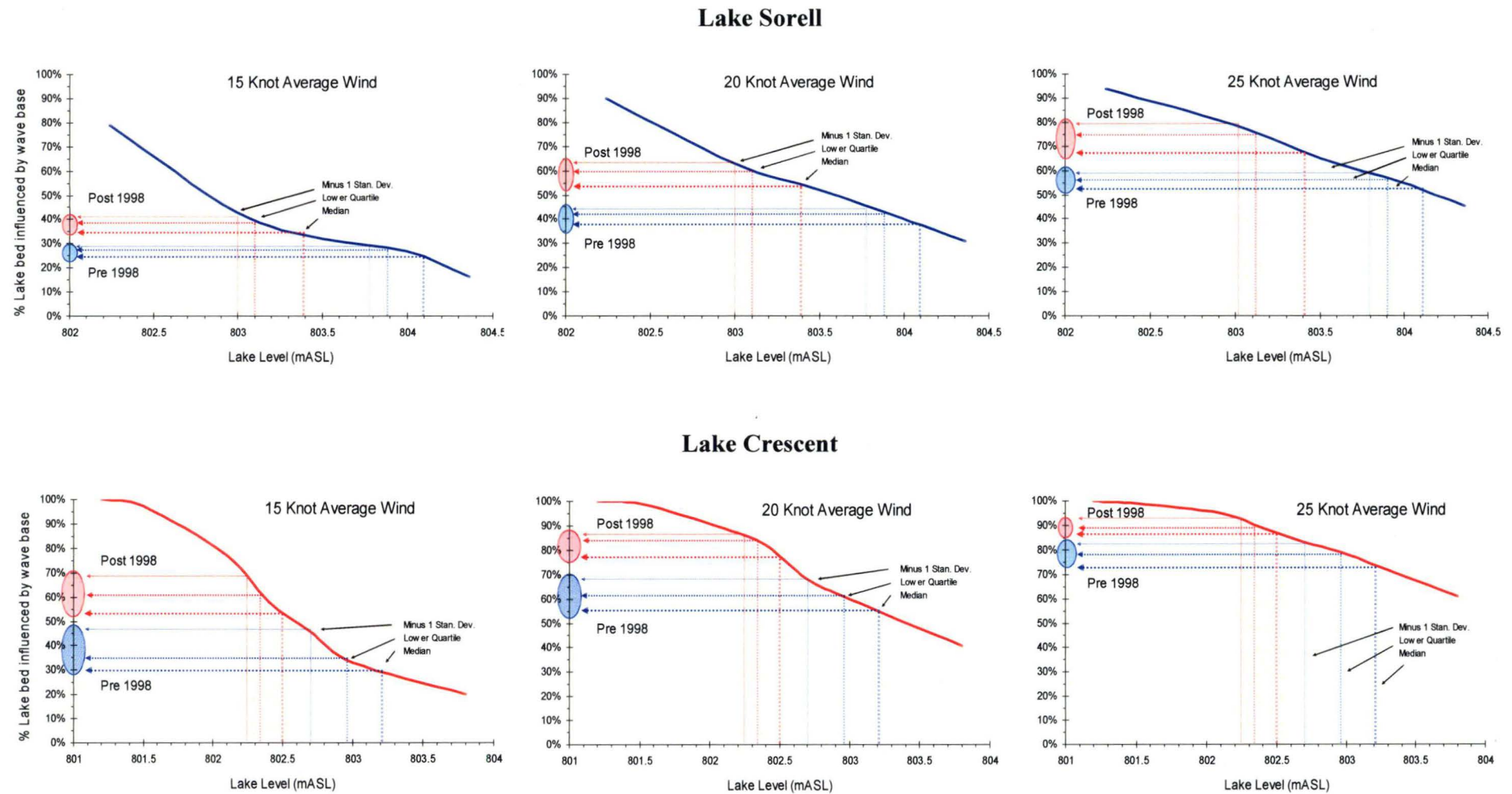
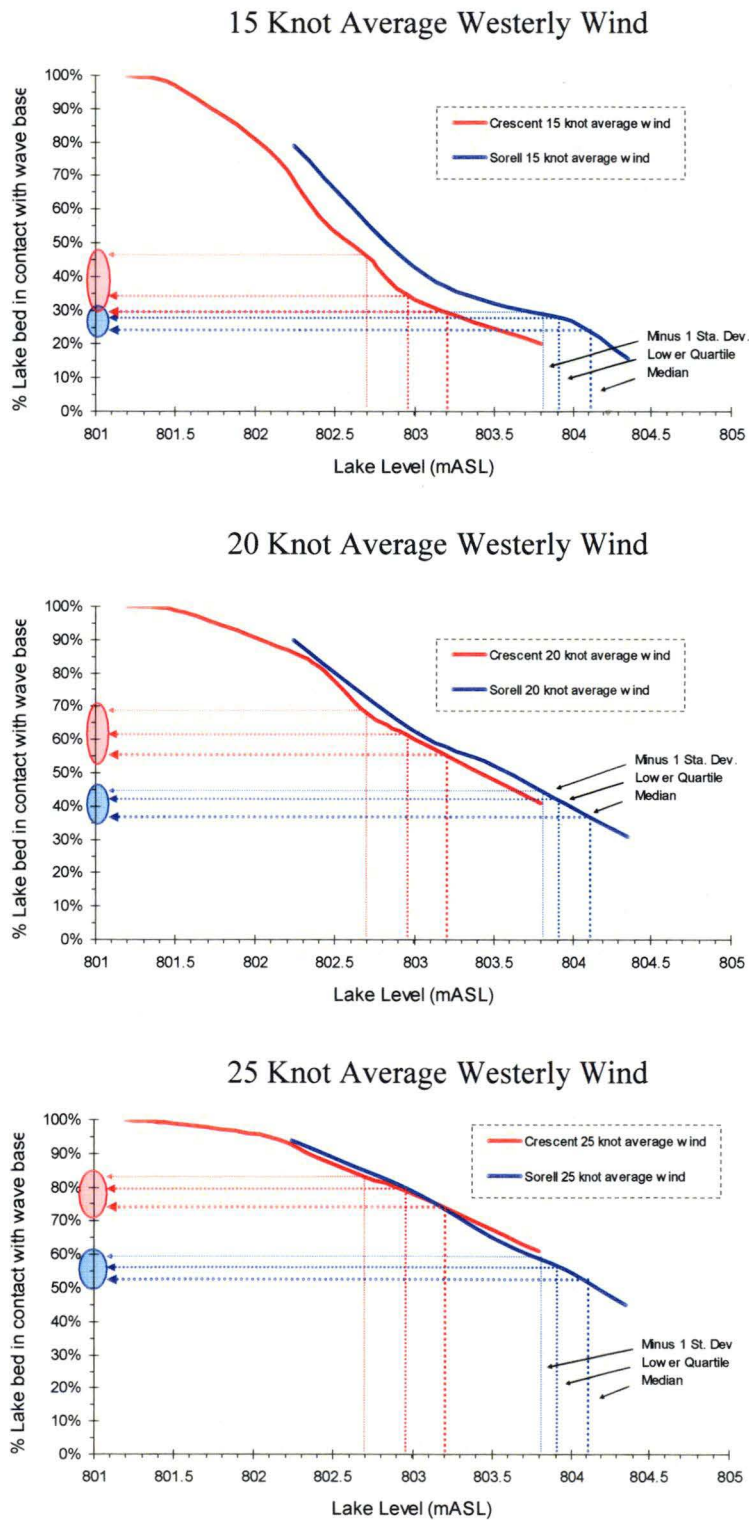


Figure 32 Comparison of areas of lake bed in contact with the wave base for lake levels commonly reached in Lake Sorell and Lake Crescent between 1970 and 1998.



Approximate water residence times are presented in Table 15 and Table 16. Rainfall landing directly on the surface of the lakes accounts for 1.5 to 2 times the total inflows of Lake Sorell and 1 to 3 times the total inflows for Lake Crescent. This reflects the low yield that the lakes receive from their catchments due to the low ratio of catchment area to lake surface area of the lakes (49% for Lake Sorell and 42% for Lake Crescent; (Cheng and Tyler 1973a)). Evaporative loss is also large: ~ 900 mm per annum. Moreover, the water residence times of both lakes are variable. Water exchange in Lake Crescent depends strongly on water release from Lake Sorell, and these releases fluctuate considerably between years as highlighted by the four fold difference in water residence times of Lake Crescent for 2000/2001 compared with 2001/2002.

Table 15      Approximate water input volumes and water residence time for Lake Sorell for July 2000 to June 2001 and July 2001 to June 2002 assuming lake volume to be at an average level typical of the lake (1970 – 1997; average level 804.09 mASL = 142 000 ML). Average evaporative loss for Bushy Park ~ 2.5mm per day = 910 mm per year.

Year	Rainfall (mm)	Rainfall Volume (ML)	Evaporative Loss (Avg - ML)	Inflow Volume (ML)	Net Volume Input (ML)	Water residence time at avg. level (yrs)
07/00 – 06/01	733	37750	-45000	17600	<b>10350</b>	13.7
07/01 – 06/02	590	30400	-45000	20700	<b>6100</b>	23.3



Table 16 Approximate water input volumes and water residence time for Lake Crescent for July 2000 to June 2001 and July 2001 to June 2002 assuming lake volume to be at an average level typical of the lake (1970 – 1997; average level 803.30 mASL = 34 000 ML). Average evaporative loss for Bushy Park ~ 2.5mm per day = 910mm per year.

Year	Rainfall (mm)	Rainfall Volume (ML)	Evaporative Loss (Avg - ML)	Inflow Volume (ML)	Net Volume Input (ML)	Water residence time at avg. level (yrs)
07/00 – 06/01	733	14250	-17460	4400	<b>1190</b>	28.6
07/01 – 06/02	590	11450	-17460	10650	<b>4640</b>	7.3

What may be more informative as an indication of possible export rates of fine colloidal material is the comparison of water releases from both lakes as a fraction of lake volume for both the 2000/2001 and 2001/2002 irrigation seasons (Table 17). It is evident that relative water release is greater from Lake Crescent than Lake Sorell, leading to an increased relative transport of solutes and suspensoids from Lake Crescent.

Table 17 Water release from Lake Sorell and Lake Crescent for the 2000/2001 and 2001/2002 irrigation seasons and the change relative to lake volumes.

Year	ML Release	Average Lake Volume (ML)	% Volume Released
Sorell Nov 00 - Apr 01	2300	96000	2%
Sorell Oct 01 - May 02	7200	122000	6%
Crescent Nov 00 - Apr 01	4360	18100	24%
Crescent Oct 01 - May 02	3080	27400	11%

Additional to this, the maximum allowable water allocation available to downstream irrigators for water released from Lake Crescent is 10 000 ML per season, representing an average allowable release of approximately 30% of lake volume per

annum, assuming a lake volume typical of the period running 1970 and 1997. A comparable release of 10 000 ML per annum from Lake Sorell would represent a release of 7% of total lake volume at a comparable average lake level.

The importance of dilution and concentration on colloidal turbidity are evident in Table 18. The modelling worked well as the expected colloidal turbidity closely reflected observed colloidal turbidity for the time periods in question. With Lake Sorell maintaining a relatively high lake level for 2002, little evidence of further entrainment of colloids was detected (Figure 26), with the observed increase in colloidal turbidity over January 2002 being attributable to concentration of surface waters through evaporative loss from Lake Sorell (Table 18). Obviously, at low lake levels, further sediment disturbance and entrainment of colloidal material would be expected.

Table 18 Changes in colloidal turbidity in Lake Sorell assuming dilution from inflow water and concentration by evaporation to drive colloid levels for the season in question. The 'Difference' is equal to the actual measured colloidal turbidity minus the expected or predicted colloidal turbidity.

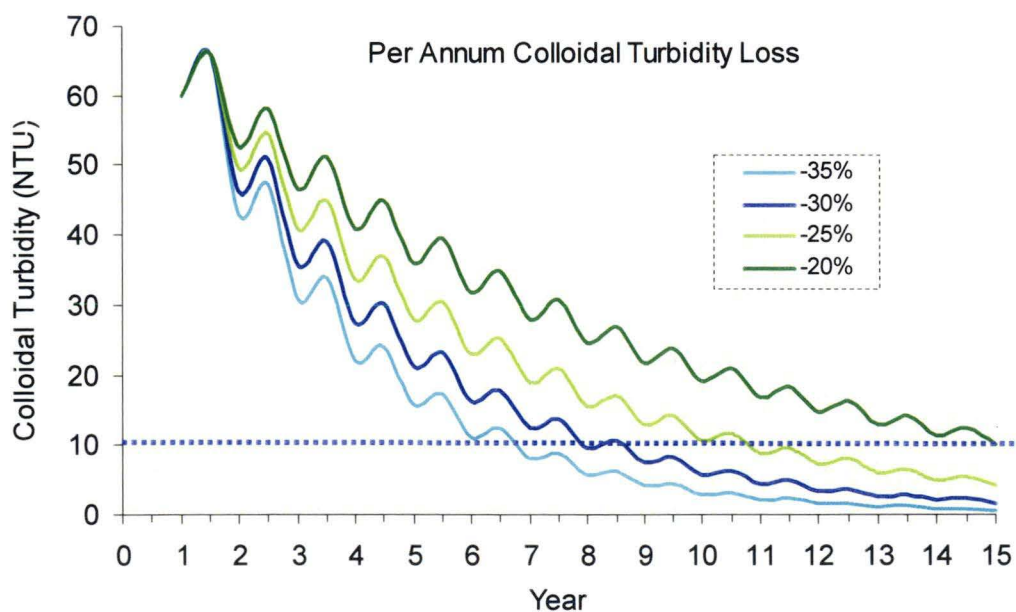
Lake Sorell	Season	Initial Colloidal Turbidity (NTU)	Initial Volume (ML)	Final Volume (ML)	Expected Colloidal Turbidity (NTU)	Actual Colloidal Turbidity (NTU)	Difference (NTU)
Inflow - dillution	May 01 - Dec 01	93.0	90150	127300	65.9	59.0	-6.9
Evaporation - concentration	Dec 01 - June 02	59.0	127300	116500	64.5	65.0	+0.5
Inflow - dillution	May 03 – Oct 03	69.3	99200	140100	49.1	53.1	+5.4
Evaporation - concentration	Dec 03 - May 04	53.1	134400	122900	58.1	60.7	+2.6
Inflow - dillution	May 04 - Aug 04	60.7	122900	137500	54.3	53.1	-1.2

If, in the absence of further entrainment of colloids at high water levels, and assuming the dominant mechanism limiting and reducing colloids is dilution and flushing, there is potential for colloidal turbidity to remain at high levels for a considerable period of time. To date, observed reductions in colloids are almost directly proportional to lake volume change (Table 18). Additionally, increases in colloidal turbidity over the summer-autumn period of 2002 and 2003 can be largely accounted for by changes in concentration of lake water through evaporative loss which results in an approximate



10% reduction in lake volume for a comparable time period. Taking Lake Sorell as an example, and assuming the lake to be held at full supply with a possible net loss of 30% of lake volume being directly proportional to inflow volume as observed over 2000 and 2001, and an average decrease in lake volume of approximately 10% through evaporative loss, Figure 33 outlines the possible scenarios for changes in colloidal turbidity over time. A colloidal turbidity of 60 NTU as observed over 2003 was used for the initial level. From Figure 33 it is seen that even with a best case scenario of a net 30% loss, colloidal turbidity will maintain high background turbidity levels for a significant number of years. In all likelihood, the actual net loss of colloids from Lake Sorell will be much lower than 30% per annum, and in the case of a 20% loss per annum, a significantly higher net loss than has been observed over the last three years, with no further entrainment of this material, colloidal turbidity could remain above historic levels for over 15 years.

Figure 33 Estimated changes in colloidal turbidity over time in Lake Sorell assuming the overriding mechanism for change is dilution, flushing and evaporation. The dashed line represents colloidal turbidity levels typical of Lake Sorell for the period running 1996 to 1998.



### **3.4 Discussion**

#### **3.4.1 Temporal trends in light climate**

It is concluded that inorganic sediment and dissolved colour and colloids are largely responsible for the degradation in the light climate of both lakes. These variables have been derived internally, as the work undertaken quantifying inputs from subcatchments for the same period have shown imports of inorganic sediment to be negligible (Section 2.3.4; Appendix 8), with inflows being of extremely good water quality. It is widely accepted that inorganic suspended sediment in particular and organic sediment to a lesser extent is influenced heavily by the process of sediment resuspension (Scheffer 1998).

Therefore:

- Resuspended inorganic material and dissolved colour and colloids are responsible for the high levels of turbidity, light attenuation and reduced water clarity.
- Autochthonous material such as algae and detritus do not influence turbidity significantly.
- At times of reduced levels of suspended sediment, dissolved colour and colloids dominate.

Under current environmental conditions, the extreme degradation of water clarity can be attributed to high levels of inorganic suspended sediment, dissolved colour and colloids. Wind induced sediment resuspension appears to be the mechanism leading to increased tripton levels. Effective management relies on understanding the extent and severity of sediment resuspension and identifying possible ways to alleviate the problem.

#### **3.4.2 Lake levels**

It is evident that the degradation in water quality is largely caused by the entrainment of sediment through sediment resuspension events. As mentioned in Chapter 1, the frequency, extent and severity of wind resuspension is related directly to water depth, wind speed and fetch. Understanding the relationship between lake level, sediment

disturbance and water clarity is important in understanding the interactions between low water levels and the current degradation in water quality.

It appears several factors combine to influence colloidal concentrations in these two lakes. The increased 'average' bed exposure typical of Lake Crescent results in greater sediment disturbance and increased sediment sorting that over an extended period of time; would release a significant amount of available colloidal material from the primary sediment layer. This, coupled with increased water residence time and higher relative volume release of water, combines to remove a greater proportion of colloids from the system, resulting in a reduced amount of fine particulate material available in Lake Crescent for resuspension. This is reflected in the lower levels of colloidal turbidity reached in Lake Crescent compared to Lake Sorell (approximately 30% lower) over 2000 and 2001, even in the face of extensive bed erosion, reflecting a reduced level of 'new' material exposed and resuspended.

Additionally, lake management practices in Lake Crescent in the period predating 1998 periodically reduced water levels to levels that compare to low water levels reached between 1999 and 2003 (Appendix 10). Therefore, the increase in bed exposure and sediment disturbance experienced by Lake Crescent between 1999 and 2003 have likely been experienced in the past. This would mean that sediment being entrained during 1999 to 2003 would likely have been exposed and entrained in the past giving the opportunity for the release and export of a significant fraction of the fine colloidal fraction that would be present in the sediments.

In the case of Lake Sorell, a reduced level of bed exposure, even at high sustained wind speeds typical of the lake for the period preceding 1998 would give rise to the occurrence of low energy areas of sediment accumulation where settlement of fine material may result. The sediment primary layer in Lake Sorell in high to medium energy areas would also be reduced below which the lake bed would contain a significant amount of colloidal material as the sediment would likely have never been exposed to the extent that the low lake levels post 1998 resulted in. This coupled with low water residence time and low relative volume release of water would help to retain fine colloidal material within the lake and retard export of this material from the system.

Post 1998, in the face of increased sediment exposure in both lakes, resulted in different responses of colloidal turbidity in the two systems. The record low lake levels reached for sustained lengths of time over 2000 and 2001 resulted in a marked increase in the frequency and intensity of sediment entrainment in both systems. The significant lag in increasing colloids in Lake Sorell in the face of decreasing lake levels reflects erosion and entrainment of material below the sediment primary layer into the sediment secondary layer.

The sediment secondary layer, being well consolidated would take increased and sustained exposure to overcome the cohesive forces of the sediments and initiate sediment resuspension as the fine clay particles tend to become cohesive and therefore much more difficult to resuspend if left undisturbed for a significant period of time (Rasmussen and Rowan 1997). This layer, having been little disturbed in the past, would contain increased amounts of fine colloidal material compared to the sediment primary layer.

The increase in erosion depth due to increased and sustained shear stress at the sediment surface penetrated below the primary layer and released into suspension significant colloidal material bound in the sediment secondary layer. This material has continued to remain in suspension with little evidence of settling out. Combined with low water residence time and low relative volume release in Lake Sorell, colloidal turbidity appears likely to maintain significantly high levels of background turbidity for a long period of time. Basic empirical modelling, assuming a best case scenario in which lake levels remain above a level that will eliminate further exposure of the sediment secondary layer and limit further release of colloids, predicts that with a generous exchange of water each season that colloidal turbidity will remain well above the level of 10 NTU typical of Lake Sorell in the past for 7 to 15 years.

From the relationships determined between suspended solids, light attenuation and euphotic depth, it is shown that even with an extreme decrease in suspended 'particulate' material down to levels around 10 mg/L, the presence of colloidal suspensoids will continue to limit the euphotic depth to around 60 cm. The extreme impact colloidal turbidity has on limiting light availability, reducing water clarity and

increasing the transport of plant nutrients such as nitrogen and phosphorus in both lakes Sorell and Crescent, combined with the dependence on dilution and flushing to decrease colloid concentrations, especially in the case of Lake Sorell, reflects the utmost importance of reducing the level of bed exposure in the lakes to limit and ultimately eliminate the release of this material in the water column. Initial observations identified a critical water level in Lake Sorell below which colloid concentrations increase. This 'critical' level lies at approximately 803.2 mASL. Future management of Lake Sorell should maintain lake levels well above this threshold level to reduce the likelihood of further increases in colloidal turbidity.

The highly significant relationships between decreasing water level and increasing turbidity in both lakes Sorell and Crescent coupled with increased hysteresis in measured suspended sediment concentrations and release of fine colloidal particulates into suspension well beyond levels documented in the past points strongly towards increased sediment resuspension as being responsible for the severe degradation in water quality. A detailed investigation is therefore warranted to determine the extent of sediment resuspension at various lake levels in both systems to determine accurately the importance this process has on defining ecosystem behaviour and to determine if lake level management may be a solution to a problem that has likely resulted in the wide scale loss of aquatic macrophytes from Lake Sorell and a severe degradation in the water quality of both lakes.

Erosional areas are typical of near-shore bottom areas where sediments are frequently eroded. These areas, due to resuspension occurring often, have a limited amount of material available for resuspension (Bengtsson, Hellstrom et al. 1990). Further from shore the bottom is scoured less regularly, and sediments are deposited to and entrained at irregular intervals, these areas represent transportation areas. The sum of the erosion and transportation areas of a lake represent the total lake area influenced by sediment resuspension (Hakanson and Jansson 1983). The remaining areas of the lake represent accumulation zones where fine material settles out and sedimentation is final (Blais and Kalff 1995) and, if left undisturbed consolidates over time (Bengtsson, Hellstrom et al. 1990) and is thus removed from the cycle of resuspension and sedimentation.

The increase in disturbance of sediments in Lake Crescent would lead to increased sediment sorting and a reduction in low energy areas where fine sediments would accumulate (Hakanson 1982; James and Barko 1993). The increased turbulence would likely increase the depth of the sediment primary layer and lead to a greater degree of release of fine particulates into suspension, this material would have a greater opportunity to be removed from the system through water release. This would reduce the amount of colloidal material available for resuspension and explain why colloid levels in Lake Crescent were markedly lower than Lake Sorell, even at the extreme low lake levels reached over 2000 and 2001.

From Table 15 through Table 17, it is evident that Lake Crescent has a significantly higher water residence time on average than Lake Sorell along with a higher relative outflow volume. This coupled with the more dynamically turbulent environment of Lake Crescent that increases the relative frequency and extent 'on-average' of sediment resuspension, would result in higher entrainment and flushing of fine particulate material from Lake Crescent compared to Lake Sorell

Additionally, it is worth noting that on a lake loading basis, Lake Sorell water release flows into Lake Crescent so for the past few years significant volumes of colloidal material would have been discharged into Lake Crescent. However, in years previous to the increase in colloidal loading in Lake Sorell, the water flowing into Lake Crescent from Lake Sorell would have had a low colloid concentration increasing the net export of colloidal material from Lake Crescent.

The loss of colloidal material through sedimentation under current environmental conditions appears largely insignificant, with observed reductions that coincided with increased lake levels appearing to be dependent on dilution from inflows combined with a marginal export from the system through water release. Additionally, at high lake levels in the absence of further entrainment, the reduction in lake volume that occurs each summer through evaporative loss results in a comparable increase in colloidal turbidity, almost directly proportional to the change in lake volume.

The lack of evidence of any net sedimentation of colloids in Lake Sorell reflects the colloids behaving largely as a conservative substance, and if so, colloid concentrations will be dependent on inflow/outflow volumes and the water residence times of the lakes (Hakanson and Jansson 1983). For example, the residence time of a chemical or fraction may be defined as (Hakanson and Jansson 1983):

$$T_r = V * C / Q * C_{in} \quad 3.3$$

Where  $T_r$  is the residence time of the fraction in question in time units,  $V$  is the lake volume,  $C$  the lake concentration or output concentration,  $Q$  the water discharge and  $C_{in}$  the input concentration. From the detailed work carried out on subcatchment inputs of colloidal and suspended solids, it is possible to assume the inflow concentration to be negligible. Therefore, from equation 3.3, the residence time of the colloids, if behaving as a conservative substance will be heavily reliant on the discharge to lake volume ratio in exporting material from the system, coupled with continued inflows to maintain increased lake levels to retard any further entrainment of this material.

Therefore, reductions in colloidal turbidity will depend largely on high inflow volumes diluting colloids coupled with release of significant volumes of water from the lakes whilst maintaining lake levels as high as possible to limit further sediment disturbance so that colloids do not increase further.

## **Chapter 4   Modelling wind resuspension in lakes Crescent and Sorell**

### **4.1   Introduction**

Evidence from observations on changes in water clarity, nutrients and suspended particulate material (SPM) from 1996 to 2001 indicates increasing levels of sediment entrainment as being largely responsible for the steep decline in water quality in both lakes Sorell and Crescent. The increased turbidity and extreme hysteresis in measured SPM concentration coinciding with sustained record low lake levels over 2000 and 2001, coupled with the release of large amounts of colloidal particulate material from the sediments to levels far in excess of previous observations suggests that an increase in the frequency, extent and severity of sediment resuspension is the cause for the degradation in water clarity.

A detailed investigation was therefore warranted to explore the dynamics of sediment resuspension in lakes Sorell and Crescent and to quantify the interaction between sediment resuspension and changing lake depth. Understanding resuspension dynamics in the lakes should allow appraisal of realistic management options available to reduce sediment resuspension. The aims of such actions are to improve water clarity, increase light availability and provide a better environment for the re-establishment of aquatic macrophytes in both lakes, and these aims are consistent with the goals of improving the health and aesthetic value of shallow lakes elsewhere (Van Duin, Blom et al. 1992; Blom, van Duin et al. 1994; James, Best et al. 2004). Re-establishment of aquatic macrophytes through an improved light climate and a reduction in physical disturbance (Spence 1982) should, in turn, provide further reductions in sediment resuspension (Jackson and Starrett 1959; Dieter 1990; James and Barko 1990; James and Barko 1994; Koch 1996; Barko and James 1998; James, Barko et al. 2004).

From historical anecdotal evidence, both lakes have been susceptible to episodic periods of high turbidity which were attributed to high winds during storms and extremes in weather in the area. Although turbid at times, both lakes returned to their characteristic states over a short time period of days to weeks. The difference with the



situation prevailing in the late 1990s and early 2000s is that high SPM loading has been maintained for a significant period of time along with extremes in the values of SPM recorded in the lakes.

Several additional characteristics typical of these lakes indicate the importance of sediment resuspension. Evident in Lake Crescent, and in Lake Sorell to a lesser degree, is a significant amount of sediment sorting in a west to east direction (prevailing winds) reflecting disturbance and sorting of sediments (Hakanson and Jansson 1983) with the eastern shores of Lake Crescent having sand moraines present along the shoreline and extensive sand flats (Cutler, Kinrade et al. 1990) extending into the lake reflecting high energy erosion zones. The contrast is the lee western shores which are dominated by fine silts and clays and vegetated wetland areas (Cutler, Kinrade et al. 1990) likely reflecting lower energy zones of accumulation and transportation (Hakanson and Jansson 1983). The strong supporting evidence detailed in Chapter 2 and Chapter 3 showing increased water column loading of suspended solids to be derived from internal sinks as opposed to increased sediment loading from the catchment, further substantiates increased sediment resuspension to be highly significant and the dominant process influencing water quality in the lakes.

Sediment resuspension is primarily a function of wave height and water depth (C.E.R.C. 1977). In shallow lakes, reducing water depth usually increases the area of the lake bed affected by wind generated waves at a given wind speed, and lakes with large shallow areas exposed to high winds, such as lakes Sorell and Crescent, are most susceptible to sediment resuspension (Jackson and Starrett 1959). Decreasing water levels have been identified as being the cause of increased turbidity in studies elsewhere, with Nolen et al. (1985) finding changes in average depth alone to explain 33% of variation in measured turbidity.

Resuspension of sediments takes place when the shear stress (i.e. the force exerted at the sediment surface, measured in Newtons per square metre ( $\text{N m}^{-2}$ ) or dynes per square centimetre ( $\text{dynes cm}^{-2}$ )) is greater than the cohesive forces binding the sediments together (Bloesch 1995), resulting in sediment entrainment (Douglas Evans 1994; Bailey and Hamilton 1997). Resuspension is, therefore, partly a function of the properties of the sediments which, in turn, depend on numerous factors such as

sediment grain size, water content, density and organic content (Bloesch 1995). The frequency that sediments are disturbed also influences the critical shear stress because sediments that have been left undisturbed consolidate over time and require more energy to resuspend than sediments that are more frequently entrained (Bengtsson, Hellstrom et al. 1990). These factors combine to determine the level of cohesion of the sediments and the critical shear stress threshold necessary for sediment entrainment to occur. The critical shear stress threshold may therefore be system specific as the nature of sediments change markedly between water-bodies.

An additional complexity results from human manipulations of lake levels. These changes in level interact with the bathymetry of the lakes to expose different quantities of the lake bed to a critical shear stress. These changes inevitably interact with prevailing climatic conditions (especially wind speed and direction), which make it difficult for managers to make quantitative predictions about different management scenarios that might be pursued. Modelling procedures of the type described in Chapter 1 promise to make such predictions, and the broad goal of this chapter was to evaluate the performance of the purely physical attributes of the DYRESYM – CAEDYM modelling framework based on a 2-year data set from lakes Crescent and Sorell, and then use the model to make some provisional contrasts between potential options for managing the levels of these two lakes.

DYRESM-CAEDYM, the coupled one-dimensional hydrodynamic and ecological model summarised in Chapter 1, was used here because it has been designed to model two separate size classes of suspended solids to account for the fast and slow settling velocities of the coarse and fine fractions of resuspended sediment (Herzfeld and Hamilton 2000). This is important for accurately modelling the variable characteristics of particles in suspension (Weyhenmeyer 1998). DYRESM-CAEDYM predicts horizontally averaged suspended solids concentration and therefore reflects an average ‘lake-wide’ suspended solids concentration in a shallow turbulent water body that do not undergo significant levels of stratification, such as lakes Sorell and Crescent (Cheng and Tyler 1973a).

The specific steps of this process were as follows. First, local estimates of the value of the critical shear stress were made from field data, and these estimates were cross-

checked with observed resuspension events in the time period of intensive modelling. Second, wind and hypsographic data for these lakes was collated to estimate the changes in the area of the lake bed subject to shear stresses greater than the critical value under different lake levels. Finally, DYRESM – CAEDYM models were calibrated using physical data alone, and then used to simulate different scenarios of lake level manipulation, and the results evaluated in terms of the observed behaviour of these lakes over the past 6 years. If a relationship between increasing shear stress and changes in observed SPM concentrations is established, then the role sediment resuspension plays in driving ecosystem process can be quantified and investigated under various theoretical management regimes, thus allowing possible management scenarios to be scrutinised and the benefits of each assessed.

## **4.2 Methods**

For both lakes Sorell and Crescent, detailed information was collected over two extended time periods that allowed comparisons to be made between changes in SPM concentration and estimates of shear stress at the sediment surface calculated from information on effective fetch, wind speed and water depth.

### ***4.2.1 In-situ sediment resuspension measurements***

A Greenspan TS300 (0-500 NTU) (Greenspan Technology Pty. Ltd., Warwick, Queensland, Australia) turbidity logger was deployed for extended periods of time in both lakes Sorell and Crescent on two separate occasions at extreme low and moderately high water levels (Table 19). Measurements of turbidity were made at half-hourly intervals.

An accurate relationship between turbidity and SPM concentration was determined for each lake using linear regression analysis (Appendix 7) from data collected during routine sampling trips (Chapter 2, Section 2.2.2– 2.2.3). Using the regression relationships, continuous records of SPM concentration were estimated from turbidity logger measurements (Gippel 1989; Hawley and Lesht 1992).

Table 19 Time periods of intensive sediment resuspension sampling in lakes Sorell and Crescent.

Lake Level	Lake Sorell	Lake Crescent
Extreme Low Water Level	18/8/00 – 6/9/00	9/9/00 – 24/9/00
Moderately High Water Level	28/2/02 – 5/4/02	30/11/01 – 12/12/01

#### 4.2.2 Wind, fetch and depth characteristics

The location of the turbidity logger was determined by GPS. Wind speed and direction measurements were made at hourly intervals from a wind anemometer located on a reef offshore in Lake Sorell (Chapter 2, Figure 1).

Effective fetch ( $EF$ ) was calculated in preference to fetch ( $F$ ) as effective fetch takes into account the influence of shoreline irregularities on wave generation for angles up to  $42^\circ$  from the direction of the wind (C.E.R.C. 1977). Effective fetch is computed by measuring the distance from land,  $X_i$ , for the main axis of the wind direction and 7 radials at  $6^\circ$  intervals on either side of this axis and multiplying the length of each radial by the square of the cosine of the corresponding angle,  $A_i$ , and dividing the sum of these 15 products by 13.5 (Gons, Veeningen et al. 1986):

$$EF = \frac{\sum \left( X_i (\cos A_i)^2 \right)}{13.5} \quad 4.1$$

For each observation of wind direction, the effective fetch of the octant closest to the recorded direction was used to approximate the effective fetch of the point of interest for the preceding 4 hours. The effective fetch, measured wind speed and water depth were used to model shear stress using equations 4.2 to 4.6 for the period of interest.

Water depth was measured directly and related to daily lake level records for the duration of the run.

### 4.2.3 Modelling shear stress

#### 4.2.3.1 Estimating critical shear stress, $\tau_c$

Bottom shear stress,  $\tau$ , may be estimated from laminar wave theory and is a function of wind velocity, effective fetch and wind direction.  $\tau$  may be estimated from the following equation (Luettich, Harleman et al. 1990; Bailey and Hamilton 1997; James, Barko et al. 2004):

$$\tau = H \left[ \frac{\rho (v (2\pi / T)^3)^{0.5}}{2 \sinh(2kD)} \right] \quad 4.2$$

where  $\tau$  ( $\text{N m}^{-2}$ ) is the calculated bottom shear stress,  $H$  is the wave height (m),  $\rho$  is the density of water ( $\text{kg m}^{-3}$ ),  $v$  is the kinematic viscosity of water (at 20 °C,  $0.001005 \text{ kg m}^{-1} \text{ s}^{-1}$ ; (Stumm and Morgan 1996)),  $T$  is the wave period (seconds),  $k$  is the wave number ( $2\pi/L$  where  $L$  = wavelength, m) and  $D$  is the water depth (m). Wave characteristics ( $H$ ,  $T$  and  $L$ ) may be calculated using the Sverdrup-Munk-Bretschneider (SMB) shallow water wave equations (C.E.R.C. 1977) using information on wind speed and duration, wind fetch and average water depth (Sheng and Lick 1979).

When Airy wave theory is able to be applied, the wavelength,  $L$ , may be calculated by (Hamilton and Mitchell 1996):

$$L = \frac{gT^2}{2\pi} \tanh\left(\frac{2\pi D}{L}\right) \quad 4.3$$

Where  $T$  is the wave period (s). As  $L$  appears on both sides of the equation, an approximation of equation 4.3 is given by (Herzfeld and Hamilton 2000):

$$L = 2\pi D \frac{\left[ \tanh\left(\frac{4\pi^2 D}{gT^2}\right) \right]^{0.5}}{\left(\frac{4\pi^2 D}{gT^2}\right)} \quad 4.4$$

And for wave height ( $H$ ) and wave period ( $T$ ) (C.E.R.C. 1977):

$$H = \frac{0.283 U^2}{g} \tanh\left[0.530 \left[\frac{gD}{U^2}\right]^{0.75}\right] \tanh\left[\frac{0.0125 \left[\frac{gF}{U^2}\right]^{0.42}}{\tanh\left[0.530 \left[\frac{gD}{U^2}\right]^{0.75}\right]}\right] \quad 4.5$$

and:

$$T = \frac{2.4\pi D}{g} \tanh\left[0.833 \left[\frac{gD}{U^2}\right]^{0.375}\right] \tanh\left[\frac{0.077 \left[\frac{gF}{U^2}\right]^{0.25}}{\tanh\left[0.833 \left[\frac{gD}{U^2}\right]^{0.375}\right]}\right] \quad 4.6$$

where  $H$  is the wave height (m), and  $T$  the wave period (s).  $F$  is the fetch (m),  $U$  the wind speed 10m above the surface of the water ( $\text{m s}^{-1}$ ) and  $D$  the lake depth (m).  $g$  is the gravitational constant ( $9.81 \text{ m s}^{-2}$ ).

Having calculated shear stress, the relationship between increasing shear stress and SPM concentration was analysed with piecewise linear regression in order to determine the critical or threshold shear stress,  $\tau_c$ , needed to initiate sediment resuspension.  $\tau_c$  was estimated at the inflection point where SPM concentration increased significantly above background conditions (James, Best et al. 2004). Seven relatively isolated resuspension events were identified from the empirical record where shear stress had remained below  $0.03 \text{ N m}^{-2}$  and  $\text{SPM} < 35 \text{ mg L}^{-1}$  for at least 6 h. These criteria avoided interference and hysteresis effects from antecedent resuspension events. SPM lagged by 1 h was then regressed against shear stress, with

the lag justified on the grounds that resuspension would not be instantaneous once a threshold had been attained. The seven estimates of  $\tau_c$  were then averaged, and the 95% confidence interval computed using conventional methods (Quinn and Keough 2002).

The decision to use an in situ estimate of  $\tau_c$  is further justified by James, Barko et al. (2004) who compared laboratory estimates of  $\tau_c$  to in-situ estimates of  $\tau_c$  based on  $\tau$  calculation from wave theory; they concluded that in-situ estimates were closely comparable with controlled laboratory estimates.

The estimate of  $\tau_c$  was further evaluated by plotting depth of water minus the depth of the wave base against modelled shear stress to estimate the point at which the wave base interacts with the sediments (Carper and Bachmann 1984). The depth of the wave base is a function of wavelength which was calculated from wind speed and fetch (Section 4.2.3) and the shallow water wave forecasting models of the Sverdrup-Munk-Bretschneider (SMB) method (C.E.R.C. 1977; Sheng and Lick 1979; Hamilton and Mitchell 1996). The point at which wind driven waves ‘touch’ the lake bed (i.e. the depth of the wave-base) is defined as the point at which the depth of water is less than one-half the wavelength (Carper and Bachmann 1984; Arfi, Guiral et al. 1993; Scheffer 1998). This was done for 745 points across Lake Sorell at a depth of 803.5 mASL and 266 points across Lake Crescent at a depth of 802.7 mASL for a westerly wind of an average speed of 20 knots. Where the estimate of water depth minus wave base depth intersect the  $x$ -axis of these plots shows where water depth and wave base are equal and should correspond with a value close to  $\tau_c$ .

#### **4.2.3.2 Effects of changes in lake area on shear stress**

Having determined  $\tau_c$ , coupled with information on changes in SPM concentration with increasing shear stress, it was possible to model areal changes in shear stress across both lakes Sorell and Crescent for various water depth and wind speed scenarios. Based on historical water level records, chosen water depths approximated low, medium and high lake levels for the lakes. Included were specific water levels of interest such as the “sill” which corresponds to the level to which the lakes may be drawn down via water extraction. Modelled wind speeds ranged from light to gusty conditions that were typical of the area. From this it was possible to draw conclusions

as to the likely impacts modelled shear stress would have on sediment resuspension dynamics.

This was achieved by taking wind speed, fetch and water depth characteristics of 745 points across Lake Sorell and 266 points across Lake Crescent and modelling shear stress exerted at the sediment surface for differing water depths and wind speeds. The results were then displayed via contour maps.

#### **4.2.4 Initialisation data for DYRESM-CAEDYM**

DYRESM-CAEDYM requires a number of initialisation files specific to the lakes in question. Files include detailed meteorological information, accurate estimates of the volume and composition of inflows, data on withdrawal volumes and initial conditions in the lakes in regards to composition, lake level and temperature. A comprehensive set of initialisation files were constructed for the period running from the 1<sup>st</sup> of January 2000 to the 31<sup>st</sup> of December 2001.

##### **4.2.4.1 Meteorological information**

DYRESM-CAEDYM requires a detailed ASCII file of meteorological information that is applicable to the region in question, for the duration of the simulation run.

Detailed meteorological information for the Interlaken area was obtained and collated for the period between January 2000 and March 2002. The DYRESM meteorological data file consists of seven columns including year and day number, short-wave radiation ( $\text{W m}^{-2}$ ), long-wave radiation (either  $\text{W m}^{-2}$  if measured directly or as decimal fraction of cloud cover), air temperature ( $^{\circ}\text{C}$ ), vapour pressure (hPa), wind speed ( $\text{m s}^{-1}$ ) and rainfall (m). Meteorological data were obtained from a number of sources including the Bureau of Meteorology, Hydro Tasmania and direct measurements made by the IFS. A section of the ASCII file of the meteorological information for the simulation run is given in Appendix 11. Details of the sources of data for each of these variables follow.

###### **4.2.4.1.1 Short-wave radiation**

Daily estimates of short-wave radiation ( $\text{W m}^{-2}$ ) were derived from satellite imagery processed by the Bureau of Meteorology from the Geostationary Meteorological



Satellite GMS-5 of the Japan Meteorological Agency. The satellite data site corresponding to the Interlaken area was located at 42° south, 147° east. Satellite estimates were compared to a limited number of local measurements made by Hydro Tasmania at their Lagoon of Islands weather monitoring station located approximately 15 kilometers west of lakes Sorell and Crescent. Both data sets correlated well (Appendix 12). It was therefore concluded that satellite estimates for the Interlaken area were accurate.

#### 4.2.4.1.2 Long-wave radiation

Daily values of long wave radiation were estimated by DYRESM from measurements of the decimal fraction of cloud cover and the water surface temperature calculated during the simulation. The most appropriate and closest station with measurements of cloud cover was the Bureau of Meteorology's Liawenee weather station located approximately 44 kilometres WNW of lakes Sorell and Crescent. Early values (1/1/00 – 17/1/01) were taken from station #96065. Values from 18/1/01 to 30/4/02 were taken from station #96033. Average daily values were obtained by taking the average between the 9am and 3pm observations.

#### 4.2.4.1.3 Air temperature

Average daily values of air temperature were obtained from the Bureau of Meteorology's Interlaken weather station serviced by Hydro Tasmania. The weather station is located at the Interlaken Canal. For missing values of air temperature, estimates were made from values obtained from Hydro Tasmania's Lagoon of Islands weather station. A regression relationship was derived between the air temperature at Lagoon of Islands and the air temperature at the Interlaken Canal (Appendix 12). This relationship was then used to approximate the air temperature at the Interlaken Canal for any missing data.

#### 4.2.4.1.4 Vapour pressure

Vapour pressure estimates were made from wet and dry bulb temperatures using the relationship given in Antenucci (2001) where vapour pressure ( $ea$ ):

$$ea = eas - 0.00066 \times (1 + 0.00115 \times qW) \times P \times (qD - qW) \quad 4.7$$

and  $qW$  = wet bulb temperature,  $P$  = atmospheric pressure,  $qD$  = dry bulb air temperature, and  $eas$  = saturation vapour pressure is calculated from:

$$eas = \exp \left[ 2.303 \left( \left( \frac{a \times qD}{a \times qD + b} \right) + c \right) \right] \quad 4.8$$

where  $a = 7.5$ ,  $b = 237.3$ ,  $c = 0.758$ , and  $qD$  = dry bulb air temperature ( $^{\circ}\text{C}$ ).

Wet bulb temperatures for the Interlaken area were estimated by taking the average between the vapour pressure measured at the Bureau of Meteorology's Tunnack and Liawenee weather stations. The two stations were located approximately 45 km SE and 44 km WNW from lakes Sorell and Crescent respectively. The average altitude above sea level between the two stations closely approximated the altitude of lakes Sorell and Crescent. It was therefore determined that these weather stations gave the best available estimate of vapour pressure for lakes Sorell and Crescent.

#### 4.2.4.1.5 Wind speed

Daily average estimates of wind speed were obtained from hourly measurements made by a wind anemometer deployed on a rocky reef in the middle of Lake Sorell (Chapter 2, Figure 1). Hourly measurements of average wind speed, maximum wind speed and wind direction were recorded from August 2000 to March 2003. For the period from January 2000 to August 2000, wind speeds for lakes Sorell and Crescent were approximated from recordings made by Hydro Tasmania at their Lagoon of Islands weather monitoring station, which was the nearest wind station to these lakes. Lagoon of Islands wind speeds were regressed against Interlaken wind speeds, and the derived relationship used to approximate Interlaken wind conditions from Lagoon of Islands observations (Appendix 12).

#### 4.2.4.1.6 Rainfall

Daily rainfall measurements were provided by the Bureau of Meteorology from their Interlaken weather station. Missing data values were substituted from records of daily rainfall obtained from a rain gauge maintained by the IFS at Lake Crescent.

#### **4.2.4.2 Morphometry and bathymetry**

DYRESM-CAEDYM requires detailed morphometry that describes the hypsographic curve of the lakes and consists of a table of elevations above a specified datum versus the cross-sectional area of the lake at that elevation (Appendix 11). Included in this file is information on the number of inflows entering the system, details specific to each inflow and the number of outlets from the system and the elevation of the outlets.

The hypsographic data was determined by digitising the bathymetric maps of both lakes Sorell and Crescent (Chapter 2, Figure 1 and Figure 2) and importing the images into the GIS software package MapInfo V6 for analysis of the surface areas of each contour. The bathymetric map of Lake Crescent was produced by the Survey and Geographic Information Department (1996) of the then Hydro Electric Corporation. The bathymetry of Lake Sorell was produced by the Rivers and Water Supply (1902) (Peterson and Missen 1979) and validated by the Survey and Geographic Information Department at the time Lake Crescent was surveyed in 1996. It was concluded that the bathymetric map of Lake Sorell created in 1902 described the lake bed satisfactorily. It was therefore assumed both maps described the depth characteristics of the lakes adequately to estimate lake volume and surface area changes with changes in lake level. The wetland areas of both lakes were excluded from the morphometry file so that only open water areas were used in the calculation of fetch and shear stress (Bachmann, Hoyer et al. 2000).

#### **4.2.4.3 Inflow volumes and water quality**

Inflows included for Lake Sorell were Mountain Creek, Silver Plains Creek, a small unnamed creek north of Silver Plains Creek and the drain discharging from Kemps Marsh. Inflows included for Lake Crescent were the Interlaken Canal and Agnews Creek. Water volume, temperature, salinity, total nitrogen, total phosphorus and suspended solids were the components estimated for each inflow for a period from the 1<sup>st</sup> of January 2000 to the 31<sup>st</sup> of December 2001 using the methods detailed in Chapter 2, Section 2.2.5. The resulting ASCII inflow file is presented in Appendix 11.

#### **4.2.4.4 Outflow volumes**

Accurate monitoring of the volume of water released from Lake Sorell into Lake Crescent was made during both the 2000 and 2001 summer periods. Discharge measurements were made by the Inland Fisheries Service, using the methods described in Chapter 2, Section 2.2.5. Detailed information of the volume of water released from Lake Crescent down the Clyde River for the 2000 and 2001 seasons was obtained from the Department of Primary Industries, Water and Environment's Water Resources Division. The resulting ASCII outflow file is presented in Appendix 11.

Detailed information on the remaining initialisation files for running DYRESM-CAEDYM can be found in Antenucci (2000). In all cases, field data collected during the course of the project was used.

#### **4.2.5 Calibration of DYRESYM and CAEDYM**

DYRESM was successfully initialised and run for both lakes and its accuracy was checked by plotting modelled lake levels and temperatures against observed values for the period in question. Because DYRESYM performed very well against these criteria (Section 4.3.2), modelling could progress to using CAEDYM.

For each parameter required by CAEDYM, information was available from the literature or calibrated directly from field data. Calibration of each variable progressed in a sequential manner with critical shear stress and sediment density being set first, and then progressing to calibrations of particle size and resuspension rate as detailed below.

##### **4.2.5.1 Critical shear stress**

Values of the estimated critical shear stress,  $\tau_c$ , determined by methods detailed in Section 4.2.3 and presented in Section 4.3.1 were used to set the critical shear stress in CAEDYM.

#### **4.2.5.2 Sediment density and particle size**

Based on work carried out by Hakanson (1977) a sediment density of  $2650 \text{ kg m}^{-3}$  was defined in CAEDYM as being representative of the density of the largely inorganic sediments of lakes Sorell and Crescent.

It was decided that the best approach to calibrating particle size and sedimentation rates in CAEDYM would be to look in detail at 'in-situ' settling velocities as opposed to laboratory analysis of the settling velocities of surficial sediments. This follows from work of Bailey and Hamilton (1997) who found discrepancies of almost two orders of magnitude in the settling characteristics of 'naturally' resuspended sediments as opposed to surficial sediments collected from cores.

To calibrate the particles sizes for the two separate fractions modelled in CAEDYM, resuspension events were identified in which modelled shear stress abruptly fell below the threshold value and approached zero. At this point, it was assumed that the behaviour of the particles in suspension could be adequately described by Stokes Law (see Chapter 1) thus leaving the particle size as the most significant factor determining settling velocities. Having identified several of these 'passive' sedimentation events with minimal applied shear stress it was possible to compare modelled output from CAEDYM for a number of different combinations of particle sizes to the measured 'in-situ' field data in order to calibrate particle sizes for the lakes.

#### **4.2.5.3 Resuspension rate calibration**

The final stages of calibrating DYRESM-CAEDYM to model sediment resuspension involved the calibration of the resuspension rate for the two separate size fractions of suspended solids. The sediment entrainment parameter cannot be readily calculated or measured directly and, as a result, calibration is achieved through adjustment of the entrainment parameter coupled with comparisons of modelled output with observed conditions (Bailey and Hamilton 1997).

Initial investigations were made into the effect of changing the resuspension rate (alpS) in CAEDYM on sediment resuspension for an initial 2 month period from April 2000. This represents the start date of collated meteorological data for these lakes. Four separate particle sizes were selected to reflect the major particle sizes in

the lakes (Appendix 13). This established initial base resuspension rates for a more detailed calibration of sediment resuspension outlined in Section 4.3.2.

An investigation into measured resuspension rates for several events in lakes Crescent and Sorell under low water and high water conditions were made to determine a minimum resuspension rate based on measured field data. Areal concentrations of suspended solids were calculated for each of four active resuspension events in Lake Crescent for a period of both low water level and moderately high water level, and five active resuspension events in Lake Sorell at moderately high water level. For each, it was assumed that the measured concentration of suspended solids approximated the average suspended solids concentration for the lake. Information on lake volume and surface area were then used to approximate the sediment concentration in  $\text{g m}^{-2}$  and, for the ascending limb of a specified resuspension event, the concentration was then regressed against time in seconds. The slope of this regression therefore approximated the resuspension rate in  $\text{g m}^{-2} \text{s}^{-1}$ .

During a sediment resuspension event, with an applied shear stress above  $\tau_c$ , both sediment resuspension and sedimentation occur simultaneously (Bailey and Hamilton 1997). The calculated ‘resuspension rate’ from such a regression analysis therefore approximates an absolute minimum rate as it does not take into account sedimentation rates. The values used for DYRESM-CAEDYM in practice will therefore need to be higher than the rates calculated using this analysis.

Once these “underestimates” of resuspension rates were derived, fine scale calibration was achieved by re-running the model for Lake Crescent at a half-hourly time step for a period of low water running from 9/9/00 to 29/9/00 and a period of increased water level running from 8/12/01 to 14/12/01. Resuspension rates were modified and modelled and observed outputs compared graphically to determine the most appropriate resuspension rate. Additionally, a model run comparing modelled to observed daily averaged data made from fine-scale half-hourly observations in Lake Crescent was made for the period running 9/9/00 to 18/10/00. This was done to assess the ability of DYRESM-CAEDYM to model sediment resuspension dynamics at a coarser time scale that would be more appropriate for running model scenarios of a year or longer.

Once the best estimates of resuspension rates from these analyses had been chosen, the model was run for both lakes over the period April 19<sup>th</sup> 2000 to December 31<sup>st</sup> 2001, and the correspondence between modelled and observed values plotted and analysed using regression. The modelled values were deemed a good fit if this regression did not differ significantly from a 1:1 relationship between observed and modelled values.

#### **4.2.6 Modelling management scenarios with DYRESM - CAEDYM**

After calibrating DYRESM-CAEDYM to simulate sediment entrainment and sedimentation, the model was used to investigate the likely benefits of possible management options such as water level manipulation with the goal of limiting the frequency and magnitude of sediment resuspension events.

From Chapter 3, Section 3.3.2, it appears there exists an optimum water level for both lakes Crescent and Sorell, below which a significant increase in the magnitude and hysteresis in turbidity and SPM results. Modelling runs were undertaken in an attempt to determine the benefits of maintaining water levels at various heights. The heights investigated incorporated the 'optimum' water levels derived in Chapter 3 along with water levels above and below this point (Table 20 and Table 21). The corresponding changes on sediment resuspension dynamics were then summarised.

Table 20 Lake Crescent lake levels chosen for resuspension modelling runs.

Lake Level (mASL) and Label	Definition
802.15 – Actual Conditions	The actual water level of Lake Crescent at the beginning of the time period when modelling commenced
802.7 – Preferred Water Level	The water level identified in Chapter 3 below which point a significant increase in the magnitude and hysteresis in turbidity and SPM results
803.5 – Marsh Level	The water level at which point the marshes surrounding the lake begin to be inundated
803.8 – FSL	Lake Crescent level at full supply

Table 21 Lake Sorell lake levels chosen for resuspension modelling runs.

Lake Level and Label	Definition
802.8 – Actual Conditions	The actual water level of Lake Sorell at the beginning of the time period when modelling commenced
803.2 – Critical Water Level	The water level identified in Chapter 3 below which point values of colloidal turbidity increase significantly
803.5 – Preferred Water Level	The water level identified in Chapter 3 below which point a significant increase in the magnitude and hysteresis in turbidity and SPM results
804.36 – FSL	Lake Sorell lake level at full supply

To investigate the role water levels play in limiting sediment resuspension in both lakes, model runs were carried out with an initial water level or stage height being set to return higher water levels during the run of the model. Each model run employed the same inflow, outflow and meteorological files, but differed in the initial stage height. Lake levels were set so that water depths reached during the course of the runs differed significantly to allow comparisons to be made across an array of lake levels that straddled both the actual conditions, water levels that may correspond to ‘optimum levels’ and levels at which the lakes reached full supply.

## 4.3 Results

### 4.3.1 Critical shear stress

All seven regressions of in-situ measurements of SPM concentration on  $\tau$  estimated from wave theory were highly significant (all  $P < 0.001$ , all  $R^2 > 0.75$ ). An example of one of these regressions is given in Figure 34. The mean  $\tau_c$  ( $\pm 95\%$  confidence interval) was  $0.053 \pm 0.031 \text{ N m}^{-2}$ , which corresponds closely to estimates from elsewhere (Lick 1982; Carper and Bachmann 1984; Arfi, Guiral et al. 1993; Hamilton and Mitchell 1996). For example, Sheng and Lick (1979) found  $\tau_c$  of  $\sim 0.05 \text{ N m}^{-2}$  in Lake Erie as did Arfi et al. (1993) in Lake Ebrič, while Romero et al. (2002) termed this value as the ‘characteristic’  $\tau_c$  when evaluating the likelihood of sediment resuspension in Lake Pamvotis. Further support for a value of  $\tau_c \sim 0.05 \text{ N m}^{-2}$  is provided by Figure 35 where  $0.05 \text{ N m}^{-2}$  closely corresponds to the point at which the



wave base influences the lake bed for both lakes. Consequently, in all subsequent modelling  $\tau_c$  is taken to be  $0.05 \text{ N m}^{-2}$ .

Figure 34 Piecewise linear regression of SPM concentration lagged by 1 h vs shear stress for a resuspension event in Lake Sorell in March 2002. The estimate of  $\tau_c$  for this event ( $\pm 1$  standard error) was  $0.055 (\pm 0.020)$ , and is shown by the vertical broken line on the graph. (Regression statistics:  $F_{(4,11)} = 681.3$ ,  $P < 0.0001$ ,  $R^2 = 0.944$ ).

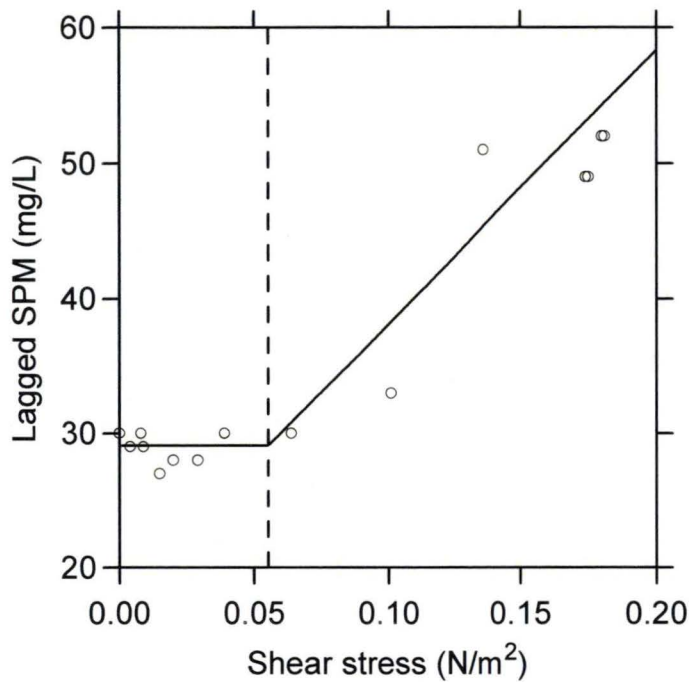
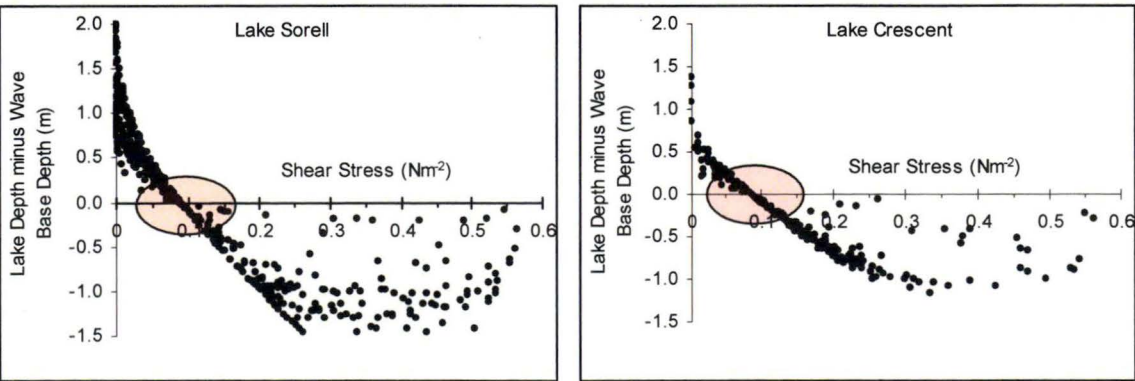


Figure 35 Modelled shear stress for a 20 knot westerly wind versus the wave base depth for 745 points across Lake Sorell and 266 points across Lake Crescent. Where the wave base equals water depth (circled portion),  $\tau_c$  occurs. Points at which the lake depth minus the wave base depth is negative are where the wave mixing depth is greater than the lake depth resulting in resuspension.



The disparity in the frequency and severity of sediment entrainment between the two extremes of water level was considerable. Figure 36 depicts changes in shear stress and SPM concentrations in Lake Sorell for a 2-3 week period in August 2000, at low water levels, and during March 2002, at higher water levels. Figure 37 depicts a similar data set for Lake Crescent, with the periods being September 2000 and December 2001, respectively. Figure 36 and Figure 37 show SPM concentrations rising rapidly and almost simultaneously with increasing shear stress resulting in two to five fold increases in concentration. When shear stress was reduced, concentrations declined sharply which suggests the suspended material had relatively fast settling velocities (Lavelle, Mofjeld et al. 1984).

The frequent occurrence of sediment resuspension events in both lakes Sorell and Crescent and the extreme increases in SPM concentration that these events resulted in is evident with reference to the Figure 36 and Figure 37. Obvious from these figures is the influence higher water levels have on limiting the amount of sediment being resuspended, thus reducing the magnitude of SPM concentrations. The reduced 'background' level of SPM (excluding colloidal solids) present during quiescent periods between the low and high water runs is also evident. For both lakes, the background concentrations of SPM fell from around  $200 \text{ mg L}^{-1}$  to approximately  $30 \text{ mg L}^{-1}$  reflecting a huge drop in the suspended sediment load on the water column, in turn reflecting a considerable reduction in bed erosion occurring at marginally higher water levels.

By comparison, the greatest SPM concentration reached during resuspension events at higher water levels in early 2002 did not reach the magnitude of background concentrations present in both lakes at lower water levels during 2000. This would most likely be caused by a reduction in the area of the lake bed exposed to high energy erosion coupled with longer time periods during which sediments were left undisturbed; this would allow some limited opportunity for consolidation thus reducing the magnitude of entrainment reached during sediment resuspension events.

Figure 36 Lake Sorell resuspension figures of fluxes in SPM versus shear stress; ‘low water level’ (802.88m to 803.08 mASL), and ‘high water level’ (803.65m to 803.47 mASL)

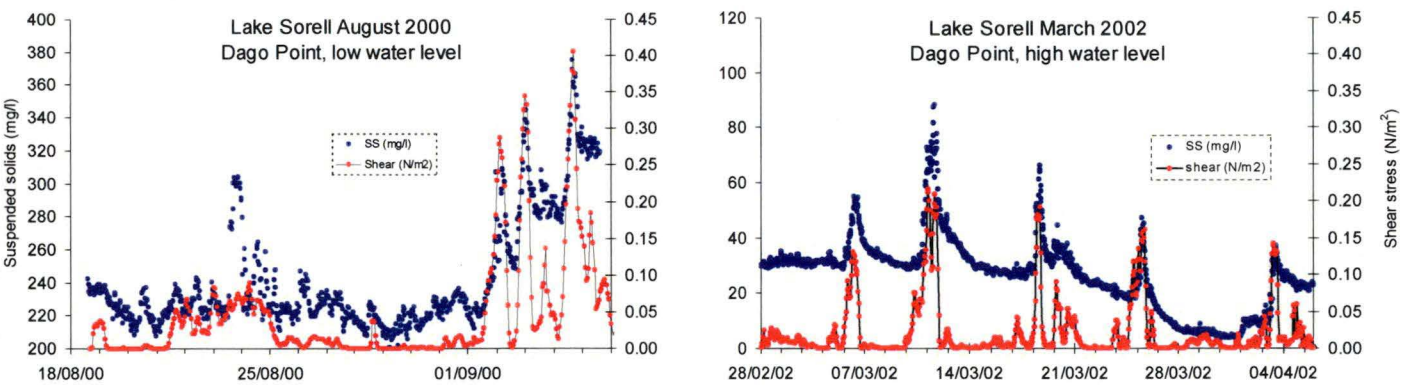
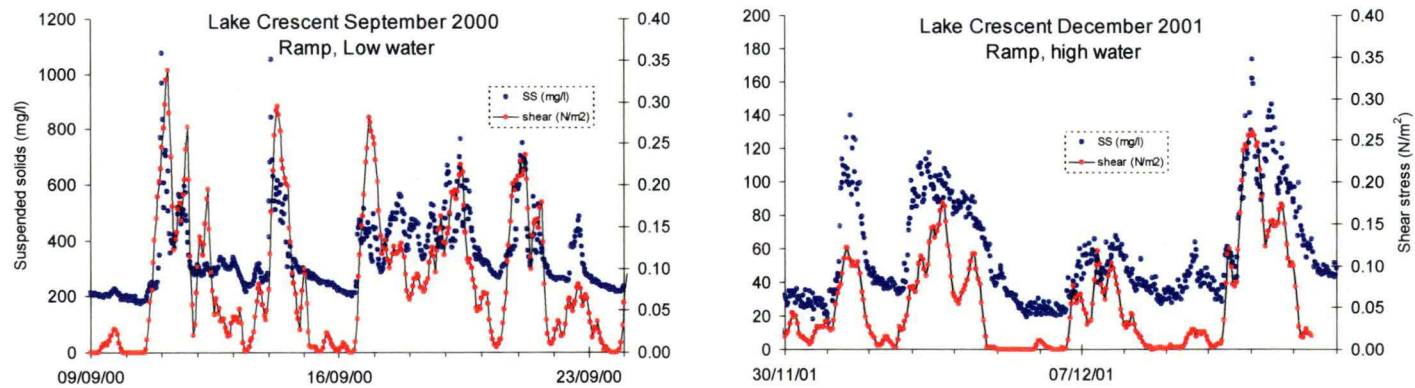


Figure 37 Lake Crescent resuspension figures of fluxes in SPM versus shear stress; ‘low water level’ 802.30m to 802.28 mASL; ‘high water level’ 802.70m to 802.66 mASL



In a number of studies, researchers have found the concentration of suspended solids to increase linearly with increased shear stress above  $\tau_c$  (Ostubo and Muraoka 1988; Hamilton and Mitchell 1996; Bailey and Hamilton 1997). This is similar to the relationships found in Lake Sorell. Table 22 outlines results for regression analysis carried out on seven discrete resuspension events in Lake Sorell in early 2002. The analyses show SPM concentration to be highly correlated with increasing shear stress.

It may be concluded that once shear stress surpasses  $\tau_c$  of  $0.05 \text{ N m}^{-2}$  in lakes Sorell and Crescent, sediment resuspension results. For increasing shear above the threshold, SPM concentrations increase linearly reflecting an increase in the amount of sediment entrained from the lake bed and an increase in the erosion depth of sediments. Increasing shear stress therefore has significant ramifications on water quality variables linked with SPM concentrations.

Table 22 Regression analysis of shear stress versus SPM concentration for seven discrete resuspension events in Lake Sorell between the end of January 2002 and the start of April 2002.

Date	Coefficient	Intercept	$R^2$	$N$	$P$
25/01/2002	173.1	17.6	0.87	79	< 0.001
12/02/2002	201.4	21.4	0.77	43	< 0.001
14/02/2002	201.4	21.4	0.69	36	< 0.001
05/03/2002	158.7	27.0	0.76	21	< 0.001
10/03/2002	269.7	22.0	0.74	31	< 0.001
17/03/2002	167.4	27.7	0.83	20	< 0.001
02/04/2002	179.5	7.9	0.85	30	< 0.001

The effects of varying lake area on the extent of the bed exposed to critical shear stress are depicted in Figure 38 through Figure 41. Figure 38 and Figure 39 are contour plots of shear stress in lakes Crescent and Sorell for several different lake levels for an average 15 knot prevailing westerly wind; Figure 40 and Figure 41 show the influence that increased wind intensity has on shear stress at a fixed lake level of 802.7 mASL in Lake Crescent and 803.5 mASL in Lake Sorell.

Remembering that sediment resuspension increases significantly above a shear of  $0.05 \text{ N m}^{-2}$ , and that the scale of erosion and entrainment is proportional to the magnitude of shear stress, Figure 38 and Figure 39 show clearly that higher water levels markedly reduce the extent and magnitude of sediment resuspension in both lakes. Figure 40 and Figure 41 show that at a lake level that historically returned acceptable levels of water clarity, episodic wind events would still lead to sediment resuspension at high wind speeds.



Figure 38 Shear stress contour plots for Lake Crescent at six different lake levels for a 15 knot (average) westerly wind. Contour intervals as  $\text{N m}^{-2}$ . Purple areas correspond to a shear stress  $< 0.05 \text{ N m}^{-2}$ .

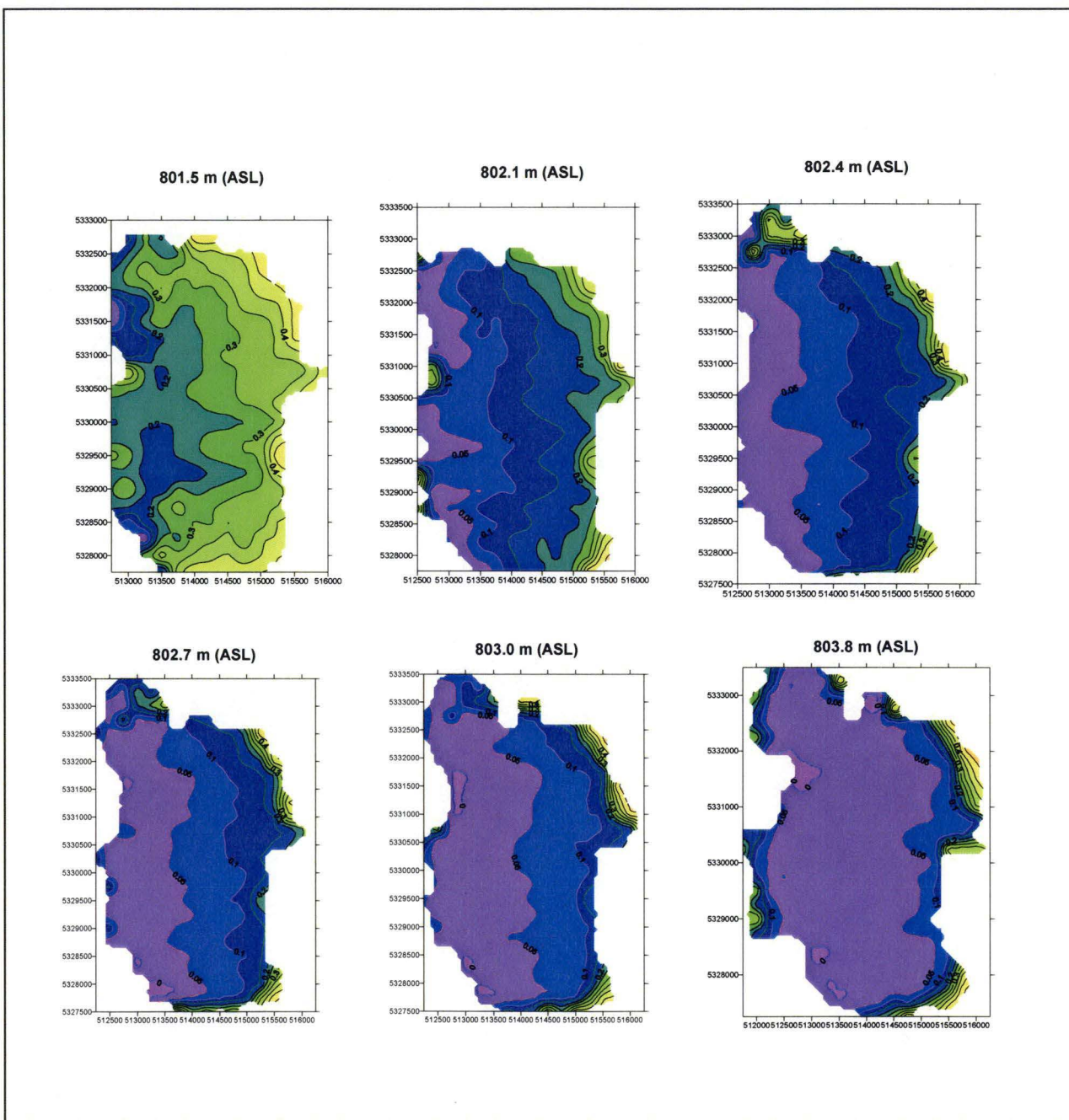


Figure 39 Shear stress contour plots for Lake Sorell at five different lake levels for a 15 knot (average) westerly wind. Contour intervals as  $\text{N m}^{-2}$ . Purple areas correspond to a shear stress  $< 0.05 \text{ N m}^{-2}$

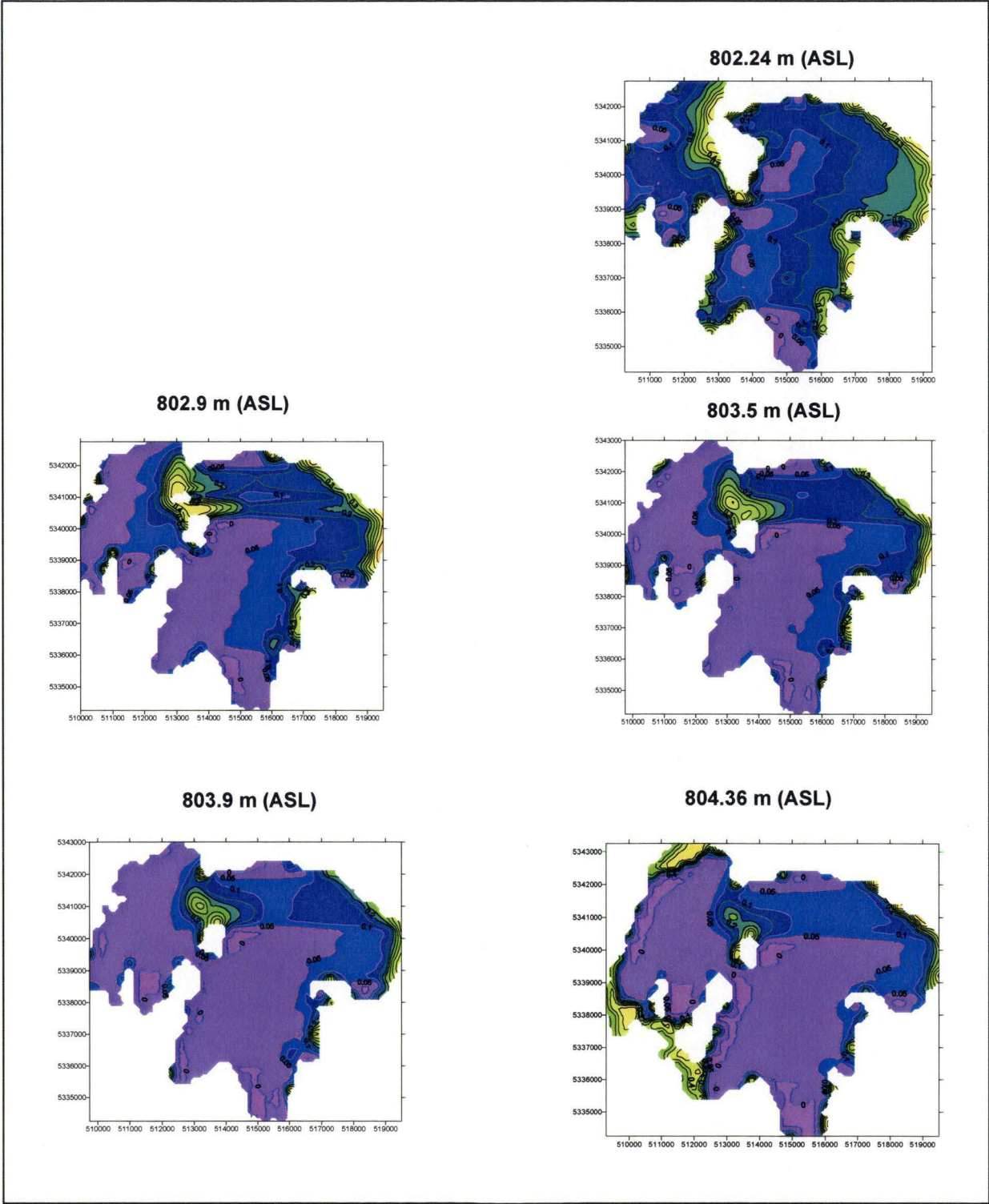




Figure 40 Shear stress contour plots for Lake Crescent at a lake level of 802.7mASL for four different wind speeds. Contour intervals as  $\text{N m}^{-2}$ . Purple areas correspond to a shear stress  $< 0.05 \text{ N m}^{-2}$ .

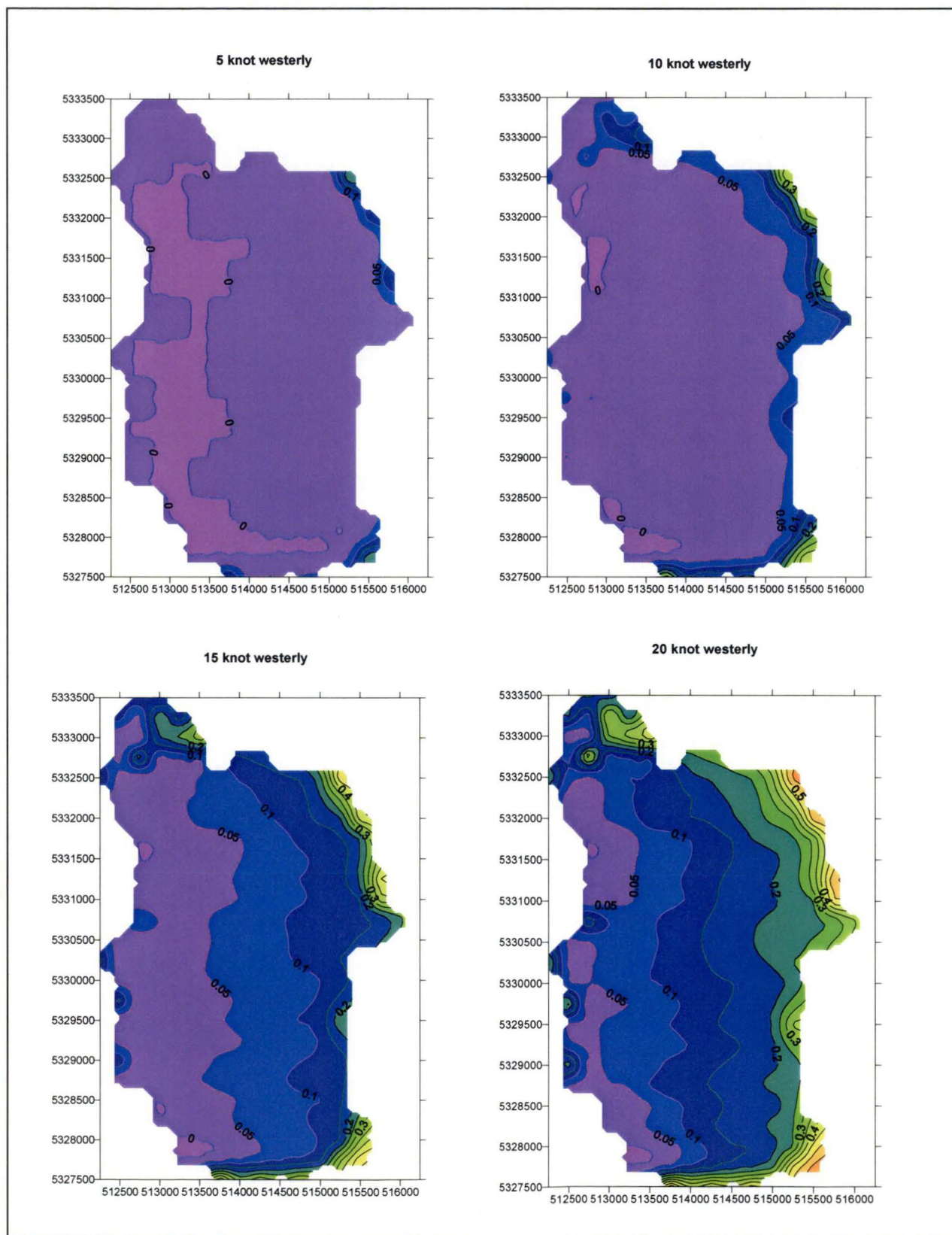
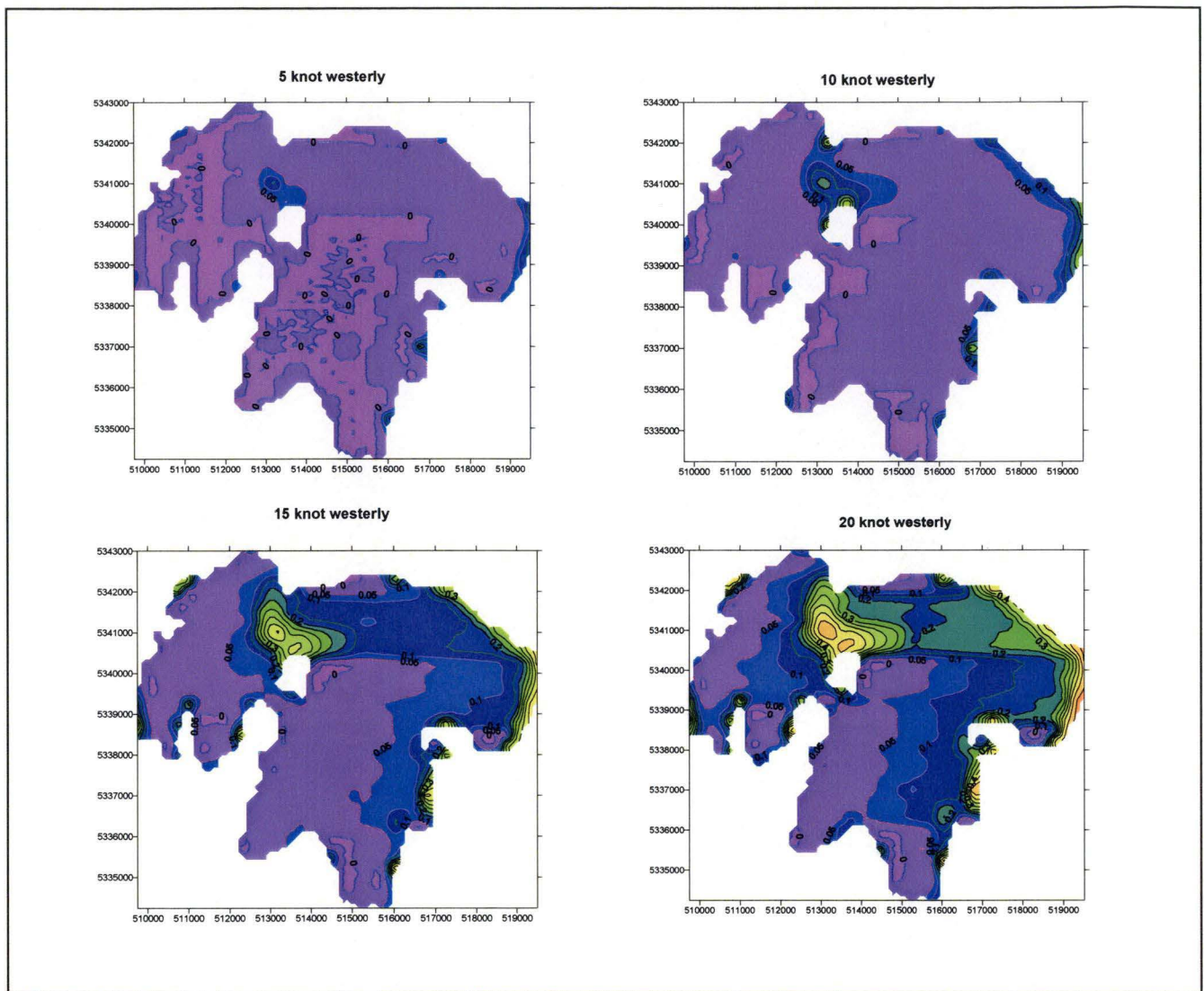


Figure 41 Shear stress contour plots for Lake Sorell at a lake level of 803.5mASL for four different wind speeds. Contour intervals as  $\text{N m}^{-2}$ . Purple areas correspond to a shear stress  $< 0.05 \text{ N m}^{-2}$ .



### 4.3.2 Calibration of DYRESM-CAEDYM

#### 4.3.2.1 Water levels and temperature for DYRESM

Figure 42 to Figure 45 compare modelled estimates of temperature and lake level to measured field data for both lakes Sorell and Crescent. DYRESM predicts changes in lake level and temperature for Lake Crescent and Lake Sorell extremely well (Table 23). This shows that the information used for the meteorological forcing file and the input files of tributary inflow volumes and withdrawals from the lakes are accurate. It may be concluded that the initialisation and running of DYRESM was a success and therefore allows the incorporation of CAEDYM into the modelling framework.



Figure 42 Modelled versus observed lake level change in Lake Crescent, April 2000 to December 2001. DYRESM Lake Level = modelled lake level.

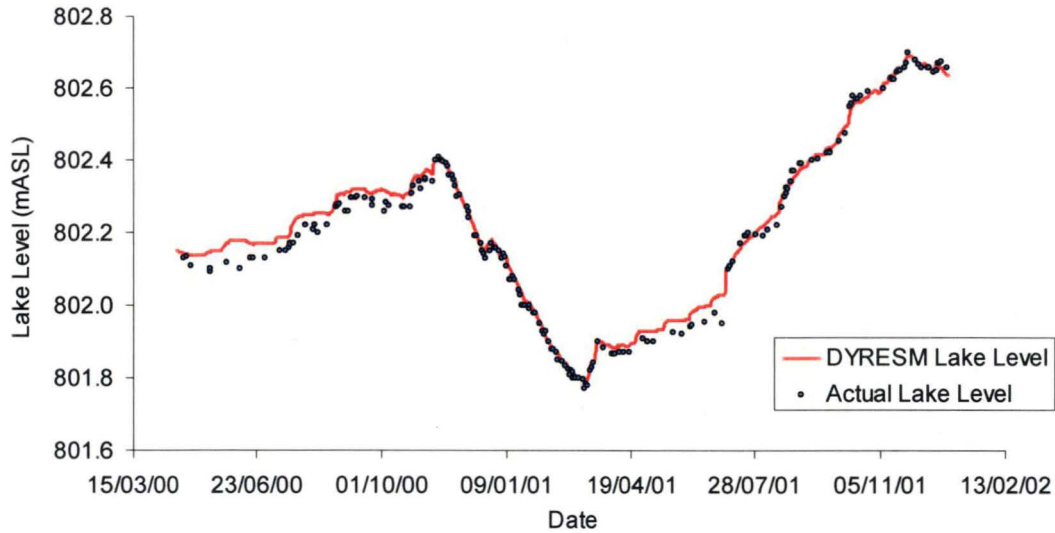


Figure 43 Modelled versus observed temperature changes in Lake Crescent, April 2000 to November 2001. DYRESM Temperature = modelled temperature.

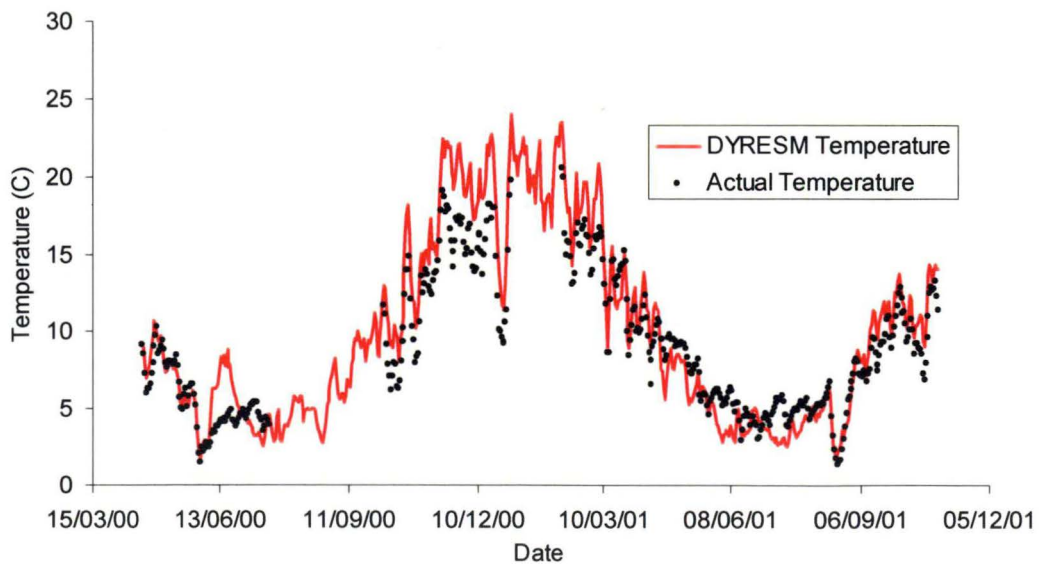


Figure 44 Modelled versus observed lake level changes in Lake Sorell, April 2000 to February 2002. DYRESM Lake Level = modelled lake level.

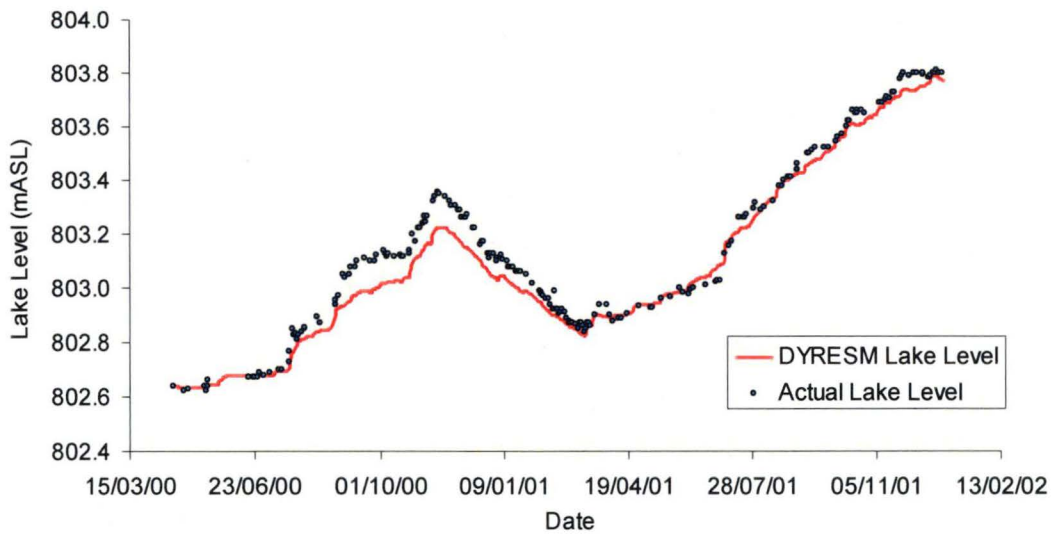


Figure 45 Modelled versus observed temperature changes in Lake Sorell, April 2000 to November 2001. DYRESM Temperature = modelled temperature.

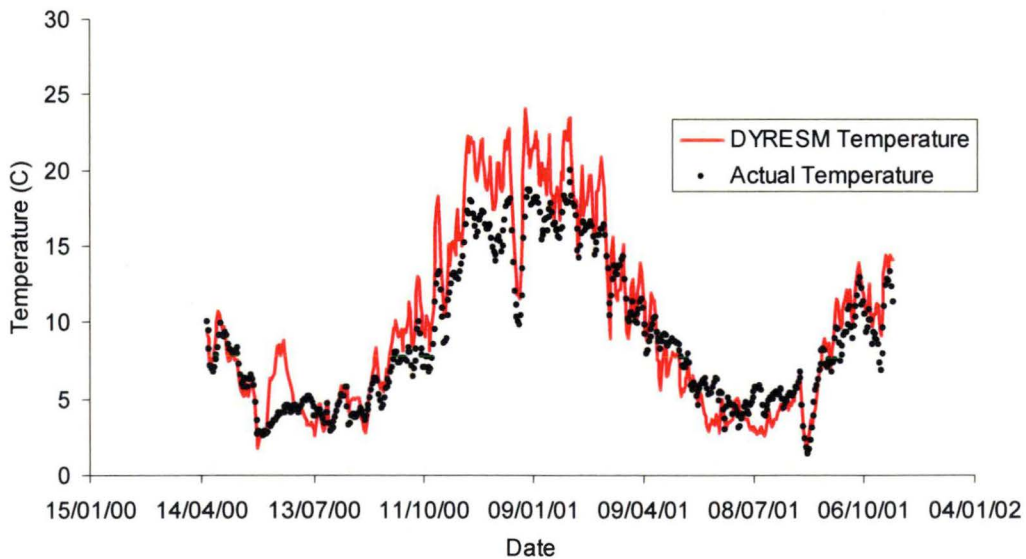


Table 23 Overview of regression analysis of modelled versus observed output for the DYRESM simulations.

	Intercept	Slope	$R^2$	$N$
Lake Crescent Temperature	2.19	0.72	0.93	386
Lake Crescent Lake Level	-8.80	1.01	0.99	197
Lake Sorell Temperature	1.47	0.75	0.93	560
Lake Sorell Lake Level	-2.52	1.00	0.98	216

#### 4.3.2.2 Estimation of particle size

Figure 46 shows the four events used to calibrate particle sizes for the modelling. All showed an abrupt decline in modelled shear stress to fall well below the value of  $\tau_c$ , thus implying that Stokes Law applies. Figure 47, Figure 48 and Figure 49 compare modelled output from CAEDYM to measured concentrations of SPM for the four separate events identified in Figure 46. The combination of the two SPM fractions modelled in CAEDYM allows for a more accurate representation of the behaviour of the resuspended particles as identified by Weyhenmeyer (1998) who states that fast and slow settling particles (coarse and fine) must be considered to adequately describe the flux of sediment in the water column.

Inclusion of a fine fraction was necessary due to the high contribution of colloids to suspended solids in both lakes (refer Chapter 3, Section 3.3.2). A 1  $\mu\text{m}$  particle size was set for this fraction as this corresponds closely to the size limit of material measured in the analysis of suspended solids (refer Chapter 2, Section 2.2.3). Initial modelling runs were made with a 1  $\mu\text{m}$  fraction with the addition of a 5, 6 and 7  $\mu\text{m}$  fraction to represent the coarser material in suspension (Figure 47). Visual inspection revealed an extremely close fit between modelled and measured SPM concentrations for each individual event. The 1  $\mu\text{m}$  and 6  $\mu\text{m}$  combined fractions appeared to best represent the average particle sizes to best approximate the sedimentation rate of the measured field data (Figure 47).

To confirm the inclusion of a 1µm fraction, model runs were undertaken with increased particle sizes for the fine fraction. In all cases, increasing the fine fraction from 1µm to 2µm (Figure 48) and 3µm (Figure 49) failed to adequately simulate the slow settling characteristics of the fine fraction present in the resuspended material.

Figure 46      Passive sedimentation events for Lake Sorell that correspond to events 1, 2, 3 and 4 depicted in Figure 47, Figure 48 and Figure 49.

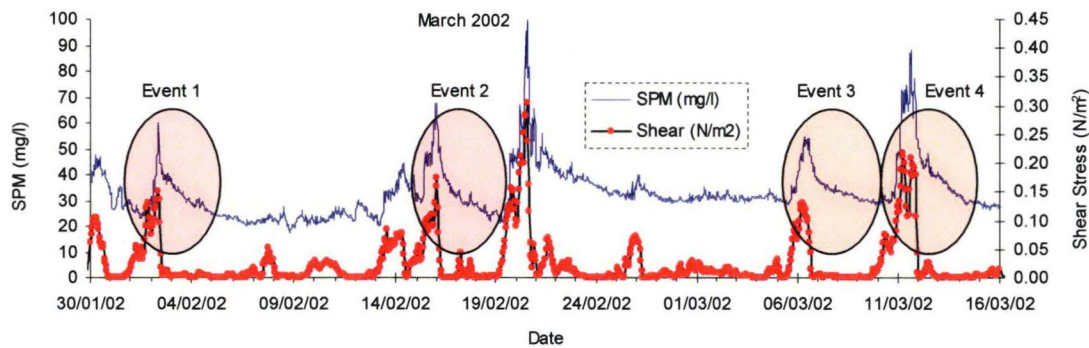


Figure 47      Lake Sorell sedimentation rates for a 1µm particle size component plus a 5, 6 and 7µm component.

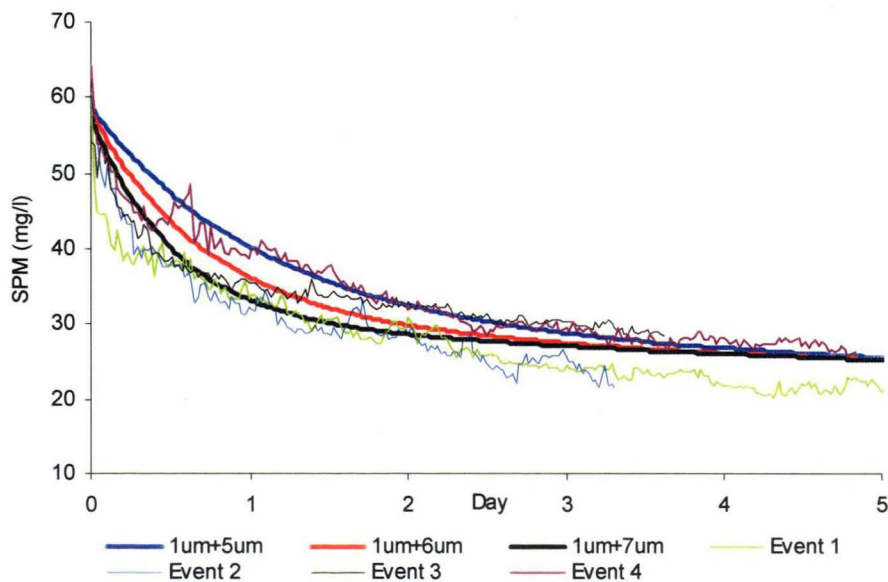


Figure 48 Lake Sorell sedimentation rates for a 2µm particle size component plus a 5, 6 and 7µm component.

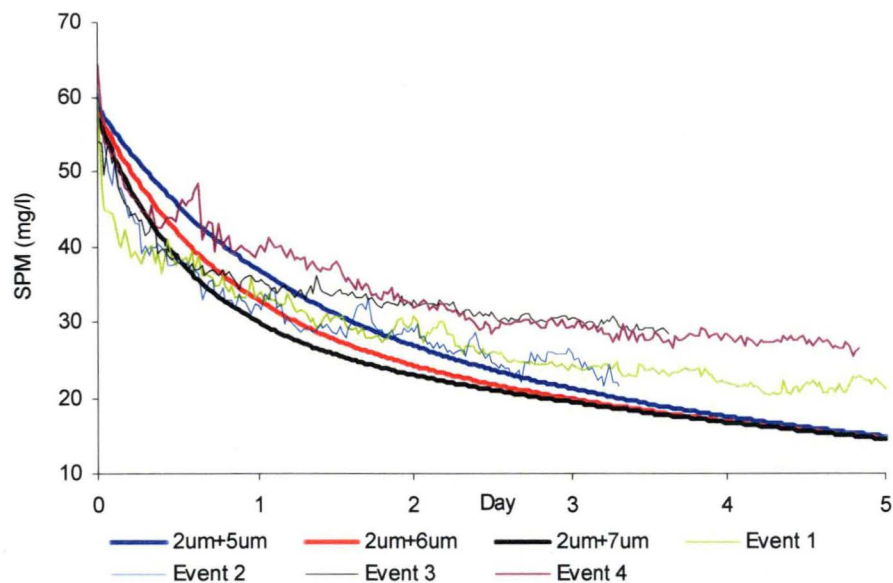
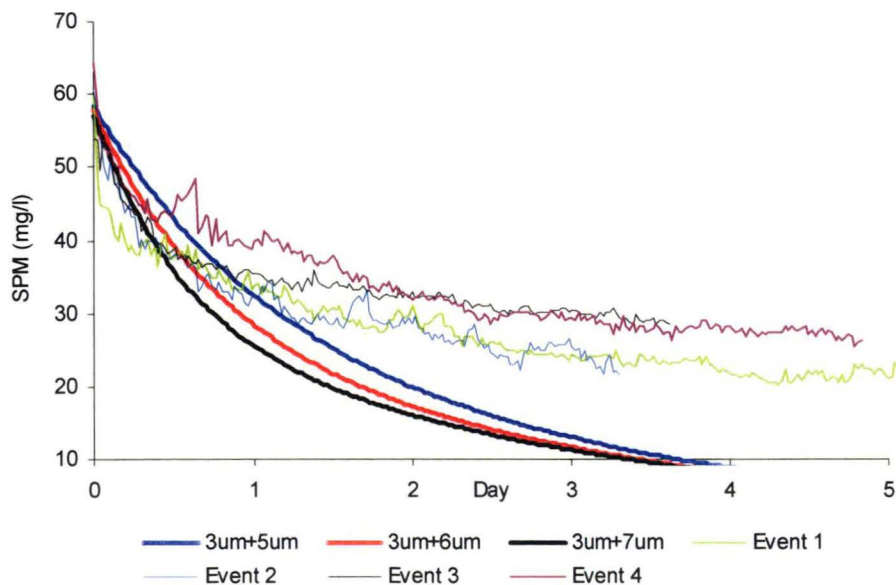


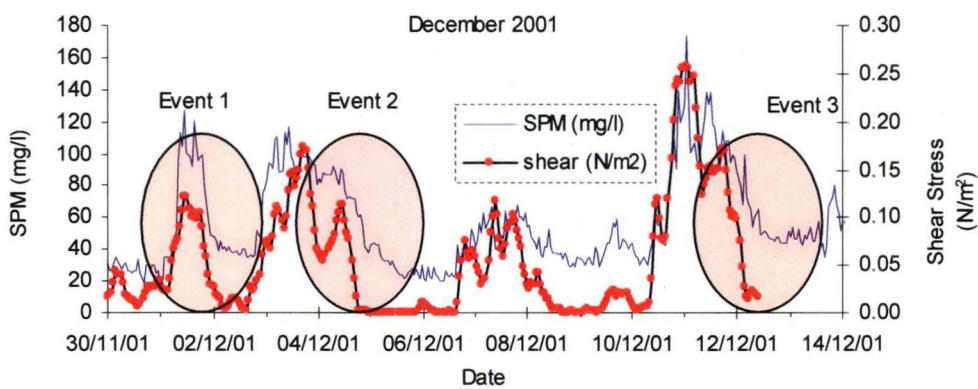
Figure 49 Lake Sorell sedimentation rates for a 3µm particle size component plus a 5, 6 and 7µm component.





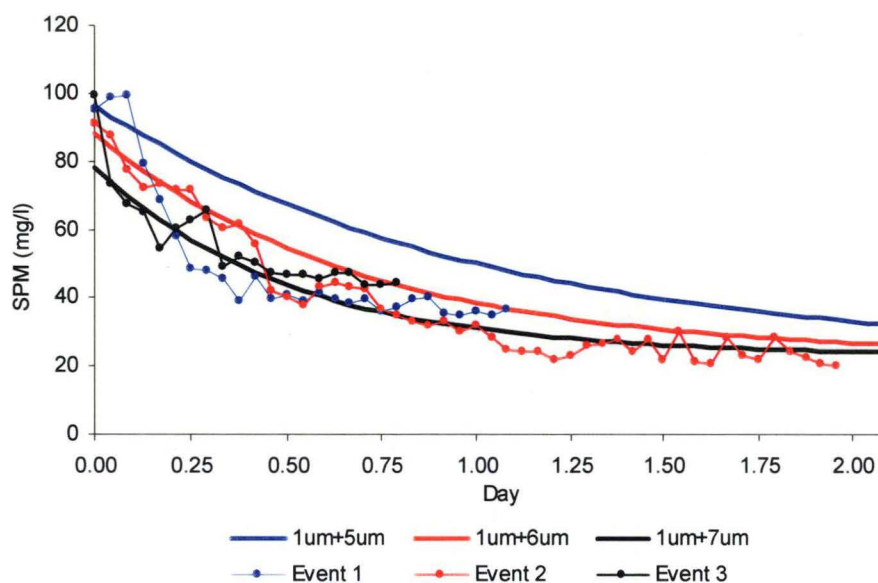
A similar investigation was undertaken with data obtained from Lake Crescent during December 2001. This was done to confirm that settling particle sizes for the two modelled fractions of  $1\mu\text{m}$  and  $6\mu\text{m}$  would adequately represent the settling characteristics of the material in suspension when compared to measured field data. Figure 50 identifies the passive sedimentation events used in the modelling analysis (Figure 51).

Figure 50 Passive sedimentation events for Lake Crescent that correspond to events 1, 2, and 3 depicted in Figure 51.



As was found with Lake Sorell, the inclusion of a  $1\mu\text{m}$  particle component enabled CAEDYM to model the slow settling fraction that is evident in the water column after the initial coarse component has largely settled out. Also, the inclusion of a  $6\mu\text{m}$  component to account for coarse material in suspension appeared to adequately reflect the composition of suspended material as measured in the field.

Figure 51 Lake Crescent sedimentation rates for a 1µm particle size component plus a 5, 6 and 7µm component. Events 1, 2 and 3 correspond to passive sedimentation events labelled in Figure 50.



It was concluded that for both lakes Sorell and Crescent, setting the particle sizes for the two fractions modelled in CAEDYM to 1µm and 6µm respectively would adequately reflect the actual average sizes of material resuspended.

#### 4.3.2.3 Resuspension rate

The active resuspension events are shown in Figure 52, Figure 53 and Figure 54, and the estimates of the resuspension rates derived from the regressions are summarised in Table 24, Table 25 and Table 26. From this work, it was concluded that for extreme events typical of the lakes at low water levels in which the coarse fraction dominates (based on the work on in-situ settling velocities), the resuspension rate would be  $> 0.1 \text{ g m}^{-2}\text{s}^{-1}$ . For the finer fraction, which from the measured rate of settling in the field data appears to dominate, the resuspension rate is  $> 0.005 \text{ g m}^{-2}\text{s}^{-1}$ .

Figure 52 Active resuspension events for Lake Crescent that correspond to events 1, 2, 3 and 4 summarised in Table 24.

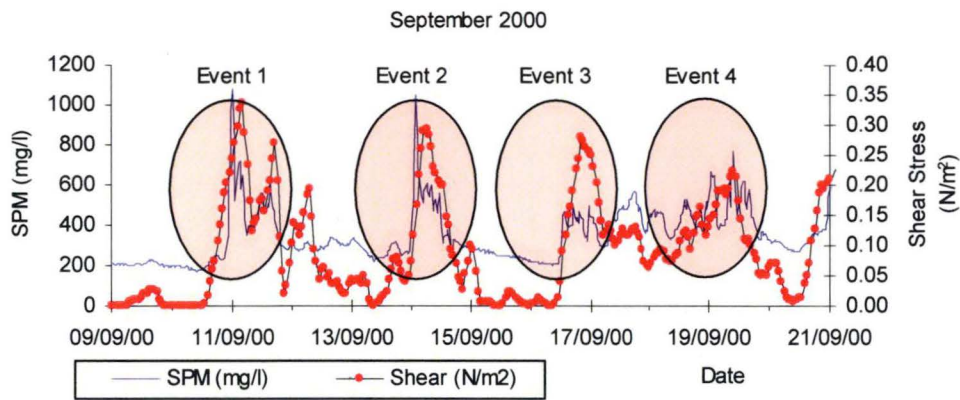


Table 24 Regression analysis of four active resuspension events for Lake Crescent for September 2000 that correspond to events 1, 2, 3 and 4 labelled in Figure 52.

	Resuspension rate (g/m <sup>2</sup> /s)	Intercept	R <sup>2</sup>	n	Date
Event 1	0.109	216.8	0.94	6	10/9/00
Event 2	0.091	269.5	0.86	7	14/9/00
Event 3	0.048	304.5	0.88	5	16/9/00
Event 4	0.038	565.1	0.83	7	19/9/00

Figure 53 Active resuspension events for Lake Crescent that correspond to events 1, 2, 3 and 4 summarised in Table 25.

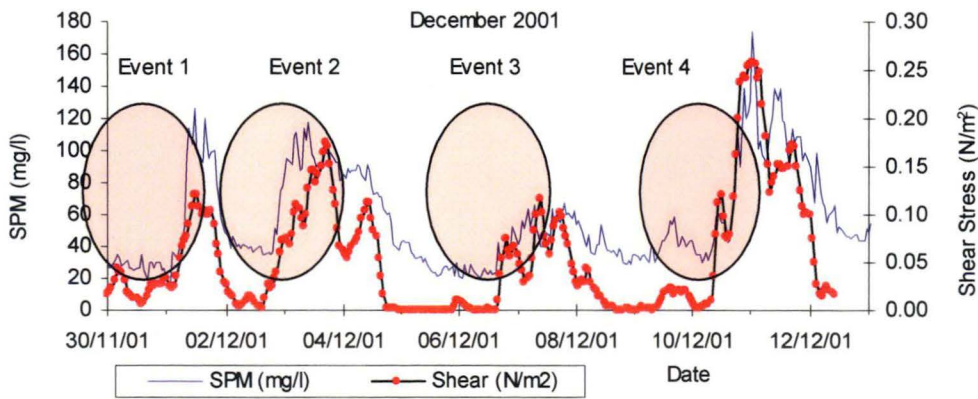




Table 25      Regression analysis of four active resuspension events for Lake Crescent for December 2001 that correspond to events 1, 2, 3 and 4 labelled in Figure 53.

	Resuspension rate (g/m <sup>2</sup> /s)	Intercept	R <sup>2</sup>	n	Date
Event 1	0.0033	58.99	0.71	13	
Event 2	0.0030	42.78	0.94	14	
Event 3	0.0012	60.27	0.62	9	
Event 4	0.0016	51.47	0.96	9	

Figure 54      Active resuspension events for Lake Sorell that correspond to events 1, 2, 3, 4 and 5 summarised in Table 26.

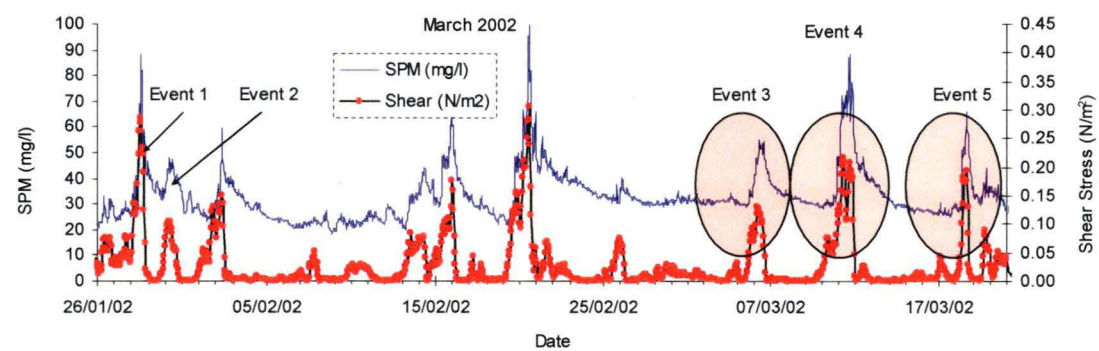


Table 26      Regression analysis of five active resuspension events for Lake Sorell for January 2002 to March 2002 that correspond to events 1, 2, 3, 4 and 5 labelled in Figure 54.

	Resuspension rate (g/m <sup>2</sup> /s)	Intercept	R <sup>2</sup>	n	Date
Event 1	0.0047	75.40	0.87	16	28/1/02
Event 2	0.0009	84.81	0.85	23	29/1/02
Event 3	0.0015	90.43	0.92	14	5/3/02
Event 4	0.0029	93.77	0.92	19	10/3/02
Event 5	0.0025	100.88	0.76	16	18/3/02

Sub-daily runs of half-hourly time steps were initially made with particle sizes of 1 µm and 6 µm and resuspension rates of 0.005 and 0.1 g m<sup>-2</sup>s<sup>-1</sup>, followed by numerous runs incrementally changing these values until the best visual fit between modelled and observed values were obtained (Figure 55 and Figure 56). From this, it

was concluded that a resuspension rate of  $0.01 \text{ g m}^{-2}\text{s}^{-1}$  for the  $1\mu\text{m}$  component and  $0.25 \text{ g m}^{-2}\text{s}^{-1}$  for the  $6\mu\text{m}$  component best approximated the actual resuspension rates of this material in the lakes at these timescales, and these estimates were used for the detailed comparison of daily DYRESM-CAEDYM output to daily average measurements for Lake Crescent between the 8/9/00 and the 18/10/00. The resuspension rates used for the daily runs based on model calibration from the sub-daily runs produced the best fit with observed data (Figure 57). It was therefore concluded that a resuspension rate of  $0.01 \text{ g m}^{-2}\text{s}^{-1}$  for the  $1\mu\text{m}$  component and  $0.25 \text{ g m}^{-2}\text{s}^{-1}$  for the  $6\mu\text{m}$  component best represented the actual resuspension rates of this material in the lakes. The relationship between modelled and observed total suspended solids concentration for this period in Lake Crescent is very strong and close to 1:1 (Figure 57).

Figure 55 Comparison of half-hourly measured SPM concentration (mg/L) and modelled SPM concentration (mg/L) for the  $1\mu\text{m}$ ,  $6\mu\text{m}$  and combined  $1\mu\text{m}$  and  $6\mu\text{m}$  (total) fractions for Lake Crescent for September 2000. DYCAD SPM = DYRESM-CAEDYM modelled SPM.

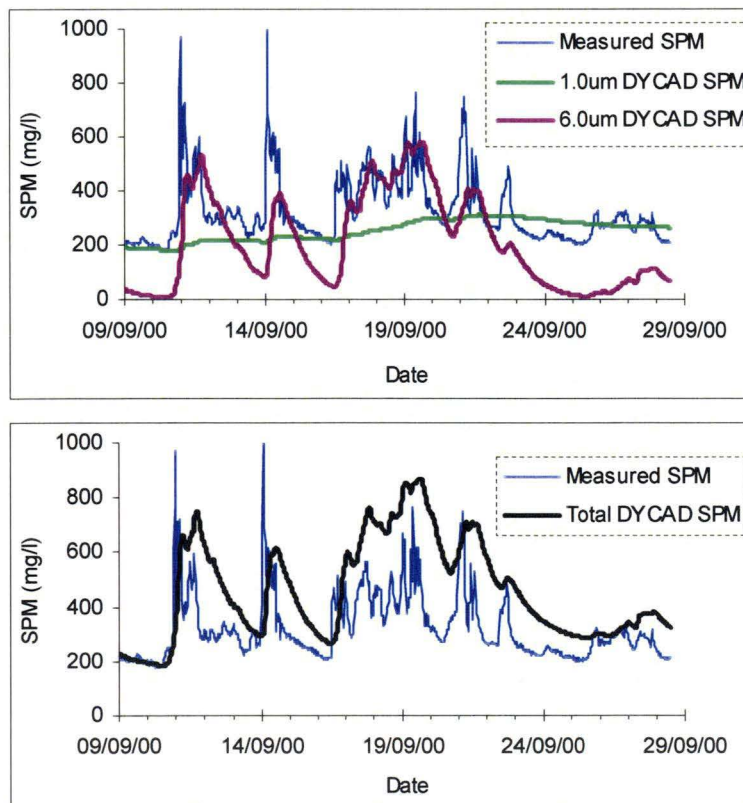


Figure 56 Comparison of half-hourly measured SPM concentration (mg/L) and modelled SPM concentration (mg/L) for the 1µm, 6µm and combined 1µm and 6µm (total) fractions for Lake Crescent for December 2001. DYCAD SPM = DYRESM-CAEDYM modelled SPM.

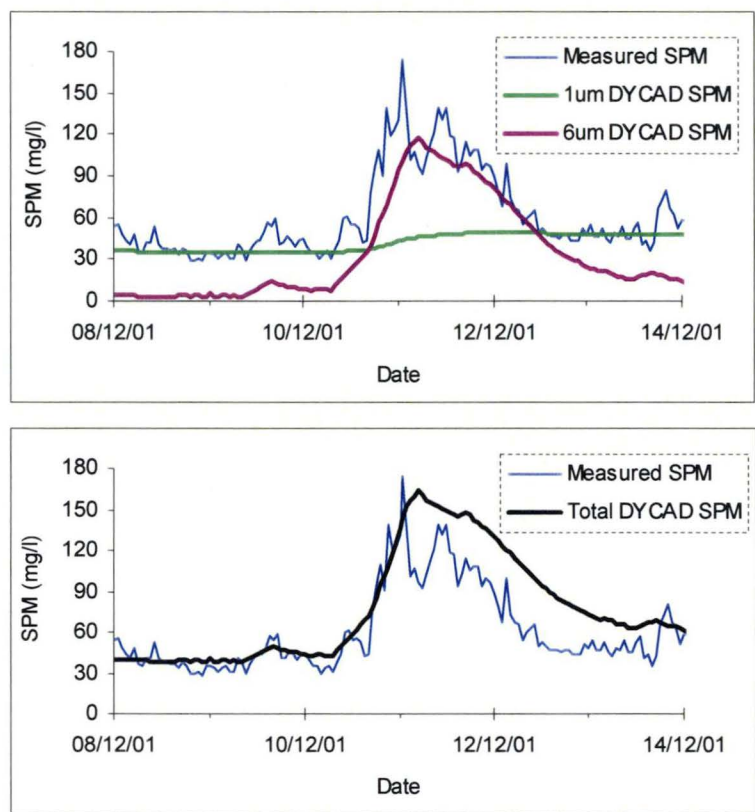
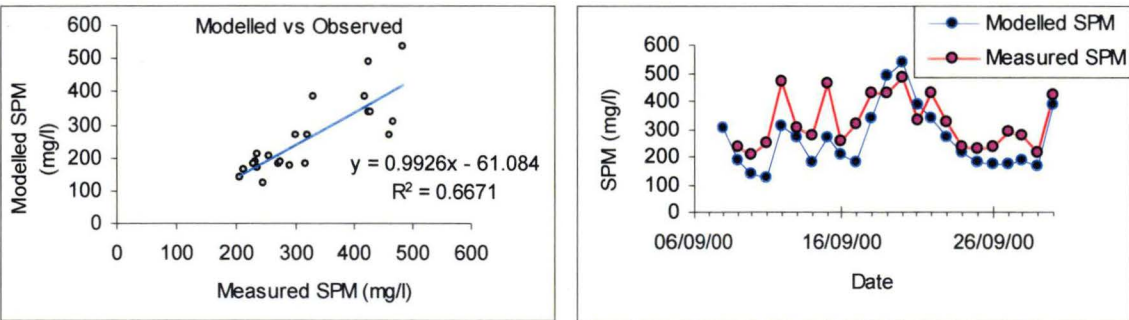


Figure 57 Comparisons of modelled and daily measured SPM concentration (mg/L) for Lake Crescent, September 2000 to October 2000.



Using these estimates, the modelled concentrations from April 19<sup>th</sup> 2000 to December 31<sup>st</sup> 2001 are depicted in Figure 58 and Figure 59. For both Lake Crescent and Lake Sorell, regression analysis of measured versus modelled total suspended solids yielded relationships with slopes not significantly different from 1 (Table 27), although Lake Sorell is less well fitted by the modelled values than Lake Crescent, with modelled values tending to be higher than observed values (Figure 60). Inspection of the time series plot for Lake Sorell shows that modelled values tended to be higher than measured values in the autumn and early winter months of 2001 as the lake level started to rise from its summer minimum (Figure 59). These results are encouraging, and suggest that DYRESM-CAEDYM returns sufficiently reliable results using the values chosen to initialise the model. The final values used in subsequent modelling are summarised in Table 28.

Table 27 Summary statistics for the regressions of observed SPM concentrations vs. modelled concentrations for lakes Crescent and Sorell. The slopes ( $\pm 1$  standard error, SE),  $P$ -value for the test of the difference of slope from 1, coefficient of determination,  $r^2$ , and sample size,  $n$ , are given.

Lake	slope ( $\pm$ SE)	$P$	$r^2$	$n$
Crescent	0.967 ( $\pm 0.139$ )	0.816	0.609	33
Sorell	0.814 ( $\pm 0.110$ )	0.062	0.731	32

Table 28 Values of constants for DYRESM-CAEDYM modelling of sediment resuspension in lakes Sorell and Crescent.

Constant	Particle Size	Value
Particle Density ( $\text{kg m}^{-3}$ )	1 $\mu\text{m}$	2650
	6 $\mu\text{m}$	2650
Critical Shear Stress ( $\text{N m}^{-2}$ )	1 $\mu\text{m}$	0.05
	6 $\mu\text{m}$	0.05
Resuspension Rate ( $\text{g m}^{-2}\text{s}^{-1}$ )	1 $\mu\text{m}$	0.01
	6 $\mu\text{m}$	0.25

It must be realised that DYRESM-CAEDYM models daily 'average' conditions and the meteorological data used in the simulations are 'average' climatic values over the time step in question. In the case of resuspension dynamics, both lakes Crescent and Sorell exhibit considerable changes in daily suspended solids concentration (Figure 52 and Figure 53). The field data used for comparisons of modelled and observed suspended solids consist of a lake-wide average of 8 samples taken over a short time period (1-2 h) for each date and, therefore, does not resolve sub-daily fluctuations in suspended solids concentration.

Figure 58 Plot of modelled SPM concentration vs date (grey solid line) for Lake Crescent, with measured values of SPM plotted with symbols. The dotted black line is the lake level in m above sea level.

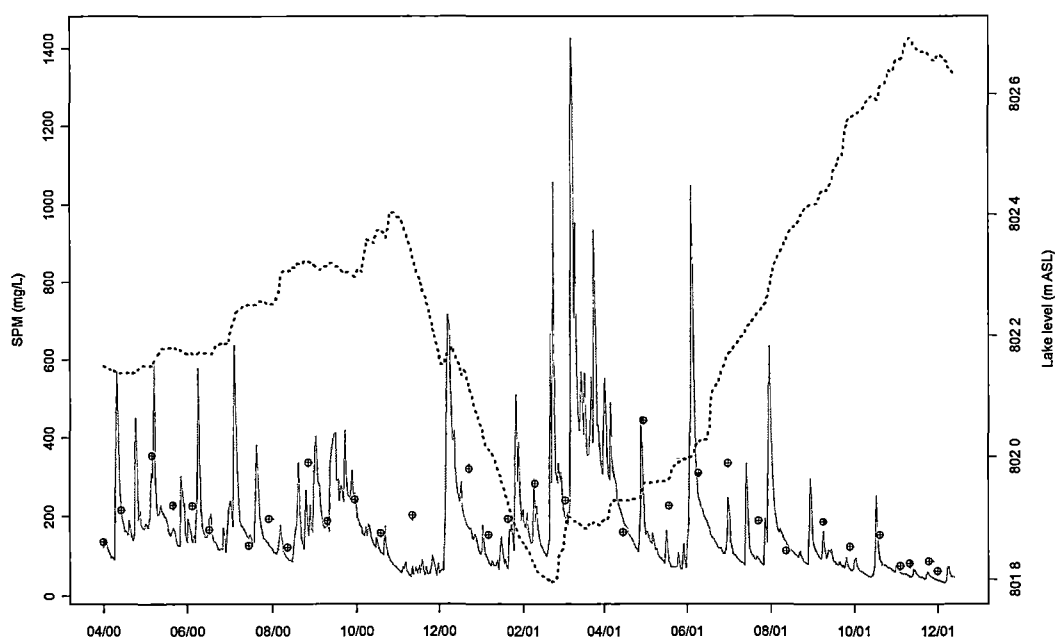


Figure 59 Plot of modelled SPM concentration vs date (grey solid line) for Lake Sorell, with measured values of SPM plotted with symbols. The dotted black line is the lake level in m above sea level.

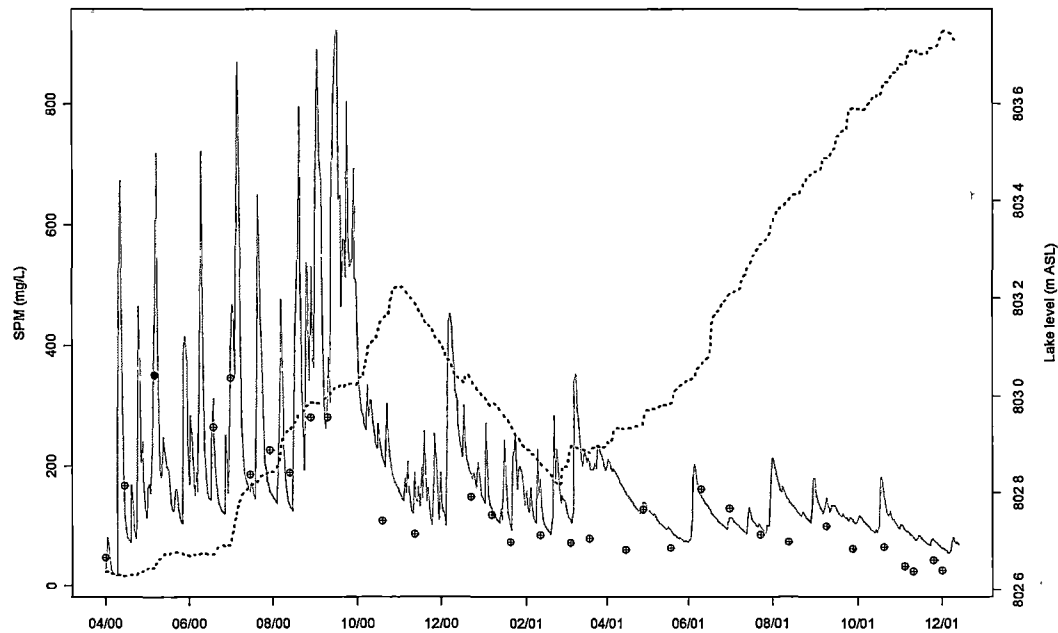
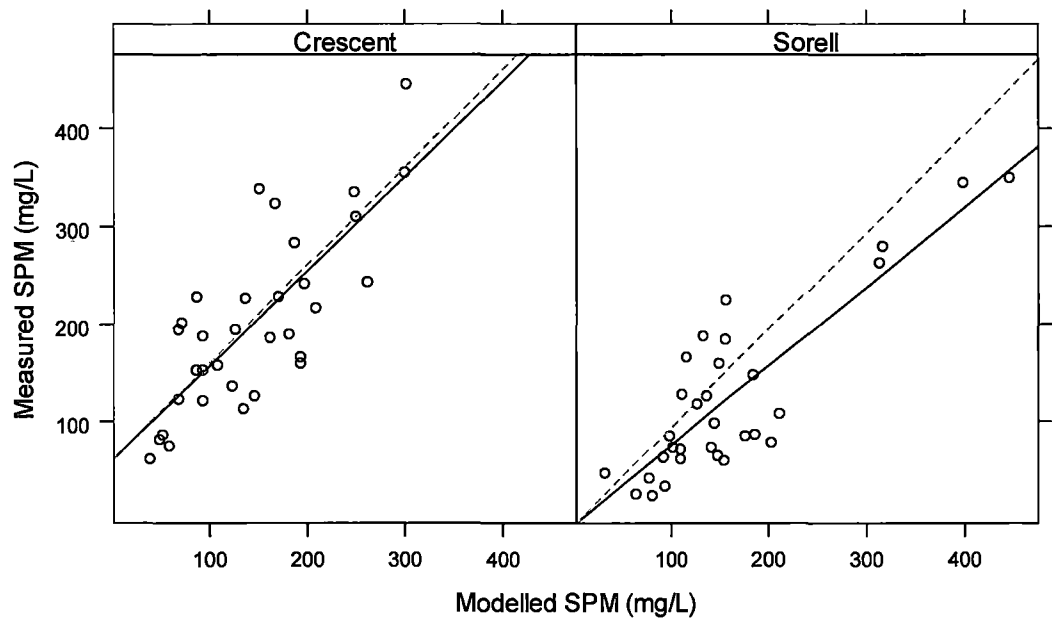


Figure 60 Relationship between measured and modelled SPM concentrations for lakes Crescent and Sorell. Solid line is the regression fit, broken grey line is the 1:1 relationship.



### **4.3.3 Modelling management scenarios**

Figure 61 and Figure 62 detail modelled output of the 1 $\mu$ m fraction, the 6 $\mu$ m fraction and the total SPM concentration for the different lake levels in lakes Crescent and Sorell. Each modelling run was for the period running from the 19<sup>th</sup> of April 2000 to the 31<sup>st</sup> of December 2001. The result is a series of model runs that use the same inflow/outflow and meteorological forcing data as the original data file but comparisons can be made with the predicted benefits of subtle changes in lake level. The results illustrate clearly the potential for limiting sediment resuspension by maintaining the lakes at higher levels.

Interestingly, with reference to Figure 61 and Figure 62, at the higher water levels approaching full supply, a major reduction in the 1 $\mu$ m fine colloidal fraction results in both lakes. Limiting the amount of colloidal material being resuspended should be a high priority as the fine colloidal fraction, due to its ability to stay in suspension for extended periods of time (Chapter 3, Section 3.3.2), would maintain high levels of turbidity even during periods of calm weather when the coarse fraction has settled out. In all cases, significant reductions in suspended solids and consequent improvements in water clarity and light attenuation result from increasing lake level.



Figure 61 Overview of the effect of manipulating lake level on sediment resuspension in Lake Crescent.

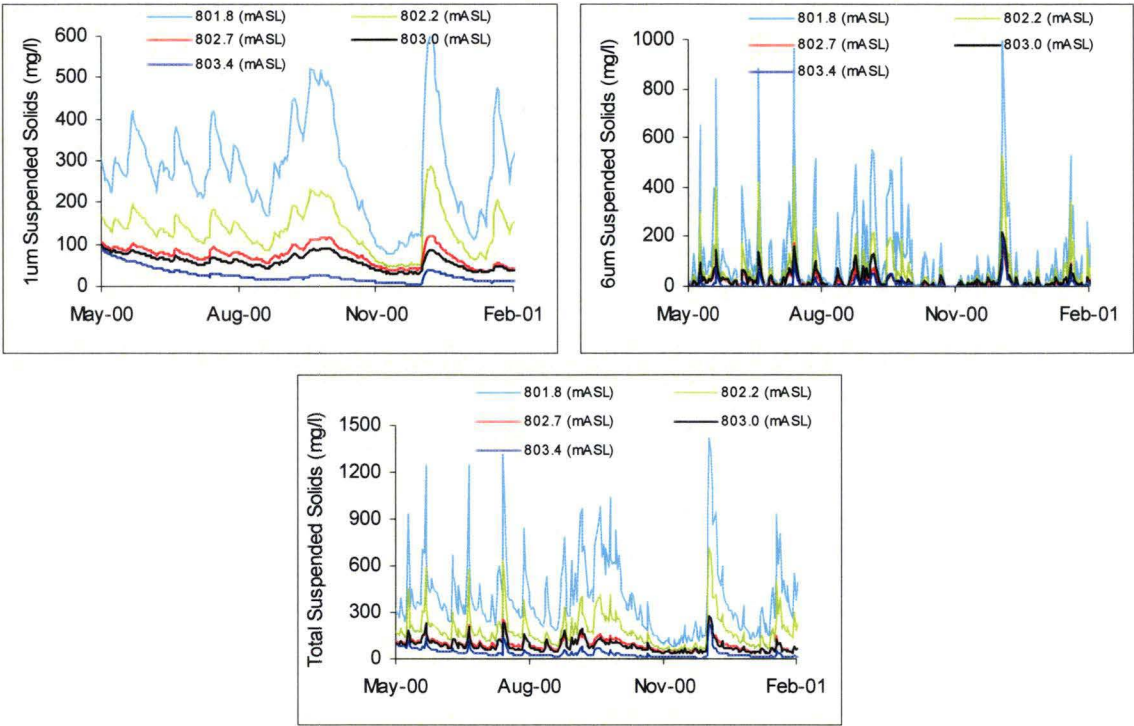
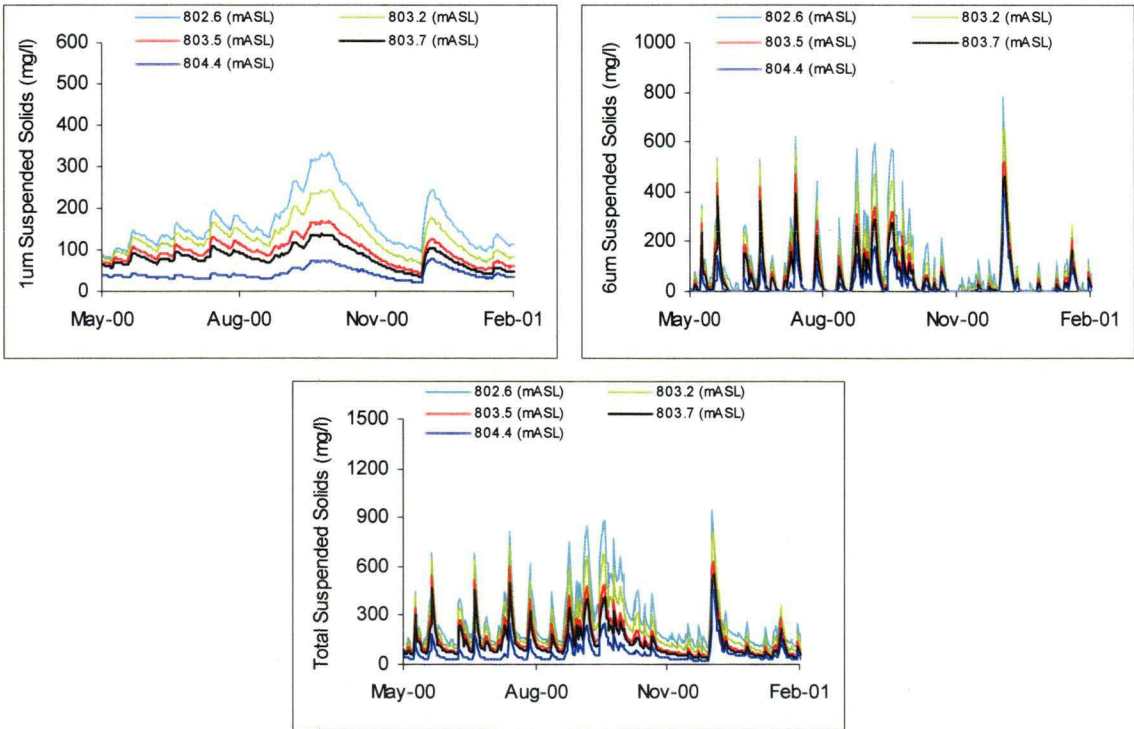


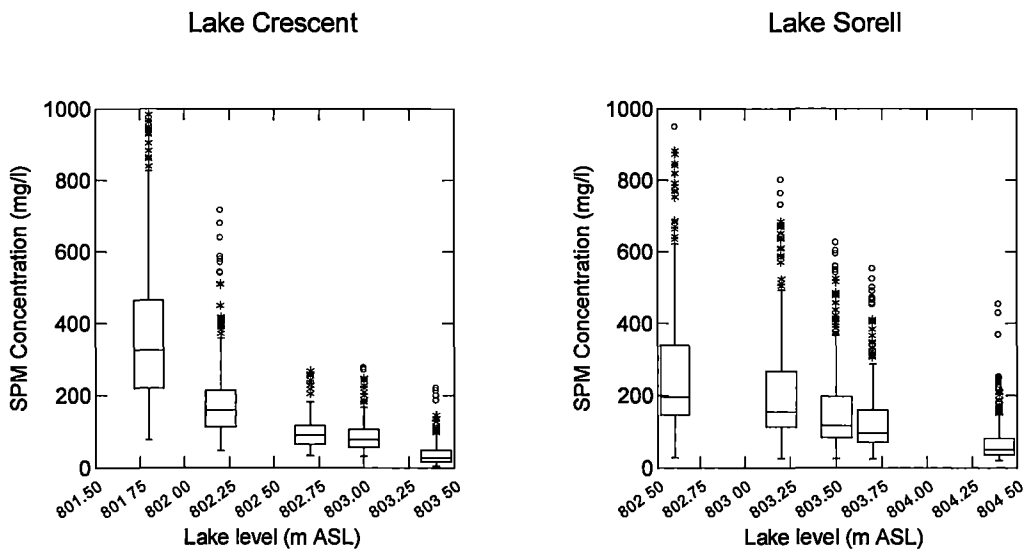
Figure 62 Overview of the effect of manipulating lake level on sediment resuspension in Lake Sorell.





DYRESM-CAEDYM predicts a considerable increase in the severity of sediment resuspension in both lakes at low water levels that were typical of those reached during 2000 and 2001 (Figure 63). This reflects a massive increase in SPM entering the water column due to increased sediment entrainment and an increased erosion depth of sediments, which, in turn, exposes material that has long remained undisturbed, possibly increasing the release of stored colloids (Chapter 3, Section 3.3.3). Lake level management appears to be a promising means to reduce sediment resuspension in both lakes.

Figure 63 Box-plot summary of the effect of manipulating lake level on SPM concentration in both lakes Crescent and Sorell. Values between the inner and outer fences are plotted with asterisks. Values beyond the outer fences, called far outside values, are plotted with empty circles.



## 4.4 Discussion

### 4.4.1 Shear stress and changes in lake area

The value of  $\tau_c$  estimated from field data accords with that found in several other studies. Moreover, the value of  $0.05 \text{ N m}^{-2}$  was further supported in the calibration runs for CAEDYM by the close agreement with observed and modelled data. It was also clear from modelling the changes in the area of lake bed exposed to shear stresses

greater than  $\tau_c$  that changes in the water levels in these lakes have a strong potential to affect sediment resuspension and the light climate in these lakes.

The pattern of distribution of coarse and fine sediments in these lakes further supports the results of these initial calculations. Continual sedimentation and resuspension leads to a sorting of material in lakes (Hakanson and Jansson 1983). The sediment in exposed shallow areas is coarser, because lateral transport causes the fine material to concentrate in deeper sheltered areas where resuspension rarely occurs (Douglas Evans 1994). Carper and Bachmann (1984) imply that the areas that reflect high energy 'erosional areas' are not an important source of suspended solids. It is the areas where resuspension rarely occurs that become important when winds or lake levels are such that resuspension of this material occurs (Scheffer 1998). In the case of Lake Crescent and Lake Sorell, the areas that experience high shear stress depicted in Figure 38 and Figure 39 at higher lake levels are areas of well-armoured reef or sandy shore. At lower water levels, shear stress increases across areas of the lake basin where resuspension rarely occurs. These areas have high levels of fine-grained 'depositional' sediment. It is resuspension of this material that has resulted in the extreme increase in suspended particulate and colloidal material in both lakes (Chapter 3, Section 3.3.2 and 3.3.3).

To limit sediment erosion and improve water clarity it is necessary to reduce the level of shear stress exerted on the lake bed, and manipulation of lake levels is one potentially effective management option.

#### **4.4.2 Model calibration**

The choices of small and large particle sizes is critical since the settling velocities of sediment particles determine the length of time that particles remain suspended in the water column (Hamilton and Mitchell 1996). Additionally, the fine settling fraction represented by the  $1\mu\text{m}$  SPM component in CAEDYM, contributed significantly to nutrient fluxes (Appendix 9) and SPM concentration (Appendix 4), as well as greatly increasing light attenuation in the water column (Chapter 3, Section 3.3.2). Hence, in lakes Crescent and Sorell, where resuspension processes are clearly important in controlling turbidity, light attenuation and nutrient fluxes, the ability to distinguish between fine and coarse particulate fractions and their settling characteristics is

advantageous in a modelling scenario, improving the representation of how these fractions behave in the lake environment along with providing valuable information as to how each individual fraction behaves under differing management scenarios.

The 'in-situ' method of calibration used here was assumed to be more representative of natural conditions than measurements made under quiescent conditions in the laboratory, as in-situ calibration takes into account the naturally turbulent conditions under which particles settle in a lake (Bailey and Hamilton 1997). The final values of 6  $\mu\text{m}$  and 1  $\mu\text{m}$  appeared to be the best choices to represent the 'average' large and small particle size fractions being resuspended in these lakes.

The final resuspension rates were within the range of values found by Vlag (1992) of  $0.01 - 0.4 \text{ g m}^{-2}\text{s}^{-1}$ . The values also straddled the resuspension rate estimates made by Bailey and Hamilton (1997) of  $0.023$  to  $0.035 \text{ g m}^{-2}\text{s}^{-1}$  but were higher than estimates made by Murphy (2001) of  $2 \times 10^{-6}$  to  $4.5 \times 10^{-5} \text{ g m}^{-2}\text{s}^{-1}$  when investigating sediment resuspension in Lake Mokoan, a large shallow lake in Victoria. This may be due to the sediments of Lake Mokoan being more dispersive and resuspendable when compared to the sediments of lakes Crescent and Sorell and the lakes studied by Vlag (1992) and Bailey and Hamilton (1997).

In the case of Lake Sorell, modelled SPM concentration was generally higher than measured values. This may be due to the substantial contribution colloids make to SPM concentration in Lake Sorell (Chapter 2, Table 11; Chapter 3, Section 3.3.2; Appendix 4). Importantly, measured SPM often underestimates this material due to fine colloidal particles failing to be retained on the surface of the filter paper during the gravimetric determination of SPM. This may explain why modelled estimates of SPM are slightly higher than measured values. It is possible that DYRESM-CAEDYM gives a closer approximation of suspended solids concentration by including this colloidal material in Lake Sorell because it includes a significant colloidal fraction in its modelled estimate.

Alternatively, the overestimate of DYRESM - CAEDYM in modelling SPM values in Lake Sorell may be due to the models failure to take account for changes in bottom

type with significant areas of Lake Sorell consisting of well armoured rocky reef and sandy shore that would provide limited material for entrainment compared to areas dominated by fine and coarse silts. DYRESM - CAEDYM assumes the lake bed to be of uniform composition. Calibration of the models took place during a time of reduced water level and from visual observation of Lake Sorell at low water levels, it was observed that an increase in water level would increase the areas of sandy shore and rocky reef inundated and likely lead to changes in the overall relative significance of these areas in changing average sediment resuspension dynamics within the lake.

A number of additional factors must be taken into account when comparing modelled to measured SPM. Most importantly, DYRESM-CAEDYM estimates horizontally averaged SPM concentration as determined from average meteorological conditions, while actual measured conditions are likely to be more variable. Additionally, several other factors that are not considered in the modelling framework of DYRESM-CAEDYM may contribute to variation in observed versus modelled output. These include lake circulation currents and changes in sedimentation rate due to changes in average particle size through flocculation and deflocculation (Hamilton and Mitchell 1996); variations in bottom conditions across the lake that influence entrainment such as the presence and density of aquatic macrophytes (James, Barko et al. 2004); the degree of consolidation (Bengtsson, Hellstrom et al. 1990; Vlag 1992); the general composition of sediments such as organic content and mineralogy (Sheng and Lick 1979); and the activity of benthic organisms (Lick 1982).

All these factors may differ spatially across the lake bed and give rise to variations in observed SPM levels under comparable levels of shear stress whereas DYRESM-CAEDYM, and in fact the majority of sediment resuspension models, assume these variables to be relatively constant within a system. The significance and importance of variations between observed and modelled data depend largely on the aimed application of the models and the interpretation of the results. For the current application of DYRESM-CAEDYM in investigating long term, lake averaged changes in sediment resuspension dynamics, such factors would be less relevant and it may be concluded that in light of these factors, changes in SPM concentration through resuspension and sedimentation as modelled by DYRESM-CAEDYM fit very well to observed values of SPM.

Due to the complex and heterogenic nature of sediment resuspension as it occurs in a natural ecosystem (Blom, Van Duin et al. 1992), applying a one-dimensional model has its limitations (Murphy 2001). The bathymetric data utilised by DYRESM is a significant simplification of the bathymetry in three dimensions and fails to account for variations in wind fetch as influenced by basin shape and local topography. Particularly in the case of Lake Sorell, basin shape would influence wind fetch around the lake considerably (Chapter 1, Figure 1 and Figure 2; Chapter 2, Section 2.2.2), with Lake Sorell possessing two distinct basins. The application of DYRESM-CAEDYM to model sediment resuspension in Lake Sorell may be further improved by modelling the two basins independently. This problem is less pronounced in Lake Crescent with a lower shoreline development than Lake Sorell (Chapter 2, Table 2). This is particularly pronounced at low water levels when the marsh area of the lakes are exposed (Chapter 2, Figure 2). Therefore winds of varying direction will differ in the impact on sediment resuspension and transport and would be more pronounced in a system with a highly variable morphology. This may further help explain the improved fit of modelled and observed data in Lake Crescent when compared to Lake Sorell.

Investigating or applying models of greater complexity would be warranted if more detailed or finer scale processes of sediment resuspension dynamics were to be investigated. In such a scenario, applying a model capable of resolving sediment resuspension and transport in three dimensions would be valuable (Bailey and Hamilton 1997) and provide insight into horizontal gradients and patchiness in sediment resuspension dynamics throughout the lakes. Such an application would allow conclusions to be drawn as to the significance of changing bottom type on influencing resuspension dynamics in lakes Crescent and Sorell.

#### ***4.4.3 Modelling management scenarios***

From work carried out to date, a highly significant relationship has been found between decreasing water levels in lakes Crescent and Sorell and increasing water column sediment loading. In general, the available options for reducing sediment resuspension and SPM concentrations are limited: water level manipulation, wind fetch manipulation or, in the case of very small water bodies, chemical flocculation.

Due to the size of both these lakes, the only feasible option is, therefore water level management.

It must be remembered that in the case of both lakes, DYRESM-CAEDYM was calibrated during a time of frequent and severe disturbance. The sediments present in the lakes would likely have never been eroded to a similar extent previously. This, coupled with an increase in the frequency of disturbance driven by the extreme low lake levels, would result in largely unconsolidated sediments which would, in turn, increase the ease with which sediments are entrained (Blom, Van Duin et al. 1992) as the duration between entrainment events is directly proportional to the critical shear velocity (Scheffer 1998). Model predictions would represent a 'worst' case scenario in regards to sediment resuspension at higher water levels. The modelled estimates made at higher water levels would fail to take into account the process of sediment consolidation, a process that would be significant at higher levels due to a reduction in the total erosion area of the lakes (Hakanson 1981), that would lead to increased consolidation as sediments would remain undisturbed for increased periods of time. Increased consolidation would in turn reduce the pore water content and increase the cohesiveness of the sediments (Vlag 1992) thus requiring increased disturbance to entrain sediments and reduce the severity of sediment resuspension.

Rerunning the calibration process at a time of increased and sustained high water levels would be beneficial to determine if sediment resuspension dynamics change in lakes Crescent and Sorell over an extended time period. Nonetheless, from the investigative modelling runs carried out thus far, substantial improvements in water clarity can be expected if both lakes were maintained at relatively high levels. If the lakes were held close to full supply, a five to eight fold improvement in SPM levels would likely be achieved. However, with the lakes maintained at levels comparable to the average lake levels of the past, sediment resuspension, although significantly reduced from the extreme levels observed recently, would still influence the lakes on a regular basis.

Due to the important role sediment resuspension plays in driving ecological processes in lakes Sorell and Crescent, the successful application of DYRESM-CAEDYM to model sediment dynamics has provided the platform for further levels of complexity

to be added to the modelling framework. This will allow the influence of sediment resuspension on primary productivity to be assessed through changes in nutrients and light attenuation, as previous work (Chapter 2, Section 2.3.3) has shown sediment entrainment to influence trophic status as indicated by increases in TN, TP and chlorophyll *-a*.

Having identified lake level management as a key to improving the current state of water clarity through a reduction in SPM loading, the next step would be to model the possible behaviour of both lakes if water clarity was to improve, as both systems have lost their long-established “equilibrium stability” of the past. Improvements in water clarity that would result from a reduced frequency and severity of sediment resuspension could possibly provide ideal conditions for phytoplankton dominance through increased light availability coupled with increased availability of TN and TP (Scheffer 1998). Further understanding the interaction between the different trophic levels of the lakes and determining the dominant ecosystem processes would increase our understanding of the way the lakes may behave when water clarity improves, along with providing further options for possible management strategies aimed at manipulating specific aspects of the ecosystem and helping to promote a clear-water macrophyte dominated state.

The successful application of DYRESM-CAEDYM to model sediment resuspension provides a good foundation for investigating more complex ecosystem processes such as trophic interactions and the importance sediment resuspension plays in influencing primary productivity through changes in light availability and nutrient dynamics. This is particularly relevant to lakes Crescent and Sorell in which sediment resuspension has proven to be a critical process in influencing the ecology of the lakes.

## **Chapter 5 The limnological paradox of Lakes Sorell and Crescent revisited. Stable state theory and the current situation: the switch from clear to turbid water.**

### **5.1 Introduction**

#### **5.1.1 Background**

The contrasting trophic characteristics typical of lakes Sorell and Crescent prior to the degradation in water quality in the late 1990s were dramatic. Based on Cheng and Tyler's work in the early 1970's (Cheng and Tyler 1973a; Cheng and Tyler 1973b; Cheng and Tyler 1976a; Cheng and Tyler 1976b) and unpublished data held by the Inland Fisheries Service, Lake Sorell was a clear, macrophyte dominated lake, while Lake Crescent was turbid and dominated by phytoplankton. Some of the major contrasts between the lakes are summarized in Table 29. Because these striking differences persisted in these adjacent lakes with almost identical catchments in terms of geology, vegetation and land-use, Cheng and Tyler dubbed these differences a "limnological paradox".

More recently, differences between clear-water and turbid water conditions in shallow lakes have been attributed to lakes "flipping" between alternative stable states (Scheffer, Bakema et al. 1993; Jeppesen, Jensen et al. 1999), and the kinds of trophic differences that are apparent between Lakes Crescent and Sorell appear consistent with trophic cascade models of lake systems (e.g. Carpenter, Kitchell et al. 1985). If such mechanisms do prevail in these lakes, remediation of the current poor water quality may depend on understanding and managing higher order trophic interactions in the lakes to manipulate the food web and limit phytoplankton productivity (Carpenter, Kitchell et al. 1985), which, in turn, would promote the desired clear-water, macrophyte dominated state (Scheffer, Hosper et al. 1993; Sondergaard, Jeppesen et al. 2000).

Sediment resuspension has been identified as the most important driver influencing water clarity and light attenuation at low lake levels in lakes Sorell and Crescent (Chapters 2, 3 and 4), but the importance of various processes at higher water levels is little understood. The return of high water levels should reduce the importance of



sediment resuspension in structuring the ecology of both lakes Sorell and Crescent (Chapter 3 and Chapter 4) and result in a physical environment more typical of that seen prior to 1998 where the lakes appeared to be in a reasonably stable equilibrium state that had been sustained, in large, for several decades. Lake Sorell characteristically had relatively clear water and maintained a significant cover of littoral aquatic macrophytes; Lake Crescent by comparison appeared ‘turbid’ or ‘cloudy’ due to a higher algal standing crop (Cheng and Tyler 1973a; Cheng and Tyler 1976b; Cutler, Kinrade et al. 1990).

The purpose of this chapter is to present a conceptual model for the functioning of these lakes, and evaluate the historical data and those collected for this study to evaluate which top-down or bottom-up processes are the strongest candidates for explaining Cheng and Tyler’s “limnological paradox”. Understanding the basis for the observed contrasting behaviours of the two lakes is important in understanding the significance of both bottom-up and top-down processes in defining the characteristic state of each lake.

### ***5.1.2 Conceptual model***

Figure 64 outlines the prominent abiotic and biotic pathways likely to be typical of a shallow lake ecosystem of the Central Plateau, Tasmania and is similar to other conceptual models outlining the general ecological pathways characteristic of a shallow lake ecosystem (Scheffer 1998). On the left side of the diagram are the abiotic variables, such as sediment resuspension, light attenuation and internal and external loading and nutrient cycling. Each represents ‘bottom-up’ processes influencing primary productivity. On the right are the biotic variables influencing primary productivity. Figure 65 is a simplification of the biotic processes summarised in Figure 64 and forms the basis for the major dominant trophic interactions focussed upon later in this chapter.



Figure 65 abstracts and summarises the trophic relationships between the four major biotic components in these two lakes. Introduced brown trout are the top-level piscivore preying on the golden galaxias (*Galaxias auratus*) (Stuart-Smith, Richardson et al. 2004), which is the dominant zooplanktivore. It preys heavily on large zooplankton (Hardie 2003), such as *Daphnia*, which may limit phytoplankton via grazing (Scheffer 1998), as it is the larger cladocerans, such as *Daphnia*, that typically have the greatest impact on phytoplankton productivity (Carpenter, Kitchell et al. 1987) because grazing rates and filtering rates approximately scale to the square of body length (Jackson 2003).

Lakes Sorell and Crescent share similar species assemblages at all trophic levels but exhibit significant contrasts in the density and dominance of species between lakes (Table 29). This pattern could be consistent with a trophic cascade operating in Lake Sorell, where the greater abundance of piscivorous (and introduced) trout depress the abundance of zooplanktivorous golden galaxiids sufficiently to allow large-bodied grazers such as *Daphnia* to limit phytoplankton standing stocks. To date, no research has investigated the significance and extent of the trophic relationships that exist in either lake, and it is possible that the “limnological paradox”, as identified and termed by Cheng and Tyler (1973a), is explicable by contrasting trophic interactions in the two lakes.

Table 29 Summary of the trophic characteristics of lakes Sorell and Crescent prior to 1997. These are based on data and descriptions in Cheng and Tyler (1973a) and from quantitative and qualitative unpublished records held by the Inland Fisheries Service of Tasmania.

Lake	Sorell	Crescent
Average Turbidity	< 15 NTU	> 25 NTU
Percentage macrophyte cover	> 40%	< 10%
Average Chlorophyll <i>a</i>	< 10 µg/L	> 20 µg/L
Invertebrate composition	Frequent <i>Daphnia</i>	Rotifers, copepods, small <i>Daphnia</i>
Zooplanktivore composition	Low numbers of <i>Galaxias auratus</i>	Very high numbers of <i>Galaxias auratus</i>
Piscivore composition	High biomass of <i>Salmo trutta</i>	Low biomass of <i>Salmo trutta</i>

While the broad pattern of differences between Lake Sorell and Lake Crescent seem consistent with a top-down trophic cascade operating in Sorell, it remains unclear why Crescent, which also has trout, remained dominated by phytoplankton. Moreover, conformity of a lake to a trophic pattern predicted by an hypothesised mechanism does not prove the existence of that mechanism, and further information needed to be assembled to appraise the potentially competing explanations of how these lakes function.

To further understand the potential role for higher trophic interactions to influence the dynamics of these lakes, I sought further, more intensively collected survey data on key features of the flora and fauna that have been deemed important “markers” of trophic cascades in shallow lakes. Thus, to understand the degree to which trophic interactions influence primary productivity, information on zooplankton community characteristics, particularly cladoceran species composition, size and biomass allows the potential for phytoplankton limitation by zooplankton grazing to be assessed (Scheffer 1990). Similarly, the size and species composition of the zooplankton community reflect the importance of higher order consumers on influencing zooplankton community characteristics as selective feeding by zooplanktivorous fish on large-bodied zooplankton may result in a zooplankton community dominated by small individuals (Vanni 1986; Northcote 1988; Modenutti, Balseiro et al. 1993; Donald, Vinebrooke et al. 2001). Small zooplankton have less impact on phytoplankton as grazing rate decreases with decreasing body size (Brooks and Dodson 1965; Hall, Threlkeld et al. 1976; Vanni 1987), therefore the loss of large-bodied zooplankton through selective feeding by zooplanktivorous fish may lead to an increase in algal biomass (Jeppesen, Jensen et al. 1990).

Similarly, detailed information on phytoplankton species succession may indicate the influence of zooplankton grazing on limiting phytoplankton productivity and structuring the phytoplankton community as zooplankton select for both size and type of food (McCauley and Briand 1979; Vanni 1986; McQueen and Post 1988).

To assess the potential significance of trophic interactions on limiting primary productivity in lakes Sorell and Crescent, detailed quantitative data was collected on the phytoplankton and zooplankton communities of both lakes between April 2000

and August 2002. Additional to this, detailed information was collated from parallel studies undertaken on the golden galaxias and brown and rainbow trout populations of the lakes. Qualitative data and information that dated back to November 1996 were also analysed and assessed, and provided a significant and extended data set for investigating and characterising zooplankton community dynamics for a period preceding the degradation in water quality that occurred in 1998. The goal was to evaluate the potential of trophic interactions to influence primary productivity in either Lake Sorell or Crescent.

## **5.2 Methods**

### ***5.2.1 Phytoplankton community composition***

#### **5.2.1.1 Sample collection, identification and enumeration**

Phytoplankton samples consisted of 25 mL of homogenised water taken from a depth integrated water sample from 4 separate sites spread across both lakes Sorell and Crescent. Phytoplankton samples were taken during routine water quality sampling trips (Chapter 2) and made at two to three week intervals between April 2000 and August 2002. A total of 42 separate dates during this period were sampled in Lake Sorell and 41 separate dates sampled in Lake Crescent.

The samples from each site for each sampling occasion were pooled giving a total sample volume of approximately 100 mL. The pooling of these samples was justified on the basis of yielding a lake-wide “average” community composition as there was little interest in characterising site-level, within-lake variations in phytoplankton density. The changes in community composition that typify trophic cascades are usually large, and such pooling has proven adequate in other studies (Jeppesen, Jensen et al. 1999). The combined sample was preserved using approximately 1-2 mL of glutaraldehyde per 100 mL of sample (A.P.H.A. 1992). This preservation method was preferred over the use of buffered formalin (Imojen Pearce, School of Plant Science, University of Tasmania, Pers. comm.) after comparison of duplicate samples preserved in a 4% solution of buffered formalin and samples preserved with glutaraldehyde.

Phytoplankton enumeration and identification was carried out according to Standard Methods (A.P.H.A. 1992) by Imojen Pearce, School of Plant Science, University of Tasmania. Each sample was inverted a number of times to ensure homogeneity. 1mL was then removed and examined under a Zeiss Axiovert 25 light microscope (Carl Zeiss AG, Jenna, Thuringia, Germany). Very dense samples were diluted to assist counting. Samples were examined at 40 × and 100 × magnification. Species identification and cell number were recorded. Total volume of the samples was measured and recorded to determine total counts.

Algal biomass was estimated from chlorophyll-*a* concentrations assuming a dry-weight/chlorophyll ratio of 70 (Scheffer 1998). Chlorophyll analysis was carried out by acetone extraction according to Standard Methods (A.P.H.A. 1992) on water samples collected during routine water quality trips (Chapter 2, Section 2.2.3).

#### **5.2.1.2 Analysis of community composition**

To assess whether the community composition of the phytoplankton had changed relative to the earlier studies of Cheng and Tyler (1973) and Cutler et al. (1990), the data from all studies were converted to presence/absence since there was no easy way of ensuring comparable quantitative data owing to different levels of detail in quantifying algae in these earlier studies. Community similarity was computed using the Bray-Curtis dissimilarity measure and the sample dates were clustered using UPGMA clustering implemented with the Lance and Williams (1967) flexible clustering algorithm with the intensity parameter,  $\beta$ , set at -0.1 (Belbin, Faith et al. 1992). Species were clustered using the same algorithm, but with  $\beta = -0.25$  (Belbin, Faith et al. 1992) and using Austin and Belbin's (1982) two-step dissimilarity coefficient. The results of these clusterings were displayed in a two-way table which efficiently summarises which species groups are associated with which groups of samples. These analyses were carried out in PATN version 3.03 (Belbin and Collins 2004).

## **5.2.2 Zooplankton community composition**

### **5.2.2.1 Sample collection, preservation, identification and enumeration**

Quantitative zooplankton samples were taken at the same times and sites as the algal samples. At each site, three samples were taken with a 2.2 L Van-Dorn bottle, giving a total water volume collected at each site of 6.6 L. The Van-Dorn bottle was lowered vertically into the water column so the top of the bottle was at a depth of 300 mm. The bottle was then manually released to obtain a depth-integrated sample. The water collected was then passed through a 54  $\mu$ m mesh plankton net and the material retained on the mesh was preserved using 1-2 mL of glutaraldehyde per 100 mL of filtrate (A.P.H.A. 1992). The samples from each site were pooled giving a total filtered water volume of 26.4 L.

Similar pooling procedures have proven satisfactory in a previous study of changes in zooplankton in these lakes (Burrows 1968) and were comparable to methods used by Jeppesen et al. (1999) in characterising zooplankton community characteristics in their study on trophic dynamics in shallow lakes. Also, Burrows (1968) found no evidence of vertical stratification of zooplankton in lakes Sorell or Crescent. It was therefore concluded that the method of sampling was adequate.

Qualitative zooplankton sampling was carried out by the Inland Fisheries Service between January 1996 and April 2000 at approximate monthly intervals. A total of 55 samples were collected for Lake Sorell and 56 samples were collected for Lake Crescent. These samples consisted of 154  $\mu$ m mesh net tows taken at the mid water site of both lakes Sorell and Crescent. Samples were preserved with buffered formalin to a final concentration of approximately 4% (A.P.H.A. 1992).

Each preserved quantitative sample was inverted a number of times to ensure homogeneity and sub-sampled if necessary. Cladocerans (3 species) were counted and sorted according to size class. Biomass estimations were made using the relationships of Dumont (1975). The cladocerans were the only group of animals analysed quantitatively for biomass estimates because these are the largest zooplanktonic grazers of phytoplankton in these lakes and are the most likely to limit phytoplankton standing crop through grazing pressure (Scheffer 1998).

Qualitative estimates of zooplankton community composition were determined in a similar way. A sub-sample was taken and counts were made using a stereo microscope at 20 × magnification. Individual species of Cladocera (3), Copepoda (2), and Rotifera (1) were identified and sorted according to size class. Relative estimates of species composition and size class were made. The aim of the qualitative analysis was to determine if zooplankton community composition had changed with the severe degradation in water quality (Chapter 2). For the qualitative analysis therefore, all zooplankton groups were identified and enumerated.

### 5.2.3 *Estimating the abundance of golden galaxias (planktivore)*

Quantitative estimates of the abundance of golden galaxias (*Galaxias auratus*) were made on data supplied by Hardie (2003). Six randomly located seine net hauls (100 m x 2.8 m of net with a 5 mm stretched mesh length) were made from boat in Lake Crescent on the 24<sup>th</sup> of May 2001. The total surface area of lake sampled was 400 m<sup>2</sup>, and the geometric mean number per m<sup>2</sup> was calculated since geometric means are more appropriate for population estimates (Crawley 2002 p. 71-2). This was then converted to an estimate of the total fish in Lake Crescent by multiplying up to the surface area of the lake on that sampling date (15.61 km<sup>2</sup>). A coarse estimate of the abundance of this fish in Lake Sorell was then made assuming the average 10-fold smaller abundance in Sorell documented by Hardie (2004) held at this time and using the surface area of Sorell (38.82 km<sup>2</sup>). The total wet biomass of golden galaxias in each lake was then estimated using Hardie's (2003) length-weight regressions based on a random sample of 807 fish from Crescent and 773 fish from Sorell, where the wet weight,  $y$ , of a fish can be calculated from the regression equation on the total fork length,  $x$  (both  $R^2 > 0.96$ ):  $y = 9 \times 10^{-7} x^{3.4129}$  for Lake Crescent and  $y = 8 \times 10^{-7} x^{3.5211}$  for Lake Sorell.



The mean length of a sample of 4 475 golden galaxiids from Lake Crescent spanning the period of this study was 81.8 mm (95% confidence interval:  $\pm 0.6$  mm), while the mean of a sample of 2 558 fish from Lake Sorell was  $76.6 \pm 0.9$  mm for the same period. These “average” fish lengths were used to convert the populations estimates to wet biomass. These wet biomasses were then converted to concentrations of fish as  $\text{g m}^{-3}$  using the estimated volumes of each lake ( $15.09 \times 10^6 \text{ m}^3$  for Crescent and  $84.43 \times 10^6 \text{ m}^3$  for Sorell).

#### 5.2.4 *Estimating the abundance of brown trout (piscivore)*

Brown trout (*Salmo trutta*) biomass and abundance estimates were obtained from the unpublished data held by the Inland Fisheries Service, Hobart, Tasmania. The brown trout (*Salmo trutta*) are the only significant piscivore present and prey heavily on the golden galaxias (Hardie 2003). Brown trout in Lake Sorell have been shown to selectively feed on the golden galaxiid at times (Stuart-Smith, Richardson et al. 2004) and have the potential to significantly influence galaxiid numbers.

In the past, Lake Sorell has supported a highly productive and lucrative recreational trout fishery with an Inland Fisheries Service angler survey of the 1992-3 season recording ~11 000 anglers to have visited Lake Sorell over the season catching a total of approximately 123 000 fish, with an average catch rate of about 1.5 fish per angler per day. Catch rates in Lake Sorell had not changed significantly since the 1950s despite significant increases in anglers visiting the lakes each year (Gudde 2004). With angler catch rates being as high as 123 000 fish per year, reflecting a possible trout population of over 1 million fish to exist in Lake Sorell at times (J. Diggle, Inland Fisheries Service, Hobart. Pers. comm.).

Lake Crescent by comparison has limited natural recruitment and has been managed as a ‘trophy trout’ water. This has led to significantly fewer fish that are larger in size than in Lake Sorell. Accurate estimates of brown trout numbers in the past are limited for Lake Crescent, but the total number of trout present is believed to have been as few as 10 000 fish in total (Inland Fisheries Service, Hobart, unpublished data). Hardie (2003) hypothesises that the contrast in golden galaxiid numbers between lakes Sorell and Crescent is a direct result of differing levels of predation and competition pressure by brown trout.

Recently, brown trout numbers have declined significantly in Lake Sorell. This has been attributed to poor levels of recruitment over the last 6 years (Inland Fisheries Service, Hobart, unpublished data; T. Farrell, Inland Fisheries Service, Hobart. Pers. comm.) and is possibly due to the degradation in environmental conditions and a reduction in suitable spawning resources such as low and sporadic water flows in the tributaries entering the lake. Accurate estimates of brown trout numbers at present in Lake Sorell are as little as 50 000 fish in total (T. Farrell, Inland Fisheries Service, Hobart. Pers. comm.). Therefore, major changes in the trophic dynamics of the lakes may have occurred in recent years as the heavy level of predation exerted on the galaxiid population in Lake Sorell may have been reduced.

In June 2002, 3 000 adult brown trout were translocated from Great Lake to Lake Crescent with the aim of improving fish stocks for the eventual re-opening of Lake Crescent to the public (J. Diggle, Inland Fisheries Service, Hobart. Pers. comm.). All fish were weighed, measured and tagged prior to release and formed the basis of a mark-recapture estimate of trout numbers in Lake Crescent using the Petersen estimate, with an extensive netting survey conducted in October 2002 by the Inland Fisheries Service. A similar mark-recapture survey was undertaken in Lake Sorell in 1997 and 2001. Fish were trapped in Mountain Creek from May to July during the 1997 and 2001 spawning migration. The trapped fish were weighed, measured, tagged and released. During the period running August to September of 1997 and 2001, after which the tagged fish had returned to the main body of the lake, extensive netting surveys were undertaken in Lake Sorell and population estimates made in the same manner as Lake Crescent.

Because of the disparate size classes of the recently introduced trout from Great Lake and those that had been in Lake Crescent prior to the restocking, wet biomasses for the two groups of fish were estimated from weights of 2 991 of the Great Lake fish (mean mass = 1 000 g), 203 of the remnant population of Lake Crescent brown trout (mean mass = 2 634 g) and the 330 rainbow trout (mean mass = 1 566 g) that were caught during the surveys. The total wet biomass and concentrations of these fish were then calculated as for golden galaxias.

## 5.3 Results

### 5.3.1 *Phytoplankton community composition*

There were two main groups of survey dates in the resulting clustering and eight groups of taxa (Table 30). The survey from 1987 by Cutler et al. (1990) recorded far fewer taxa than either this or Cheng and Tyler's surveys and was probably due to the much more limited scope of Cutler et al.'s consultancy report with phytoplankton samples being taken in both lakes on the 20<sup>th</sup> of March, 1987 only. Overall, however, the phytoplankton community structure has not changed substantially since the work carried out between 1969 and 1972 by Cheng and Tyler (1973a), while Cutler et al. (1990) did record many of the common taxa in species groups D and E of Table 30. Essentially, species diversity remains comparable, and the features of diatom dominance in Lake Crescent and green 'filamentous' dominance in Lake Sorell still holds: in Table 30 the diatoms in species groups B and G are generally in higher abundance categories in Lake Crescent than in Sorell, while minute Ulotrichaceae and *Pediastrum* species (from species group F) were amongst the most abundant taxa found in Sorell. The relative differences between the two lakes is further emphasised in the temporal plots of relative composition over 2000 to early 2002 (Figure 66, Figure 67).

Despite a new taxon of cyanobacteria, *Arthrospira*, being recorded from Lake Crescent in 2001, *Microcystis* remains common as reported by Cheng and Tyler. Both taxa remain absent from Lake Sorell. Moreover, the proportion of cyanobacteria is usually < 5% of the total phytoplankton abundance (Figure 66) so there is little evidence to support a contention that these lakes are undergoing a change towards domination by cyanobacteria (cf. Force 1992).

The main differences between this survey and those of Cheng and Tyler were that several taxa were only recorded from these lakes in 2001 (species groups A, B and C), including three species of Naviculaceae and two species of Aulacoseiraceae (Table 30). These five species of diatom were abundant in Lake Crescent; whereas in Lake Sorell, *Navicula* 'a', *Aulacoseira* 'a' and *Aulacoseira* 'b' were found to be abundant, with *Navicula* 'b' recorded as rare. Whether these differences reflect some changes to the algal composition of these lakes, or whether they reflect improved taxonomy since

the 1970s is difficult to determine without access to the original collections of Cheng and Tyler. Notably, however, there were no taxa recorded by them that were not recorded in the 2001 survey. The two species of Aulacoseiraceae are likely synonymous with the species of *Melosira* described by Cheng and Tyler (<http://www.calacademy.org/research/diatoms/genera/aulacoseira/> ; <http://www.fhsu.edu/biology/Eberle/DiatomListCentrics.html>). Taxonomic imperfections do not seriously affect the ecological considerations of this study.

The other marked difference between the phytoplankton of the two lakes is the differences in the proportion of taxa classifiable as meroplanktonic or benthic using Ogilvie and Mitchell's (1998) classification. Expressed as a proportion of abundance (cells/mL), Lake Crescent averaged more than a 3-fold greater proportion of meroplankton than Sorell (Table 31), with a maximum proportion of 97% on one occasion. Lake Crescent also maintained a much higher absolute abundance of meroplankton than Lake Sorell (> 24-fold on average: Table 31). The dominance of meroplankton in Lake Crescent is further exemplified by *Diatoma elongatum* (Table 32), a diatom species known to be meroplanktonic with the capacity to produce resting cells (Schelske, Carrick et al. 1995) which allow it to settle out of the euphotic zone and survive before resuspension into the water column during a later stage in the life-cycle (Reynolds 1984). The benthic inoculum of resting cells preadapts the community by providing a reservoir of colonising individuals that seed the water column during resuspension and entrainment (Harris 1986) providing a means of increasing long-term survival (McQuoid and Hobson 1996).

**Table 30** Two-way table of the taxonomic composition of phytoplankton for lakes Sorell and Crescent for surveys taken during 2001 for this study and previous surveys reported by Cheng and Tyler (1973) and Cutler et al. (1990). Black lines denote divisions between the clustering groups, and the letters under "species groups" denote the groups of species referred to in the text; grey shading denotes a non-zero record for that species. An asterisk denotes that the species was recorded in the relevant survey; numerical abundance categories are given for the 2001 survey: 1 = very rare; 2 = rare; 3 = common; 4 = abundant. Taxa in bold are known to be benthic or meroplanktonic (Ogilvie and Mitchell 1998).

Broad taxonomic division	Taxa	Species group	Crescent 1973	Sorell 1973	Crescent 2001	Sorell 2001	Crescent 1987	Sorell 1987
Cyanobacteria	<i>Arthrospira</i>	A			3			
Chlorophyta (Green Algae)	<i>Dicellula</i>	A			2			
Bacillariophyta	<b><i>Fragilaria</i></b>	A			1			
Chlorophyta (Green Algae)	<i>Genicularia</i>	A			3			
Bacillariophyta	<b><i>Navicula 'c'</i></b>	A			3			
Chlorophyta (Green Algae)	<i>Pseudophaeocystis</i>	A			1			
Chlorophyta (Green Filaments)	<i>Ulothrix</i>	A			1			
Bacillariophyta	<b><i>Aulacoseira 'a'</i></b>	B			3	3		
Bacillariophyta	<b><i>Aulacoseira 'b'</i></b>	B			4	3		
Chlorophyta (Desmids)	<i>Desmidium</i>	B			2	2		
Bacillariophyta	<b><i>Navicula 'a'</i></b>	B			4	3		
Bacillariophyta	<b><i>Navicula 'b'</i></b>	B			3	2		
Chlorophyta (Green Algae)	<i>Scenedesmus 'b'</i>	B			2	1		
Chlorophyta (Green Algae)	<i>Volvocaceae</i>	B			2	2		
Chrysophyta	<i>Mallomonas</i>	C				1		
Chlorophyta (Green Algae)	<i>Oedogonium</i>	C				1		
Bacillariophyta	<b><i>Asterionella</i></b>	D	*	*	1	1		*
Chlorophyta (Green Algae)	<i>Dictyosphaerium</i>	D	*	*	2	2		*
Cyanobacteria	<i>Gomphosphaeria</i>	D	*	*	1	1	*	
Bacillariophyta	<b><i>Melosira</i></b>	D	*	*	3	3		*
Chlorophyta (Green Algae)	<i>Oocystis</i>	D	*	*	3	2		*
Chlorophyta (Green Algae)	<i>Botryococcus</i>	E	*	*		1	*	
Chlorophyta (Desmids)	<i>Closterium 'a'</i>	E	*	*	3	3	*	*
Chlorophyta (Desmids)	<i>Closterium 'b'</i>	E	*	*	3	2	*	*
Chlorophyta (Desmids)	<i>Cosmarium</i>	E	*	*	3	1	*	*
Chlorophyta (Green Filaments)	<i>Mougeotia</i>	E	*	*	4	2	*	*
Bacillariophyta	<b><i>Rhizosolenia</i></b>	E	*	*	1		*	*
Bacillariophyta	<i>Surirella</i>	E		*		1		*
Bacillariophyta	<b><i>Cyclotella</i></b>	F	*	*	2	2		
Chlorophyta (Green Filaments)	<i>Geminella</i>	F	*	*	3	1		
Chlorophyta (Green Algae)	<i>Gonium</i>	F	*	*	2	1		
Chlorophyta (Green Algae)	<i>Kirchneriella</i>	F	*	*	2	2		
Chlorophyta (Green Filaments)	<i>Minute Ulotrichaceae</i>	F	*	*	4	4		
Chlorophyta (Green Algae)	<i>Pediastrum 'a'</i>	F	*	*	3	3		
Chlorophyta (Green Algae)	<i>Pediastrum 'b'</i>	F	*	*	3	2		
Chlorophyta (Desmids)	<i>Staurastrum 'a'</i>	F	*	*	3	1		
Chlorophyta (Desmids)	<i>Staurodesmus</i>	F	*	*	2	1		
Chlorophyta (Green Algae)	<i>Ankistrodesmus</i>	G			2	1	*	
Bacillariophyta	<b><i>Cymbella</i></b>	G		*	3	1		
Bacillariophyta	<b><i>Diatoma elongatum</i></b>	G	*		4	2		
Bacillariophyta	<b><i>Pinularia</i></b>	G		*	2	1		
Chlorophyta (Green Algae)	<i>Scenedesmus 'a'</i>	G	*		3	2		
Chrysophyta	<i>Dinobryon</i>	H	*	*	1			
Cyanobacteria	<i>Microcystis</i>	H	*		3			
Cyanobacteria	<i>Oscillatoria</i>	H	*	*	1			
Chlorophyta (Green Algae)	<i>Pediastrum 'c'</i>	H	*	*	3			
Dinophyceae	<i>Peridinium</i>	H	*	*	1			
Chlorophyta (Green Filaments)	<i>Spirogyra</i>	H		*	1			
Chlorophyta (Desmids)	<i>Staurastrum 'b'</i>	H	*	*	1			
Chlorophyta (Desmids)	<i>Staurastrum 'c'</i>	H	*	*	2			
Chlorophyta (Green Algae)	<i>Tetrahedron</i>	H		*	2			

Figure 66 Relative phytoplankton community composition in Lake Crescent for the period 19<sup>th</sup> April 2000 - 25<sup>th</sup> July 2002.

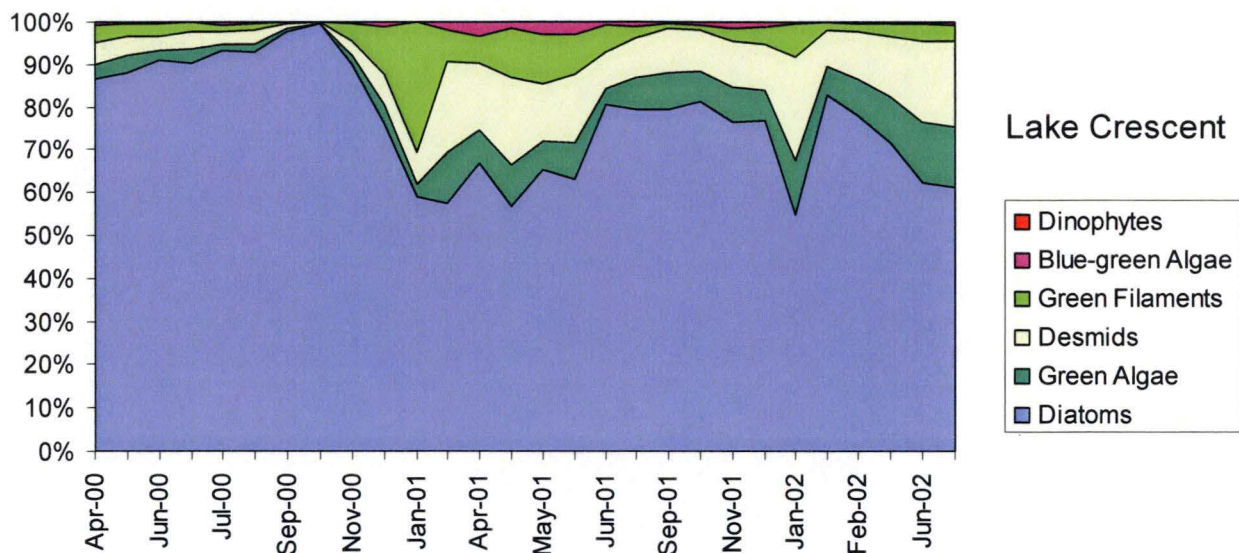


Figure 67 Relative phytoplankton community composition in Lake Sorell for the period 19<sup>th</sup> April 2000 - 25<sup>th</sup> July 2002.

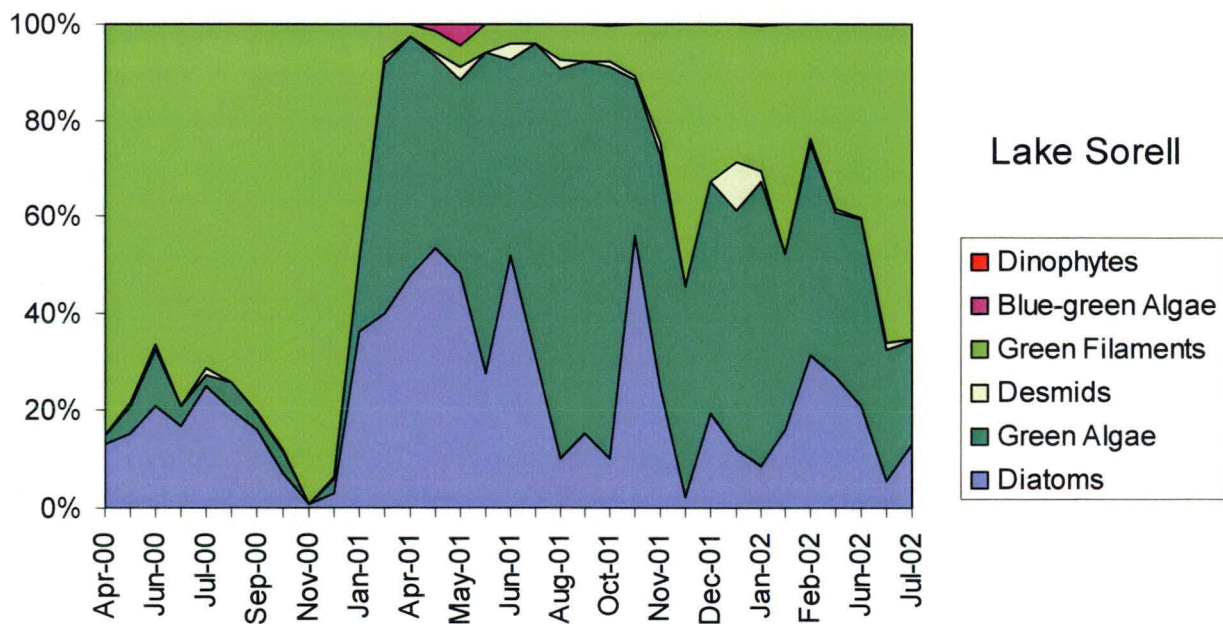


Table 31 Comparison of proportion and total individual phytoplankton counts of meroplankton in lakes Sorell and Crescent for the period 19<sup>th</sup> April 2000 - 25<sup>th</sup> July 2002.

	Sorell % Meroplankton	Crescent % Meroplankton	Sorell Total Individuals	Crescent Total Individuals
<b>Mean</b>	<b>23%</b>	<b>77%</b>	<b>850</b>	<b>20580</b>
N	32	28	32	28
Minimum	1%	55%	50	1905
Maximum	56%	98%	2785	60150
Std. Dev.	16%	13%	700	15050

Table 32 Overview of *Diatoma* dominance of the phytoplankton community in Lake Crescent for samples taken for the period running 19<sup>th</sup> April 2000 - 25<sup>th</sup> July 2002.

	% <i>Diatoma</i> of total phytoplankton	% <i>Diatoma</i> of total diatom
<b>Mean</b>	<b>61%</b>	<b>78%</b>
N	28	28
Minimum	20%	31%
Maximum	95%	98%
St. Dev.	20%	17%

In terms of seasonal changes in algal composition, the two lakes remain as different to each other as originally reported by Cheng and Tyler: Crescent has much higher standing stocks of algae both in terms of concentrations of cells and chlorophyll-*a* (Figure 68 and Figure 69; see also Chapter 2). Temporal changes in composition also closely followed those documented by Cheng and Tyler (1973a) with diatoms dominating in Lake Crescent throughout the year (Figure 66), while in Lake Sorell dominance oscillated between green algae, diatoms and green filamentous taxa (Figure 67).



Figure 68      Phytoplankton community composition in Lake Crescent for 19<sup>th</sup> April 2000 - 25<sup>th</sup> July 2002.

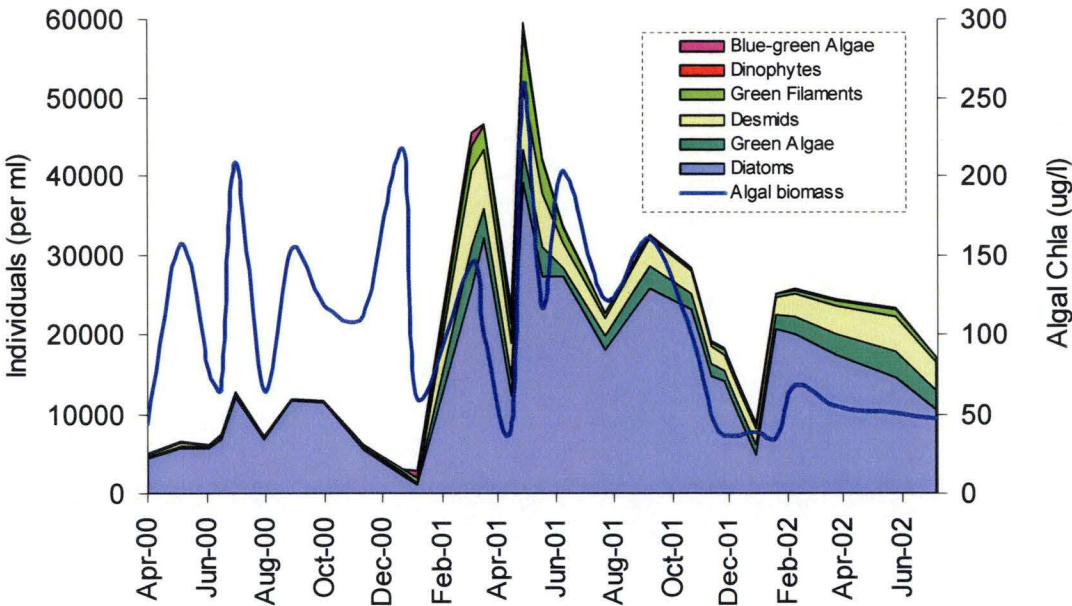
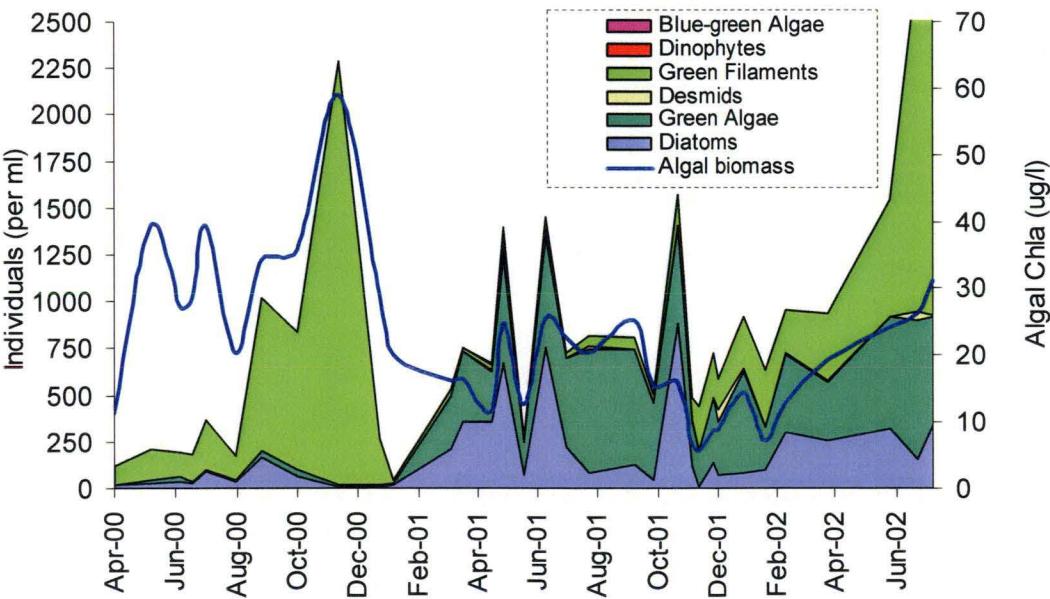


Figure 69      Phytoplankton community composition in Lake Sorell from 19<sup>th</sup> April 2000 - 25<sup>th</sup> July 2002.





### 5.3.2 Zooplankton community structure

As with the phytoplankton, the zooplankton of the two lakes differ substantially. Lake Crescent supports a much greater proportion of small copepods and cladocerans (98.7% of individuals < 1 mm) than Lake Sorell (86 % of individuals < 1 mm) (Figure 70), while Lake Sorell has a greater abundance of larger cladocerans more frequently than Lake Crescent (Figure 71). Overall, ~ 10% of the 31 990 individuals counted from Lake Sorell were *Daphnia* > 1 mm long while only 0.21% of the 18 520 individuals from Lake Crescent were of such large *Daphnia*.

Figure 70 and Figure 71 describe the temporal changes in the relative zooplankton community composition of lakes Crescent and Sorell for all samples taken between November 1996 and August 2002. The larger *Daphnia* species are all but absent from Lake Crescent although, interestingly, its occurrence preceding major changes in water quality during 1999/2000 is less frequent than after 2000, possibly reflecting a reduction in grazing pressure from the golden galaxiid due to reduced water clarity (Stuart-Smith, Richardson et al. 2004) (Figure 70). Although large *Daphnia* are more frequent in Lake Sorell, smaller copepods and cladocerans are still numerically important, with *Daphnia* “blooms” being a sporadic phenomenon (Figure 71).

Figure 70      Relative zooplankton community composition for Lake Crescent, November 1996 to August 2002.

**Cladocerans** : Bosmina, Ceriodaphnia, Daphnia.

**Copepods**: Calanoid, Cyclopoid.

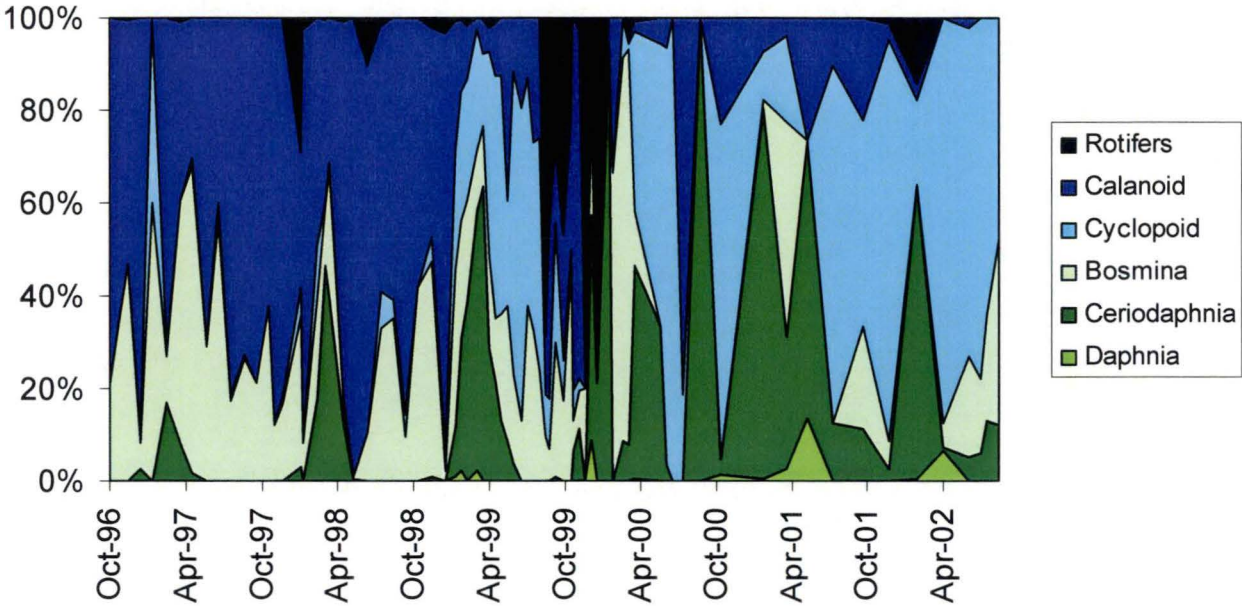


Figure 71      Relative zooplankton community composition for Lake Sorell, November 1996 to August 2002.

**Cladocerans** : Bosmina, Ceriodaphnia, Daphnia.

**Copepods**: Calanoid, Cyclopoid.

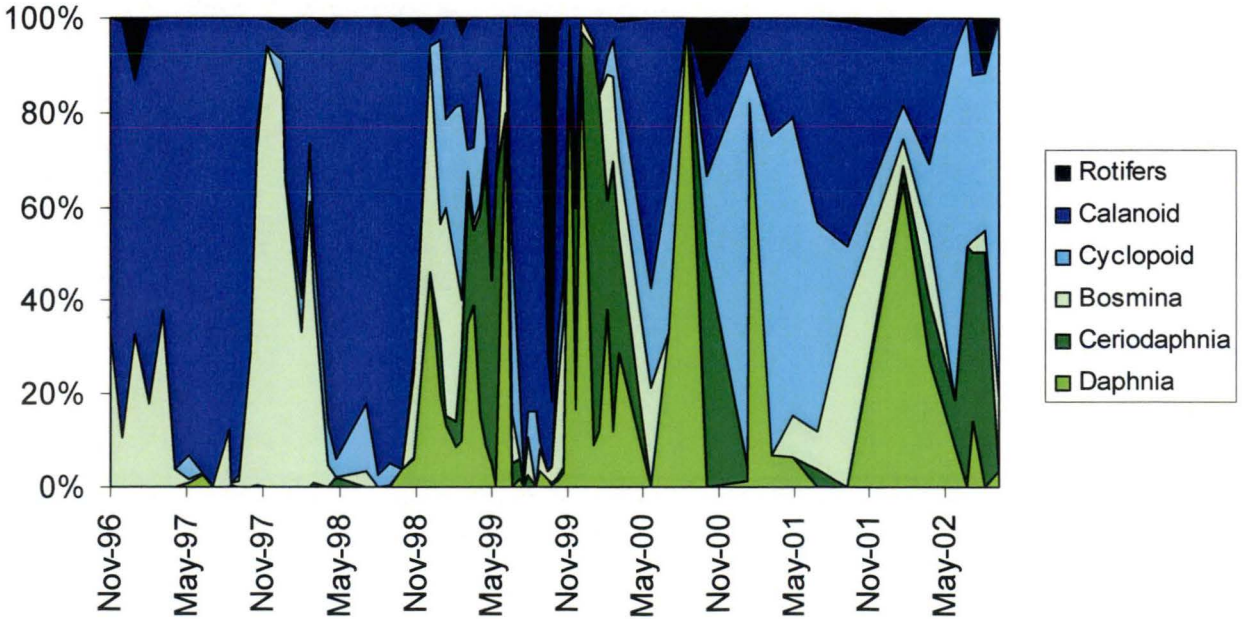


Figure 72 and Figure 73 plot changes in cladoceran densities and biomass overlaid with algal biomass from April 2000 to August 2002. Apart from re-emphasising the lack of larger Cladocera in Lake Crescent, the biomass of zooplankton is generally higher in Lake Sorell (as might be expected from the exponential relationship between body size and biomass for cladoceran species (Dumont, Van de Velde et al. 1975)). Since grazing pressure by zooplankton has been shown to be directly related to zooplankton biomass (Scheffer 1998), one might expect that grazing pressure on phytoplankton in Lake Sorell would be significantly higher than in Lake Crescent.

Although the lowest recorded levels of algal biomass do closely coincide with peaks in cladoceran biomass in Lake Sorell, overall there is little correlation between the density or biomass of cladocerans and algal biomass. This is probably due to the strong relationship between wind resuspension, increased suspended solids and algal productivity (Chapter 2 and Chapter 3; Scheffer, 1998), which may mask any influence zooplankton grazing may have in limiting phytoplankton biomass. It is also worth re-emphasising that the biomass of algae in Lake Crescent is much greater than in Sorell, and it is, therefore, unlikely that phytoplankton would be a limiting resource for the zooplankton in Crescent further diminishing the likelihood that grazing would be exerting a strong influence on the phytoplankton of this lake.

Another means of appraising the potential impact of zooplankton grazing is by computing the ratio of the dry weight of cladocerans to phytoplankton biomass for each sampling occasion. On average this ratio was 10 × greater in Sorell (0.047) than Crescent (0.004), although the median values were similar; the difference in mean values was driven by a few high values in Lake Sorell (Figure 74). It reached a peak of 0.8 in Lake Sorell in November 2001, while the second highest value was ~ 0.5 in January 2002; otherwise the ratio was generally < 0. Assuming cladocerans consume an amount of food equivalent to their body mass per day (Jeppesen, Jensen et al. 1999), the cladoceran biomass would have the potential to consume between 50% and 80% of the phytoplankton standing crop per day during these peaks in Lake Sorell.

While this suggests that zooplankton grazing could be influential in Sorell on these dates, this could underestimate the role that zooplankton could be playing in this lake.

The 2-3 weekly sampling frequency employed in this survey probably missed some “blooms” of *Daphnia*, since S. Hardie (Inland Fisheries Service) noted qualitative high concentrations of *Daphnia* while sampling fish during December 2000 that fell between the sampling trips for this current survey.

Figure 72 Cladoceran densities (by size class) and algal biomass for lakes Crescent and Sorell, April 2000 to August 2002.

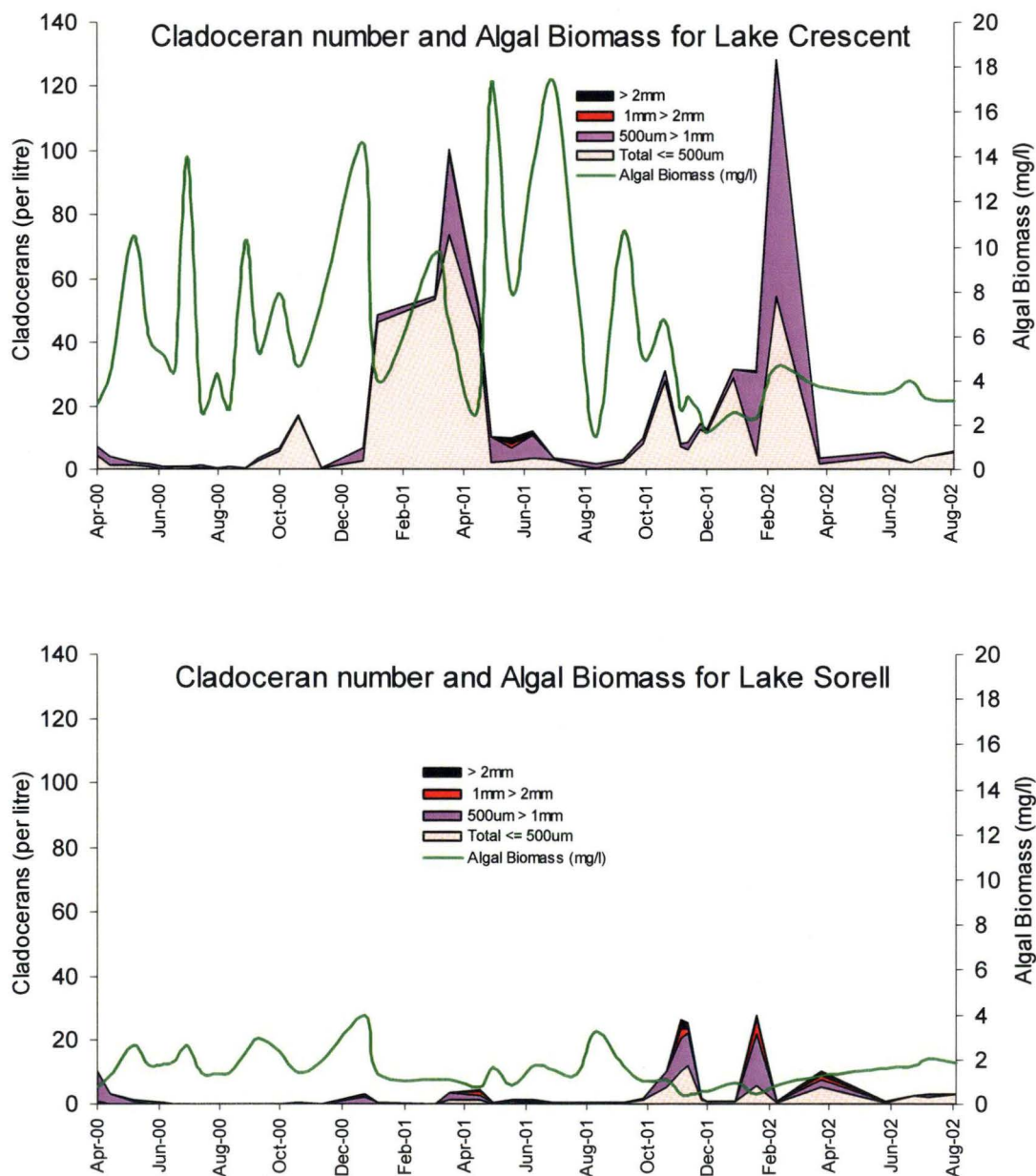


Figure 73 Cladoceran biomass (by size class) and algal biomass for lakes Crescent and Sorell, April 2000 to August 2002.

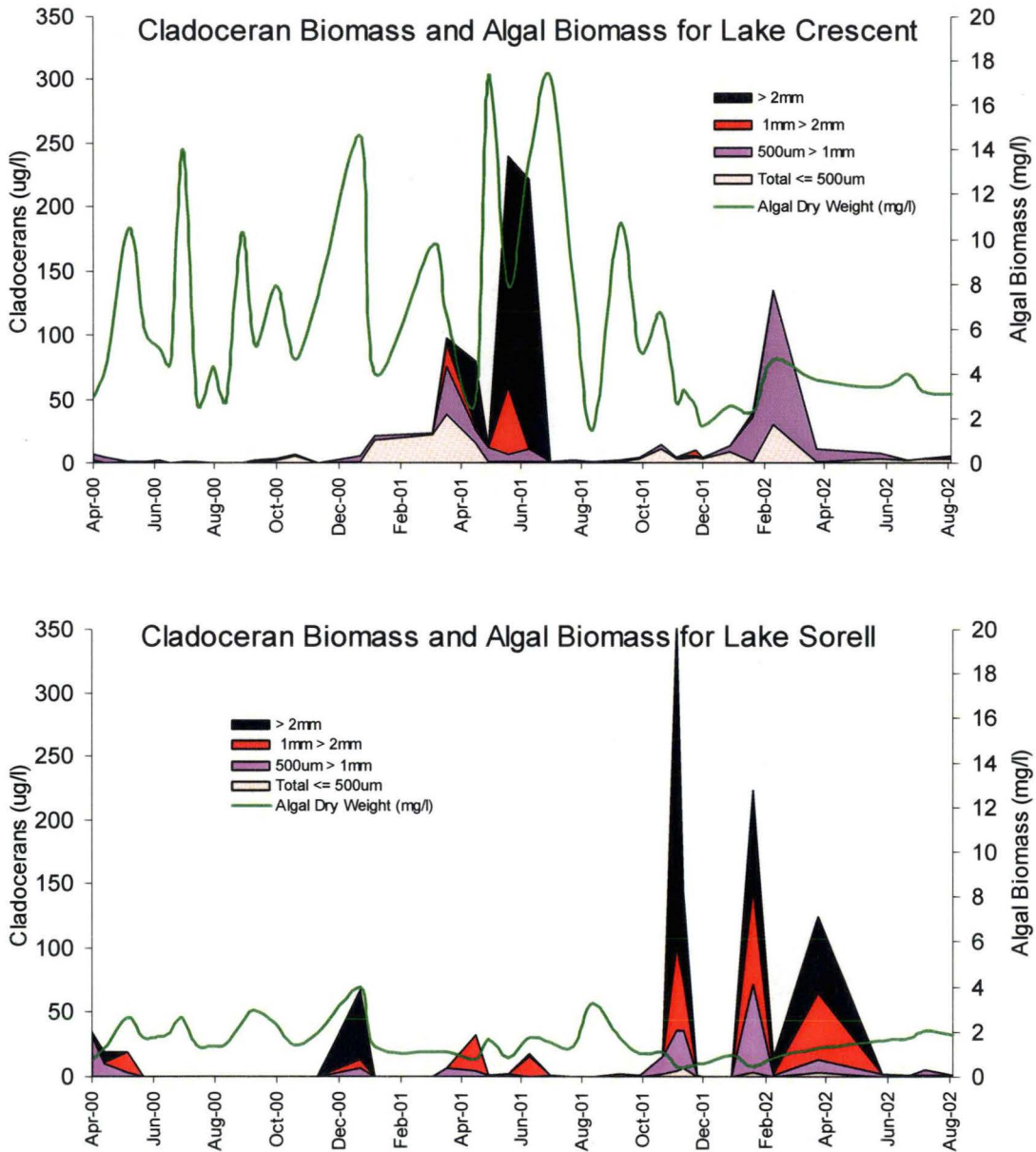
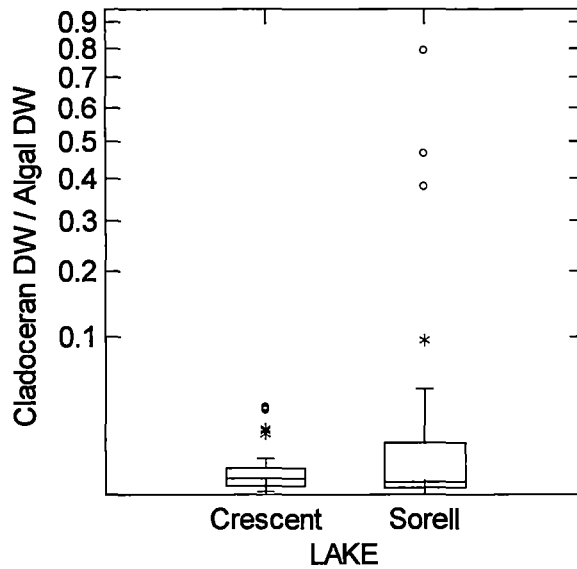




Figure 74 Summary of the ratio of cladoceran dry weight to algal dry weight for lakes Crescent and Sorell for the period running April 2000 to August 2002. 41 samples included for each lake.

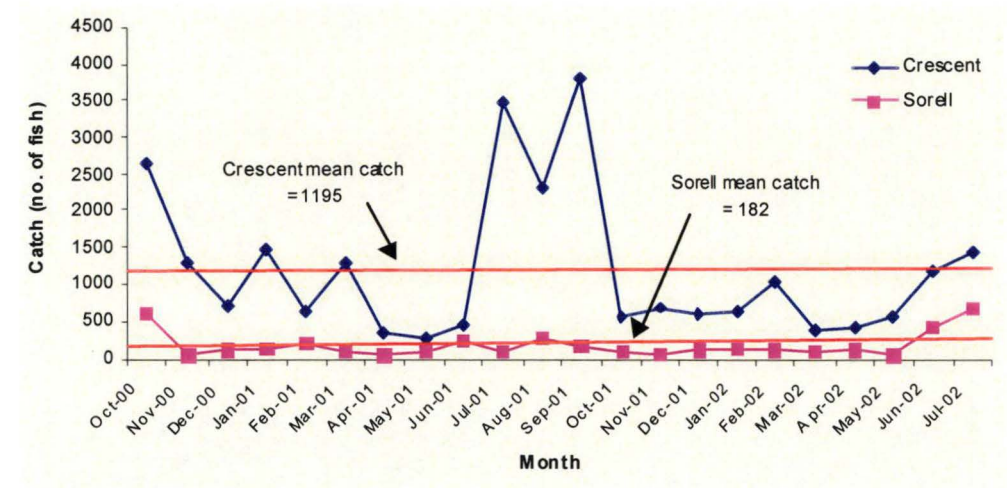


### 5.3.3 *Golden galaxias (planktivore)*

#### 5.3.3.1 Abundance estimates

Figure 75 shows that Lake Crescent has at least ten times the abundance of golden galaxias than Lake Sorell (Hardie 2003). Consequently, the predation pressure on zooplankton in Lake Crescent would be significantly higher than in Lake Sorell. Elsewhere size selective feeding on large bodied cladocerans by planktivores has been shown to be important in regulating the composition of the zooplanktonic community (Vanni 1986; Northcote 1988; Modenutti, Balseiro et al. 1993; Donald, Vinebrooke et al. 2001) and this could contribute to the dearth of *Daphnia* in Lake Crescent.

Figure 75      Total monthly catches of adult golden galaxias in Lakes Crescent and Sorell, 2000-2002. (Hardie 2003).



The geometric mean number of fish found in Lake Crescent was 51.66 per 400 m<sup>2</sup> area sampled (= 0.1292 fish m<sup>-2</sup>). (Arithmetic mean: 67.5 fish per 400 m<sup>2</sup>, median: 47.5 fish per 400 m<sup>2</sup>). This translates to 2 016 130 fish over the lake. Overall, Lake Crescent has approximately 30 × the concentration of this zooplanktivore that Lake Sorell has (Table 33).

Table 33      Calculated values of the number, wet biomass and concentrations of golden galaxiids in May 2001

	Number of fish	Wet biomass (kg)	Concentration (g/m <sup>3</sup> )
Crescent	2 016 130	9 503	0.6298
Sorell	501 385	1 729	0.0205

5.3.3.2    Assessment of the predation pressure exerted on zooplankton

Stuart-Smith et al. (2004, unpublished paper) followed methods of Elliott and Persson (1978) to estimate a daily ration of 0.65% of wet body weight per day for golden galaxiids feeding on *Daphnia*. Therefore, in Lake Crescent, galaxiid predation pressure on zooplankton may reach levels of up to 80 kg day<sup>-1</sup> compared to 15 kg day<sup>-1</sup> in Lake Sorell. Comparisons of galaxiid grazing rates based on a fish biomass per unit of lake volume results in galaxiid predation pressures on

zooplankton being approximately  $5.5 \text{ mg zoop m}^{-3} \text{ day}^{-1}$  in Lake Crescent compared to  $0.175 \text{ mg zoop m}^{-3} \text{ day}^{-1}$  in Lake Sorell, a difference of approximately 30 times.

Dietary analyses conducted by Hardie (2003) over the period 2000 – 2001 showed that *Daphnia* was absent from the diet of these fish during periods when this species was scarce (winter and early spring: Figure 72 and Figure 73), and the fish tend to target alternative, predominantly benthic species (in the Amphipoda, Isopoda and Ephemeroptera). During summer, *Daphnia* biomass increased and the galaxiids switched to planktonic cladocerans as the dominant prey item, in spite of the continued availability of the benthic taxa. This suggests that golden galaxias is not reliant on zooplankton as a dietary item, but prefers cladocerans when available (Hardie 2003). By contrast, similarly-sized juvenile brown trout (*Salmo trutta*) selectively fed on benthic Plecoptera, Ephemeroptera and Amphipoda even during the periods of high densities of *Daphnia* (Hardie 2003).

Hardie (2003) also found golden galaxiids to strongly prefer *Daphnia* > 1 mm in size. Examining average cladoceran biomass for animals > 1 mm (Figure 72 and Figure 73) and comparing concentrations of cladocerans to estimates of galaxiid predation rates further underlines the potential for galaxiid predation to influence zooplankton community structure in Lake Crescent. For example, the estimated galaxiid predation rate in Lake Crescent of  $5.5 \text{ mg zooplankton m}^{-3} \text{ day}^{-1}$  would remove approximately 45% of cladocerans > 1 mm (based on average cladoceran biomass estimates made between April 2000 and August 2002) compared to Lake Sorell, in which a galaxiid grazing rate of  $0.175 \text{ mg zooplankton m}^{-3} \text{ day}^{-1}$  would remove approximately 0.9% of average cladoceran biomass for individuals > 1 mm. From these estimates, it is clear that golden galaxiid zooplankton predation has the potential to influence the zooplankton community through selective feeding on large bodied zooplankters >1mm in size (Brooks and Dodson 1965; Hardie 2003).



### **5.3.4 Brown trout (*piscivore*)**

#### **5.3.4.1 Population estimates**

The Petersen Mark-Recapture estimate ( $\pm$  95% confidence interval) was  $3\,524 \pm 169$  trout in Lake Crescent (based on an initial release of 3 000 marked fish, 269 fish captures and 229 marked recaptures). In Lake Sorell the brown trout population in 1997 was estimated to be approximately  $100\,000 \pm 20\,000$  fish (IFS 1997) (Table 34). Therefore the population of brown trout in Lake Sorell was concluded to be between 80 000 and 120 000 fish. Unfortunately the survey undertaken in 2001 failed to catch a substantial number of tagged fish resulting in an unreliable confidence interval for the estimated 50 000 fish in the lake at this time (IFS unpublished data).

The total estimated wet biomass of brown and rainbow trout in Lake Crescent was 4 043 kg yielding a concentration of  $0.153\text{ g m}^{-3}$  (95% confidence interval  $\pm 0.007\text{ g m}^{-3}$ ), while prior to the introduction of Great Lake trout, the remnant Lake Crescent trout population consisted of approximately 535 kg of brown trout and 517 kg of rainbow trout, giving a total biomass of 1 050 kg of trout which translates to a concentration of  $0.040\text{ g m}^{-3}$  for the period covered by quantitative surveys of phytoplankton, zooplankton and golden galaxias.

The biomass and concentration estimates for Lake Sorell are summarised in Table 34, and it is clear that Sorell has a much greater concentration of trout biomass than Crescent, even when Crescent is augmented by translocated fish from elsewhere. The concentration of trout in Lake Sorell is between 10 and 20 times that of Lake Crescent, inversely comparable in magnitude to the difference in golden galaxias concentration between the lakes. In Lake Sorell, the trout population is approximately 10 - 30 times the concentration of the golden galaxias. In Lake Crescent, the golden galaxias is approximately 20 times more concentrated than the remnant Lake Crescent trout population. At current levels, the golden galaxias is approximately 5 times more concentrated than the Lake Crescent trout population. The introduction of the Great Lake brown trout has significantly increased potential predation pressure on the golden galaxias by about 5 times in Lake Crescent.

Table 34 Approximate abundance and biomass estimates of brown trout for Lake Sorell, 1997 and 2001. (IFS unpublished data).

	Abundance (n)	Average Weight (g)	Total Biomass (kg)	Concentration (g/m <sup>3</sup> )
Brown trout August 1997	102 179	906	92 574	0.757
Brown trout September 2001	51 717	840	43 442	0.388

#### 5.3.4.2 Assessing the predation pressure of trout

Trout feeding rates can be directly related to fish size and biomass (Elliott 1994). Assuming trout to have a daily ration between 2% and 6% of wet body weight (Elliott 1975; Elliott and Persson 1978; Spigarelli, Thommes et al. 1982; Koskela, Pirhonen et al. 1997), and assuming they prey exclusively on the golden galaxias, predation pressure on golden galaxias in Lake Sorell (using 2001 brown trout biomass estimates; Table 34) could reach levels of 870 to 2 600 kg of galaxiids day<sup>-1</sup> from a galaxiid population with a total estimated biomass of 2 300 kg. Comparably, for Lake Crescent, using trout biomass estimates of fish present before the introduction of brown trout from Great Lake, predation pressure on golden galaxias could reach levels of 20 to 60 kg of galaxiids day<sup>-1</sup> taken from a population with a total estimated biomass of 12 500 kg. Brown trout in Lake Sorell therefore have a greater potential to seriously limit galaxiid biomass and abundance than in Lake Crescent, with the contrast in galaxiid biomass between the two lakes being adequately accounted for by differing predation pressure from trout.

## 5.4 Discussion

A substantial increase in lake level should lead to a reduction in the extent and frequency of sediment resuspension (Chapter 4). This should increase water clarity and, with the loss of aquatic macrophytes, lead to a less desirable increase in algal productivity. This may be due to the loss of negative feedback mechanisms that exist between aquatic plants and algae that are present in a lake system with high macrophyte biomass (Scheffer 1998). The establishment of a turbid phytoplankton-dominated system is not desirable as the consequences of continued absence of aquatic macrophytes from Lake Sorell is believed to significantly degrade the recreational trout fishery. This is because the invertebrates associated with the macrophyte beds constitute a significant dietary resource for the trout for most of the year (Stuart-Smith, Richardson et al. 2004). Aquatic macrophytes have also been found to provide significant habitat for both aquatic invertebrates and the golden galaxiid (Hardie 2003; Hardie, Barmuta et al. 2004).

In spite of the recent changes wrought by low water levels, there is little evidence of any major changes to the composition of the phytoplankton or zooplankton in either lake. The historical differences between these adjacent lakes (i.e. Cheng and Tyler's "limnological paradox") remains to be explained, and the data assembled here allows closer scrutiny of a potential mechanism: differences in the trophic dynamics between these two lakes.

It may be possible to infer that the processes that characterised phytoplankton community structure in both lakes in the past are still significant today, and have a strong influence on maintaining the distinction in community characteristics between the two lakes. There also lies the possibility that bottom-up control of primary productivity is responsible for the contrast in algal standing crop (Sondergaard, Jeppesen et al. 2000). Although extensive work by Cheng in lakes Sorell and Crescent on nutrient dynamics and primary productivity was unable to unequivocally account for the sustained order of magnitude difference in algal biomass (Cheng and Tyler 1973a; Cheng and Tyler 1976a). Cheng concluded lakes Sorell and Crescent to be somewhat of a 'limnological paradox'.

It is possible that the contrast in algal standing crop between lakes Crescent and Sorell is caused by differing top-down cascading trophic interactions (Carpenter, Kitchell et al. 1987) with the higher trout concentration in Lake Sorell driving a top-down trophic cascade. This difference in piscivore biomass may cascade down through each trophic level to ultimately influence primary productivity (Carpenter, Kitchell et al. 1985; Carpenter, Kitchell et al. 1987; McQueen, Johannes et al. 1989) and may possibly drive the contrasting alternate stable-states shown by these two lakes (Scheffer, Hosper et al. 1993).

#### ***5.4.1 The likely effects of piscivores and zooplanktivores***

The management aims for the recreational trout fishery of both lakes differs significantly. Lake Sorell has been managed as a high output fishery aimed at returning the maximum number of fish to the recreational angler. Trout recruitment in Lake Sorell is managed to maintain a large number of fish in the system. By contrast, Lake Crescent has been managed as a 'trophy trout water' with a low abundance of trout reducing competition for food resources and resulting in much larger fish. Therefore, partly through active management practices, Lake Sorell typically has a trout biomass ten to twenty times that of Lake Crescent.

Comparisons of golden galaxiid densities and biomass to zooplanktivore biomasses of other studies show Lake Sorell to be low whilst galaxiids in Lake Crescent approach biomasses that may have the capacity to limit cladoceran abundance. Limnocorral studies by McQueen and Post (1988) found a strong correlation between the collapse of the *Daphnia* population and increasing biomasses of planktivorous yellow perch when biomasses reached levels of 30 to 50 kg ha<sup>-1</sup>. These biomasses are still high when compared to Lake Crescent, that had an estimated golden galaxiid biomass of 8.0 kg ha<sup>-1</sup> during 2001 compared to lake Sorell, with an estimated biomass of 0.6 kg ha<sup>-1</sup> for the same time. The golden galaxiid abundance estimates of Lake Crescent approach levels that may begin to limit cladoceran biomass when compared to zooplanktivore abundance estimates of McQueen and Post (1988), with critical biomasses from their work being 3 to 4 × that of Lake Crescent and 50 to 80 × that of Lake Sorell.

Interestingly, Brooks (1968) noted that *Bosmina* appears as the dominant cladoceran along with a few small cyclopoid species in any pond or lake in North America or Eurasia where predation is intense. This is a comparable zooplankton community to that of Lake Crescent (Figure 70). Brooks (1968) goes on to discuss the preference of facultative plantkivores for *Daphnia* of over 1.3 mm long with the effects of these fish being strongest on the *Daphnia* populations.

Previous work has shown that the biomass of benthic invertebrates in shallow lakes is potentially greater than in deeper lakes, which means that benthic-planktivorous fish stocks are less dependent on zooplankton prey than in deep lakes (Jeppesen, Jensen et al. 1997). The ability of benthic-planktivorous fish, such as golden galaxias, to shift from zooplanktonic to benthic prey allows their density to remain high even during periods of limited zooplankton. They thereby maintain a high potential predation pressure on zooplankton indefinitely due to their ability to change foraging strategies (Jeppesen, Jensen et al. 1997). As a result, the golden galaxiids do not necessarily die off after the collapse of *Daphnia* and hence the “overexploited” zooplankton state may be persistent rather than a transient phase (Scheffer 1999).

The effect of fish predation on *Daphnia* and other plankton is a major focus of current research in lake ecology (Scheffer 1999). Experiments that explore how *Daphnia* populations are affected by changes in fish predation pressure show fish have little effect on *Daphnia* density until a certain critical threshold is passed, at which time the *Daphnia* population crashes and algal blooms may develop (McQueen and Post 1988). Probably as a result of the difference in zooplanktivore predation pressure between lakes Crescent and Sorell, large bodied cladocerans are less prevalent in Lake Crescent (Hoffman, Smith et al. 2001), and this suggests that zooplankton grazing is unlikely to restrict phytoplankton productivity in this lake.

Trout, although selecting *Daphnia* as a food resource when *Daphnia* are abundant (Stuart-Smith 2001; Stuart-Smith, Richardson et al. 2004), would not have the potential to limit zooplankton biomass to the same extent that galaxiids do. This is due largely to the inability of trout to selectively target *Daphnia* when they are scarce, as demonstrated by Hardie’s (2003) comparative dietary studies. In contrast, the

galaxiids continue to graze heavily on *Daphnia*, even when *Daphnia* are in extremely low abundance (Section 5.3.3).

#### 5.4.2 Zooplankton grazing

Grazing zooplankton tend to be selective feeders (Lampert, Fleckner et al. 1986; Attayde and Hansson 2001). Therefore, if cladoceran grazing was important in reducing phytoplankton biomass in these lakes, it may be expected that certain groups of phytoplankton would disappear at times of high cladoceran biomass. The data collected during the course of this project, however, does not support this hypothesis, with little evidence of change in specific phytoplankton functional groups coinciding with increased cladoceran biomass. It is likely that the strong interaction between increased primary productivity and increased sediment resuspension acts to confound this association (Chapter 2; Chapter 3; and Chapter 5, Section 5.3.1).

Increased suspended sediment may itself adversely affect the biomass of cladocera (Arruda, Marzolf et al. 1983; Hart 1988), and this mechanism may have operated during the prolonged period of more frequent, severe resuspension events in these lakes in the late 1990s and early 2000s. Interestingly, the peaks in zooplankton biomass that occurred in Lake Sorell towards the end of 2001 coincided with the lowest recorded levels of suspended sediment for the study period. In the past, when the lakes were considerably clearer with low suspended sediment levels, zooplankton biomass may also have been higher.

Blindow et al. (2000) concluded in their study that where grazers took less than 10% of phytoplankton biomass per day, mechanisms other than zooplankton grazing were important in reducing algal standing crop. Comparison of average zooplankton to phytoplankton dry weight ratios in lakes Crescent and Sorell (Figure 74) show potential zooplankton grazing pressure to be well below this 10% threshold on all sample occasions in Lake Crescent and on all but 3 sample occasions in Lake Sorell. This coupled with the erratic occurrence of high numbers of *Daphnia* in both lakes (Figure 72), suggests that limitation of phytoplankton by zooplankton grazing is unlikely in either lake for any extended period of time. The increased zooplankton to phytoplankton ratios sporadically reached in Lake Sorell (Figure 74) of 40-80% of phytoplankton biomass potentially grazed per day, reflects zooplankton grazing in this

lake to have the potential to influence phytoplankton biomass. However, the infrequent occurrence that ratios approach these levels show zooplankton grazing to be unable to account for the low phytoplankton biomass in Lake Sorell and the significant contrast in algal standing crop between the lakes.

Further work examining the size classes of the dominant phytoplankton during periods of increased zooplankton biomass would be beneficial in assessing if zooplankton grazing influences phytoplankton as cladocerans tend to feed on a restricted range of sizes of phytoplankton with a threshold volume  $10^4 \mu\text{m}^3$  being a reasonable upper limit (Harris 1986). Zooplankton can therefore suppress the population of nanoplankton and leave the populations of larger plankton largely unaffected (Harris 1986).

#### **5.4.3 *Alternative “bottom-up” and “top-down” arguments***

An interesting alternative hypothesis potentially relevant to lakes Sorell and Crescent has been presented by McQueen et al. (1989) who offer a “bottom-up top-down” theory which, for eutrophic lakes, combines both the influence of top-order predators (top-down) and resource availability (bottom-up) in determining ‘maximum attainable biomass’. The theory predicts that top-down forces are strong at the top of the food chain but weaken towards the bottom (with little or no consequent long-term effects on phytoplankton biomass), whereas bottom-up forces are strong at the bottom and weaken towards the top. Although Lakes Crescent and Sorell are mesotrophic, they appear consistent with this model: the piscivores and zooplanktivores strongly affect their prey, but the effect of zooplanktonic grazers is unlikely to limit phytoplankton. Both these lakes have similar nutrient regimes, so any differences in bottom up effects that could contribute to the differences between the lakes result from other causes.

Work in Chapter 3 (Section 3.3.3, Figure 32) demonstrated the physical environments of the lakes differed substantially in that the morphology of Lake Crescent and past lake management practices leads to a higher susceptibility to sediment resuspension than Lake Sorell. Past studies have found significant interactions between sediment entrainment and changes to primary productivity through either direct resuspension of algae or indirectly through increased nitrogen and phosphorus loading on the water column (Hamilton and Mitchell 1988).

One obvious suggestion, that increased wind resuspension increases primary productivity through increasing nutrient concentrations (Demers, Therriault et al. 1987), is not supported by the evidence for these lakes. There was no significant differences between the lakes for TP ( $F_{(1,161)} = 0.20$ ,  $P = 0.66$ ) (Chapter 2, Figure 7), and, although Lake Crescent had statistically significantly higher concentrations of TN than Lake Sorell, the average annual difference was modest ( $\sim 0.73 \text{ mg L}^{-1}$ ) (Chapter 2, Figure 6), with nitrate concentrations in Sorell sometimes exceeding those in Lake Crescent (Chapter 2, Figure 4). It is unlikely that these differences between the lakes in nitrogen are large enough to explain the differences in algal productivity, especially in view of the fact that the evidence from nutrient ratios show that nitrogen is never limiting in these lakes.

The direct physical effects on phytoplankton of sediment resuspension can either be negative or positive. For example Hellstrom (1991) found that sediment resuspension reduced mean light intensity and limited phytoplankton productivity by up to 85% in a shallow lake in Sweden, whereas Ogilvie and Mitchell (1998) concluded that sediment resuspension may reduce photoinhibition of phytoplankton in shallow lake systems with high baseline transparency and therefore favour increased primary productivity. In the case of lakes Sorell and Crescent, the strong correlations between increased sediment resuspension and increased algal standing crop (Chapter 2, Section 2.3.3) would limit the applicability of Hellstrom's (1991) hypothesis in this instance. It may be that the situation described by Ogilvie and Mitchell (1998) is more relevant to lakes Sorell and Crescent.

A more likely mechanism prevailing in these lakes, and one which may explain the 'limnological paradox' is the role of meroplankton. Carrick et al. (1993), working on Lake Apopka, Florida, found that direct entrainment of meroplankton was highly significant in determining algal standing crop in this large shallow lake. They found that water column chlorophyll-*a* concentration was highly correlated with average daily wind speed, and that the algal community was largely dominated by diatoms, a characteristic that is shared with the Lake Crescent phytoplankton community.



Algal populations with a meroplanktonic phase are planktonic only during periods of relatively high wind-induced turbulence (Schelske 2002), and Lake Crescent is frequently dominated by meroplanktonic diatoms (Table 31), particularly *Diatoma elongatum* (Table 32), which is known to have resting cells (Schelske, Carrick et al. 1995) and is therefore adapted to spending extended periods of time out of the euphotic zone. Moreover, the empirical modelling presented in Chapter 3 shows that Lake Crescent is much more turbulent than Lake Sorell, and more prone to resuspension events. These two pieces of evidence combined strongly suggests resuspension dynamics in Lake Crescent underlies the greater prominence of phytoplankton in this lake compared with Lake Sorell.

In lakes with substantial meroplanktonic algal populations, pelagic algae can be maintained at high levels through entrainment of sediments, such as demonstrated by Schelske et al. (1995). The influence on algal standing crop of biotic top-down processes, such as grazing by zooplankton, and bottom-up processes, such as nutrient and light limitation, may be lessened (Schelske, Carrick et al. 1995), thus providing a mechanistic explanation for McQueen et al.'s (1989) model in which top-down and bottom-up processes get attenuated at more distant trophic levels. The dominance of meroplankton in Lake Crescent coupled with the weak zooplankton – phytoplankton interactions, and a temporally sustained heightened level of algal productivity during a period of extreme light attenuation (Chapter 2) are likely to be the result of such processes.

## 5.5 Conclusion

Both Lakes Crescent and Sorell appear to have strong top-down trophic interactions that decouple at the link between zooplankton and phytoplankton. Trout in both lakes prey on galaxiids, with the greater effect being expressed in Lake Sorell. In turn, the galaxiid abundance in both lakes is sufficient to influence the abundance and size structure of the zooplankton, with Hardie (2003) documenting both the ability of galaxiids to preferentially eat larger zooplankton, and to switch to alternative prey items (e.g. benthic and terrestrial invertebrates) when larger zooplankton become scarce.

Although galaxiids are more numerous in Lake Crescent, the resulting contrasts in zooplankton grazing pressure between the two lakes fails to account for the order of magnitude discrepancy in algal standing crop. The relatively infrequent occurrence of significant numbers of large bodied cladocerans in Lake Sorell and the sustained low ratio of cladoceran dry weight to algal dry weight, although reaching limits high enough to impact on phytoplankton standing crop at times, would not be capable of a sustained reduction in algal biomass leading to the order of magnitude reduction in algal biomass in Lake Sorell compared to Lake Crescent. Consequently, the trophic cascade in lakes Sorell and Crescent decouples at the interface between zooplankton and phytoplankton and that the increased productivity of Lake Crescent is likely due to factors independent of reduced zooplankton grazing pressure. Thus a top-down trophic cascade and stable-state theory does not explain Cheng and Tyler's "limnological paradox": zooplankton is not capable of limiting phytoplankton, while bottom-up factors such as nutrients and incident light are similar in both lakes.

Consequently, the historical differences between the two lakes must be due to something else, and the information assembled in this chapter implicates meroplankton as a likely cause. Meroplanktonic species depend on turbulence for resuspension: Lake Crescent has been historically dominated by meroplanktonic phytoplankton because it is the lake with the greater exposure to turbulent conditions, which in turn, have favoured meroplanktonic species. By contrast, the reduced dominance of diatoms and meroplanktonic forms of algae in Lake Sorell, combined with a historically lower incidence of sediment resuspension and clearer water has lead to a lake dominated by vascular plants and non-meroplanktonic phytoplankters.

Whether recorded increases in phytoplankton biomass that correlate closely with sediment resuspension events are related to new growth resulting from resuspended nutrients (Hamilton and Mitchell 1988) or to direct resuspension of benthic or meroplanktonic algae or resting propagules (Carrick, Aldridge et al. 1993) is hard to unequivocally determine. Increased frequency of sampling of phytoplankton before, during and after significant sediment resuspension events (Schelske 2002) coupled with detailed enumeration and identification of species composition would help in isolating the reason for the highly significant correlation between high tripton levels and increased algal biomass. Additionally, carrying out a thorough, detailed

experimental study as designed by Schallenberg and Burns (2004) aimed at teasing apart the specific effect sediment resuspension has on phytoplankton production, would be valuable in determining the degree of influence sediment resuspension plays in stimulating algal productivity in Lake Crescent.

Alternatively, applying a detailed modelling framework capable of simulating phytoplankton growth and resuspension, such as employed in Chapter 4 to investigate sediment resuspension dynamics between the lakes, would likely help in determining more precisely the reasoning for the ‘limnological paradox’ between lakes Crescent and Sorell as identified by Cheng and Tyler (1973a). This is what is attempted in Chapter 6 with DYRESM-CAEDYM with a focus on investigating the potential role meroplankton resuspension plays in controlling algal standing crop in lakes Sorell and Crescent.

## **Chapter 6 The limnological paradox of lakes Sorell and Crescent revisited. Meroplankton dominance and resuspension dynamics.**

### **6.1 Introduction**

Chapter 5 suggested that the reasons for the historical differences between Lake Sorell and Lake Crescent (i.e. Cheng and Tyler's "limnological paradox") were unlikely to be due to a top-down trophic cascade because the zooplankton are not capable of limiting phytoplankton for any significant period of time. Consequently, the historical differences between the two lakes must be due to something else, and the information assembled thus far implicates meroplankton as a likely cause.

Despite the major contrast in algal productivity and phytoplankton community structure that formed the basis for Cheng and Tyler's 'limnological paradox' (Cheng and Tyler 1973a), Cheng and Tyler (1973b) found the chemical environments of lakes Sorell and Crescent to be extremely similar with the exception of silica concentrations, with dissolved silica levels in Lake Sorell averaging around  $12.2 \text{ mg L}^{-1}$  compared to Lake Crescent with average concentrations of  $1.3 \text{ mg L}^{-1}$  (Cheng and Tyler 1973b). They concluded that this single large difference could be attributed to the increased biological utilisation of  $\text{SiO}_2$  by the greater diatom biomass that dominates the algal community of Lake Crescent.

Cheng and Tyler (1976a) hypothesised that the degree of turbulence and exposure experienced by the lakes differed on average, with Lake Crescent being more susceptible to sediment resuspension due to its typically shallower average depth. They used this to argue that the increased algal biomass of Lake Crescent was due to increased resuspension of tripton that stimulates algal productivity by increasing the availability of limiting nutrients such as phosphorus and nitrogen. This hypothesis was supported by laboratory bioassays that studied the effect on primary productivity of the independent addition of tripton from Lake Crescent and Lake Sorell to water samples from each lakes. They found that adding tripton from Lake Crescent significantly increased algal productivity, whereas the addition of tripton from Lake Sorell did not, and they attributed these results to the higher concentrations of  $\text{SiO}_2$  in the water from Lake Crescent (Cheng and Tyler 1976a).

Padisak et al. (2003) recognised the importance of turbulence in shallow polymictic lakes for increasing turbidity and making available to phytoplankton the higher nutrient content of pore water that would otherwise be released very slowly through diffusion (Bachmann, Hoyer et al. 2000). The mechanism proposed is the increased importance of sediment-water interactions in shallow lakes because of high sediment surface to water column ratios (Sondergaard, Jensen et al. 2003) that leads to an increased significance of nutrient cycling from sediments than found in deeper lakes. In deeper lakes that stratify, thermal stratification over summer reduces mixing between the hypolimnion and the epilimnion leading to reductions in nutrient availability in the epilimnion (Wetzel 1983) along with reducing the interaction between the epilimnion and the sediments significantly.

Numerous studies have found positive relationships between increased turbulence and resuspension events and increased algal biomass and productivity (Gabrielson and Lukatelich 1985; Padisak, G-Toth et al. 1990; Carrick, Aldridge et al. 1993; de Jonge and van Beusekom 1995; Hamilton and Mitchell 1997). However, in many cases, the exact mechanism leading to increased productivity is hard to establish (Schallenberg and Burns 2004) with increases in algal biomass possibly being related to increased nutrient concentrations or the result of direct resuspension of benthic or meroplanktonic algae that are present on the sediment surface (Bachmann, Hoyer et al. 2000).

The resting cells and spores of meroplankton settle or sink to the bottom sediments and spend a portion of their lifecycle in the benthos (Schelske, Carrick et al. 1995). Meroplankton are often dominated by diatoms (Bacillariophyta) that are capable of withstanding extended periods of darkness (Carrick, Aldridge et al. 1993) and have the ability to produce resting cells or spores which are able to survive for long periods under conditions not conducive to growth (Reynolds 1984). For example, resting cells of *Aulacoseira* (formerly *Melosira*) *granulata* are able to germinate and fix carbon photosynthetically within a few hours of being exposed to moderate levels of light having spent approximately 20 y in anoxic sediments (Sicko-Goad, Stormer et al. 1986). Thus, resting propagules that are physiologically dormant in darkness have, on contact with light and nutrients in the water column, the ability to become active

within a few hours (Schelske, Carrick et al. 1995) thereby ensuring population survival until resuspension returns the cells to the water column and a planktonic environment conducive to growth (Schelske 2002).

Originally Cheng and Tyler (1976a) made little mention of the possibility of direct inoculation of the water column by resuspended meroplanktonic algae as being important in increasing algal productivity in Lake Crescent. Their bioassay method (i.e. the transfer of water from one lake to the other in their laboratory tripton enrichment experiment) would have transferred meroplanktonic propagules. I contend that the differences in the responses in these two lakes to wind-driven resuspension events is sufficient to explain the historical differences in algal communities between these two lakes.

Evidence thus far (Chapters 3 – 5) supports the hypothesis that sediment and, therefore meroplankton, resuspension is a potential driver for increased phytoplankton productivity in Lake Crescent. It appears that the physical environment of Lake Crescent differs fundamentally from that of Lake Sorell (Chapter 3) and provides a more favourable environment for a meroplanktonic phytoplankton community to establish and proliferate.

To test this hypothesis, DYRESM-CAEDYM was used to model sediment resuspension and algal dynamics in both lakes as a “bottom-up” system, i.e. ignoring any role that zooplankton grazing or higher trophic levels have on the biomass of the various algal taxa in the lakes. The success of this modelling was evaluated in three ways. Firstly, if the model is capable of predicting and producing acceptable correlations between modelled and observed field data for the period of the current study; secondly, whether differences in modelled phytoplankton biomass between the two lakes may be explained by different levels of turbulence engendered by wind; and thirdly, in modelling water levels typical of both lakes under ‘historic’ conditions prior to 1999, do contrasts in algal biomass still hold that may help explain Cheng and Tyler’s “limnological paradox”. Thus, also, if it is found that phytoplankton growth dynamics may be adequately modelled excluding zooplankton, then it would further strengthen the conclusions made in Chapter 5 that top-down trophic cascades do not limit phytoplankton productivity in either lakes Sorell or Crescent.

## 6.2 Methods

The application of DYRESM-CAEDYM to model SPM resuspension (Chapter 4) was used as the basis for more detailed modelling. Nutrient and light attenuation information from Chapters 2 and 3 and algal information from Chapter 5 was used to further calibrate and initialise DYRESM-CAEDYM.

To better understand the role differences in basin morphometry plays in influencing algal dynamics in Sorell and Crescent, DYRESM-CAEDYM calibration files between the two lakes were identical (except for the algal attenuation coefficient; see Section 6.2.2) so that any differences in model outcomes between lakes resulted from the difference in basin morphometries of the lakes. This was justified by the comparable water chemistry of the two lakes (Chapters 2 and 3) (Cheng and Tyler 1973a; Cheng and Tyler 1973b).

The effectiveness of DYRESM-CAEDYM to model changes in phytoplankton biomass and community composition, total phosphorus and total nitrogen, and light attenuation, was assessed by comparing model output to field observations by linear regression and visual analysis.

### 6.2.1 *Phytoplankton community composition*

From the information presented in Chapter 5 Section 5.3.1 it was determined that diatoms and chlorophytes (termed “green algae” and “green filaments” in Chapter 5) were the two dominant algal groups to be focussed upon within the DYRESM-CAEDYM modelling structure.

Quantitative cell counts of the dominant phytoplankton species present in Sorell and Crescent on the initial date of sampling (19 April 2000) were used to determine initial conditions for model runs. For details of the methods used to determine phytoplankton community composition refer to Chapter 5, Section 5.2.1.

### 6.2.2 *Calibration and running of DYRESM and CAEDYM*

Calibration of DYRESM was not necessary due to the process-based configuration of the model (Murphy 2001). As a result, CAEDYM was the only model component that required calibration.

CAEDYM modelling of sediment resuspension (Chapter 4, Section 4.3.2) was used as the foundation for a more detailed modelling scenario that included model output of light attenuation, total nitrogen, total phosphorus and phytoplankton. The successful calibration of CAEDYM to model sediment resuspension in lakes Sorell and Crescent (Chapter 4, Section 4.3.2) was closely followed and for each additional parameter modelled by CAEDYM, information and data for model calibration was approximated from the literature or calibrated directly from field data collected between April 2000 and December 2001.

Calibrating DYRESM-CAEDYM was carried out by iteratively adjusting input values of the variables described below until model output of water quality closely resembled that of field data. The methods used to initialise DYRESM-CAEDYM are detailed in Chapter 4, Section 4.2.4, with Section 4.2.5 detailing the initialisation for sediment resuspension. Specific attenuation coefficients for light attenuation for the inorganic suspended particulate matter (SPM) and algal components were determined from linear approximations of light attenuation from the multiple regression analysis presented in Chapter 3, Section 3.3.1, Table 13 and Table 14, and are further summarised in Table 35.

Table 35 DYRESM-CAEDYM calibration parameter values for specific attenuation coefficients for SPM and Chlorophyll-*a*.

Lake Sorell	1µm Inorganic SPM (mg L <sup>-1</sup> m <sup>-1</sup> )	0.080
	6µm Inorganic SPM (mg L <sup>-1</sup> m <sup>-1</sup> )	0.080
	Algal chlorophyll- <i>a</i> (µg L <sup>-1</sup> m <sup>-1</sup> )	0.016
Lake Crescent	1µm Inorganic SPM (mg L <sup>-1</sup> m <sup>-1</sup> )	0.099
	6µm Inorganic SPM (mg L <sup>-1</sup> m <sup>-1</sup> )	0.099
	Algal chlorophyll- <i>a</i> (µg L <sup>-1</sup> m <sup>-1</sup> )	0.002



The estimated specific attenuation coefficients for the SPM and algae for Lake Sorell and the SPM for Lake Crescent were comparable in magnitude to estimates from 4 independent studies described by Van Duin et al. (2001) for lakes from the northern hemisphere, but the estimated specific attenuation coefficient for algae in Lake Crescent was lower than those reported by Van Duin et al. (2001) by almost an order of magnitude on average. However, use of the published values from Van Duin et al. had little impact on the overall light attenuation, so this peculiarity of Lake Crescent had little effect on model performance.

Calibration of DYRESM-CAEDYM to model nutrient dynamics in lakes Sorell and Crescent followed closely the initialisation values used by Murphy (2001) for Lake Mokoan, Victoria, Australia, because this lake has important similarities with lakes Crescent and Sorell. It is shallow, fully mixed for the whole year, and the resuspended sediments are dominated by clays and other colloidal material. Thus Murphy's (2001) initialisation values were used for the following: aerobic and anaerobic mineralisation rates of phosphorus and nitrogen; sediment nitrogen and phosphorus flux; settling velocities for particulate nitrogen and phosphorus; and nitrogen nitrification and denitrification rates.

The validity of these initialisation values was assessed by examining modelled and observed values of the concentrations of total nitrogen, total phosphorus and the values of light attenuation for both lakes over the period April 2000 and December 2001. Time-series plots were examined to determine any periods of sustained under- or over-estimation by the model, while linear regression of observed values against modelled values was used to determine whether the fit deviated significantly from a 1:1 relationship, and to estimate the correlation between observed and modelled values. Standard regression diagnostics were inspected to detect violations of assumptions and potentially influential outliers, but no data were omitted from the final analyses presented here. Similarly, the data were mildly skewed and heteroscedastic; although diagnostics improved by log-transformation, the overall significance of the results remained unchanged, so coefficients from the untransformed data are presented here. Finally, differences in variation between modelled and estimated values of these three variables was tested using the Ansari-Bradley test, a non-parametric analogue of the  $F$ -test for comparing variances (Myles,

Hollander et al. 1973). If the model performed well, it would be expected that it would produce a similar “spread” in values to those observed, i.e. there would not be a significant difference in the Ansari-Bradley test. The Ansari-Bradley test was preferred over the *F*-test owing to its robustness to violations of normality.

In modelling the phytoplankton, the chlorophytes and diatoms were the two focal algal groups because of their central role in defining algal community characteristics in both Sorell and Crescent (Chapter 5, Section 5.2.1). Initial values for model runs of algal biomass represented as chlorophyll-*a* for both groups were approximated from the relative contribution made by each to quantitative cell counts (Chapter 5, Section 5.2.1). Based on the relative abundance and biovolumes (as described by Cheng and Tyler (1973a)) of each, estimates of initial chlorophyll-*a* concentration of each group was made and set in the CAEDYM initialisation file.

Phytoplankton resuspension dynamics were set to be comparable in magnitude to the resuspension coefficients for suspended solids. This was justified by the strong correlation between suspended solids and algal biomass (Chapter 2, Section 2.3.3, Table 6).

Chlorophyte and diatom cell diameters were set to 8  $\mu\text{m}$  and 13.5  $\mu\text{m}$  respectively, and was based on spheroidal diameter estimates made from cell volumes by Cheng and Tyler (1973a) of 105  $\mu\text{m}^3$  for their ‘minute green filament’ (probably *Ulothrix*) and 1284  $\mu\text{m}^3$  for *Diatoma*, the two dominant chlorophyte and diatom taxa.

A series of calibration runs were made using a variety of published and unpublished values of settling velocities, and the best settling velocity for each group selected by comparing modelled output with measured field data. A settling velocity of 0.17  $\text{m day}^{-1}$  for *Diatoma* was used and this was comparable to similar diatom groups documented in Reynolds (1984), and for the chlorophyte group, resulting in a settling velocity of approximately 0.05  $\text{m day}^{-1}$ . This is consistent with the extremely small volume and size of the minute green filamentous chlorophyte which should have a much slower settling velocity than *Diatoma*.

To effectively model meroplankton dynamics in Sorell and Crescent, the sediment survival time of the meroplanktonic dominated diatom group was set to approximately 120 days in both lakes. This was done to simulate the ability of meroplankton and meroplanktonic spores and resting cells to remain viable for extended periods of time whilst not in contact with the euphotic zone (Sicko-Goad, Stormer et al. 1986; Carrick, Aldridge et al. 1993; Schelske, Carrick et al. 1995; Schelske 2002). The suitability of including a meroplanktonic form of diatom in the modelling scenarios was assessed by comparing correlations between modelled and measured diatom biomasses with and without the inclusion of a meroplanktonic form.

The suitability of these calibration values for the phytoplankton was assessed by comparing modelled values of biomass (as chlorophyll-*a*) for the *Diatoma* and minute green filamentous algal groups with observed values (as cell counts) from the period March to December 2001, since this was the period for which all phytoplankton from entire water samples were enumerated, whereas data collected earlier only recorded net phytoplankton (bigger than approximately 45  $\mu\text{m}$ ). Scatterplots of observed and modelled values were inspected for any non-linear relationships, and, since there were none, the strength of the relationship is reported as Pearson's correlation coefficient, *r*. (Regression slopes are inappropriate here owing to the differences in units between modelled and measured values.) Time series plots superimposing observed cell counts and modelled estimates of chlorophyll-*a* for the two algal groups were also inspected to determine whether the modelled values captured the gross fluctuations shown by the measured data.

As a final assessment of the model initialisation, paired *t*-tests were used to assess whether modelled values of total chlorophyll-*a* differed from measured values in each lake, and Ansari-Bradley tests assessed if there were significant differences in the spread of values. Boxplots were also inspected to determine whether modelled output replicated the observed order of magnitude difference in chlorophyll-*a* between the two lakes.

### 6.3 Results

To determine the capacity of DYRESM-CAEDYM to model nutrient, light and algal dynamics in the lakes, model output was compared to measured field data by linear regression analysis.

#### 6.3.1 Nutrients and light attenuation

Owing to equipment failure, there was one less estimate of total nitrogen and total phosphorus concentration from Lake Crescent, and two fewer observations of light attenuation from Lake Sorell. As noted in Chapter 3, Section 3.2.1, Table 12, the light meter was unavailable over the period 29<sup>th</sup> November 2000 to 30<sup>th</sup> of August 2001.

Table 36 Results of tests of concordance between modelled and measured values of total phosphorus (TP), total nitrogen (TN) and light attenuation in lakes Crescent and Sorell for the period April 2000 to December 2001. The Pearson correlation coefficient,  $r$ , was significant ( $P < 0.01$ ) for all variables. The slope of the regression ( $\pm 1$  standard error) of measured values v. modelled values is reported along the  $t$ -value of the test for the slope deviating from 1; all slopes did not differ ( $P > 0.1$ ) from 1 except for \* ( $0.01 < P < 0.05$ ) and \*\* ( $0.001 < P < 0.01$ ). The value of the Ansari-Bradley test statistic(AB), which tests for equality of dispersion, is given with  $P$  values; a non-significant result supports the null hypothesis of no difference between dispersions.

	TP		TN		Light attenuation	
	Crescent	Sorell	Crescent	Sorell	Crescent	Sorell
$r$	0.587	0.650	0.513	0.808	0.804	0.677
Slope	0.452 $\pm(0.118)$	0.971 $\pm(0.211)$	0.594 $\pm(0.188)$	1.284 $\pm(0.174)$	0.881 $\pm(0.158)$	0.708 $\pm(0.199)$
$t$ value	4.65**	0.14	2.16*	1.64	0.75	1.47
n	30	31	30	31	19	17
Ansari-Bradley test	AB = 525 $P = 0.08$	AB = 465 $P = 0.38$	AB = 472 $P = 0.84$	AB = 405 $P = 0.01$	AB = 206 $P = 0.35$	AB = 156 $P = 0.84$

Inspection of the time series plots of modelled and measured values of the nutrients showed that the modelled values were generally consistent with the measured values in Lake Sorell (Figure 76); they were moderately to strongly correlated, and the regression slopes did not differ significantly from 1:1, although modelled values tended to overestimate measured values after June 2001, when water levels started to

rise (Table 36). Modelled values of nutrients tended to be less well correlated in Lake Crescent, and the time series plots showed that phosphorus was consistently overestimated from May 2000 to October 2000, whereas total nitrogen tended to be overestimated by the model later in the sequence from March 2001 (Figure 77). Some extreme model overestimates for total phosphorus occurred during the record low lake water levels encountered during January and February of 2001, and this coincided with extremes in modelled SPM resuspension (Chapter 4, Section 4.3.2.3, Figure 58). Overall, modelled values for light attenuation correlated well with measured values for both lakes, although the lack of availability of the light meter during the extreme low water levels of the summer of 2000/2001 was unfortunate.

To determine whether resuspension rate could have been responsible for the patterns in modelled values of phosphorus in Lake Crescent, the resuspension rate constant for phosphorus was halved (from  $9 \times 10^{-6} \text{ g m}^{-2} \text{ s}^{-1}$  to  $4.5 \times 10^{-6} \text{ g m}^{-2} \text{ s}^{-1}$ ). Although this reduced the overestimation of total phosphorus, concentrations of soluble reactive phosphorus remained low ( $0.00061 \text{ mg L}^{-1}$  vs  $0.00075 \text{ mg L}^{-1}$  for the higher resuspension rate) and the manipulations of the rate constant, importantly, had no measurable impact on the modelled algal productivity or species assemblage in the lake. Therefore, in order to keep the initialisation files as similar as possible to Lake Sorell, the resuspension rate constant for total phosphorous was returned to its initial value and remained unchanged in subsequent modelling.

Overall, the modelled values for nutrients and light attenuation reproduced the differences between and within the two lakes over time. Lake Crescent had slightly higher concentrations of total nitrogen than Lake Sorell, and greater light attenuation, with light attenuation in both lakes deteriorating substantially during the low water

phase. In addition, as judged from the Ansari-Bradley tests, the spread (i.e. variance) of modelled values was similar to the measured values for all these variables except for total phosphorus in Lake Sorell (where the modelled values were less variable than the measured values). Thus it was deemed reasonable to use Murphy's (2001) calibration values to model nutrients and light attenuation in lakes Crescent and Sorell.

Figure 76 Comparisons of modelled and measured total phosphorus, total nitrogen and light attenuation values for Lake Sorell, April 2000 to December 2001.

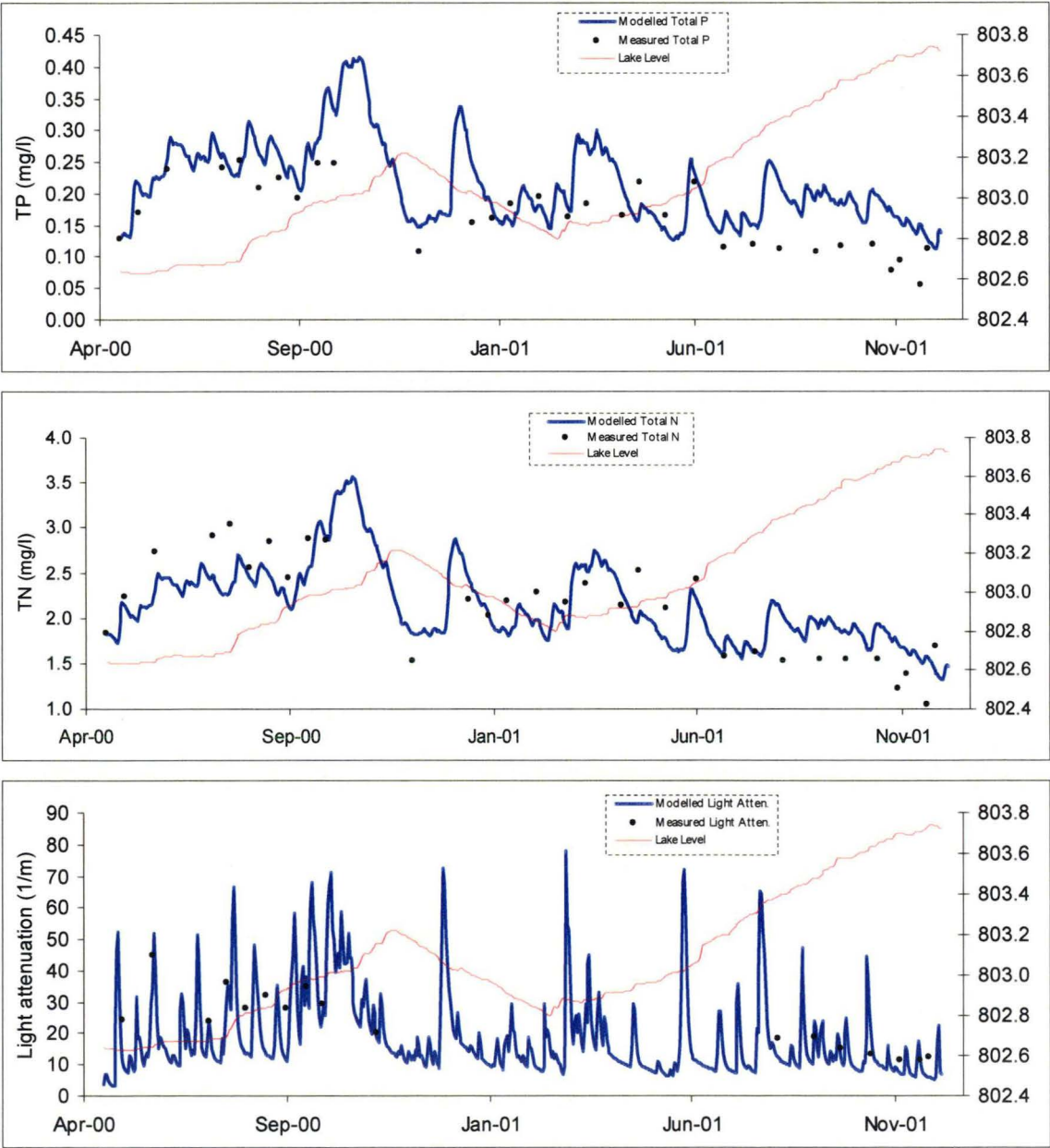
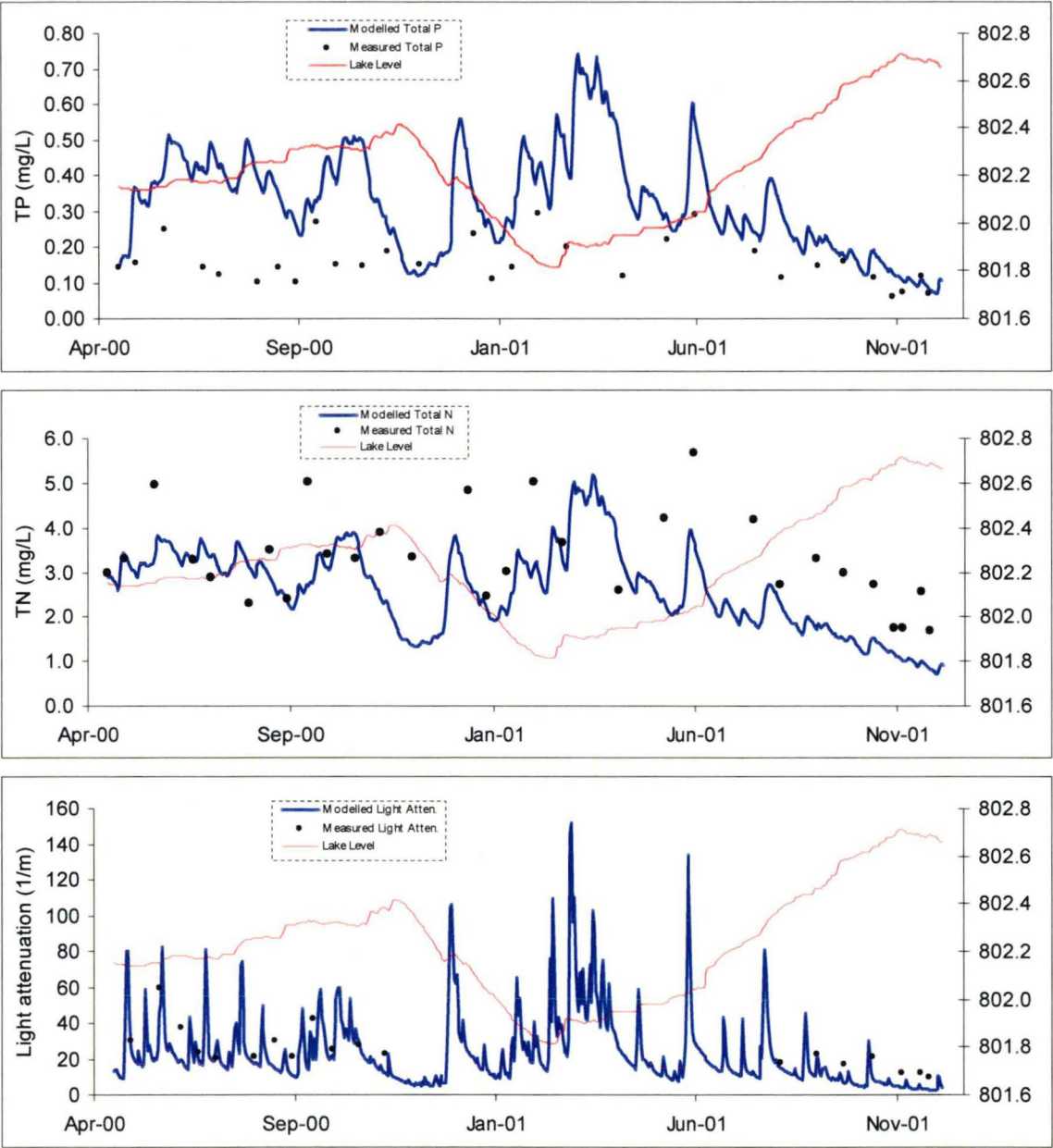


Figure 77 Comparisons of modelled and measured total phosphorus, total nitrogen and light attenuation values for Lake Crescent, April 2000 to December 2001.



### 6.3.2 *Phytoplankton*

Correlations of modelled diatom and chlorophyte biomass to measured cell counts made between February 2001 and December 2001, a period of reliable estimates of *Diatoma* and the minute green filamentous chlorophyte, are presented in Figure 78 through to Figure 81. Comparisons of correlations between modelled and observed data vary significantly between groups within each lake and between lakes (Table 37).

For Lake Crescent, modelled diatom biomass closely followed temporal trends in changes in *Diatoma* cell counts, with modelled output explaining approximately 85% of the variation in measured field data. This shows DYRESM-CAEDYM to be performing well when modelling diatom dynamics in Lake Crescent. The correlations between modelled chlorophyte biomass and observed cell counts of the minute green filamentous was lower (56%) but still highly significant.

For Lake Sorell, correlations of modelled diatom biomass to measured *Diatoma* cell counts was poor (Figure 80) with no significant relationship between the two (Table 37). This compares to the highly significant correlation between modelled chlorophyte biomass and 'minute green filament' cell counts for the same period (Figure 81), with modelled chlorophyte biomass explaining around 73% of the variation in measured field data.

The inclusion of meroplankton was assessed by comparing model runs with meroplankton excluded (diatoms with a 2 day sediment survival time) and meroplankton included (diatoms with a 120 day sediment survival time) with measured field data. Excluding meroplankton from the simulations reduced diatom productivity in Lake Crescent by approximately an order of magnitude and dropped correlations between modelled and observed data considerably. The inclusion of meroplankton in the modelling framework was therefore necessary to adequately simulate plankton dynamics in the lakes.



Figure 78     *Diatoma* cell counts versus modelled diatom biomass in Lake Crescent, March 2001 to December 2001.

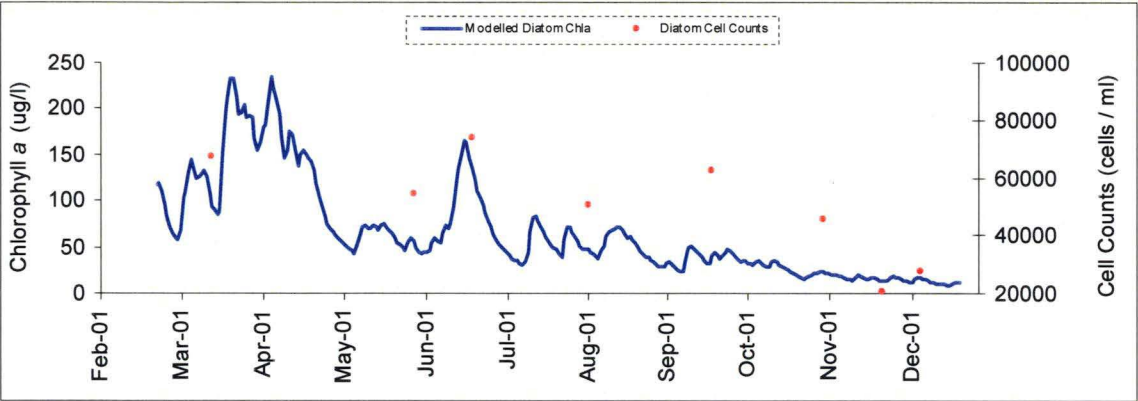


Figure 79     ‘Minute green filament’ cell counts versus modelled chlorophyte biomass in Lake Crescent, March 2001 to December 2001.

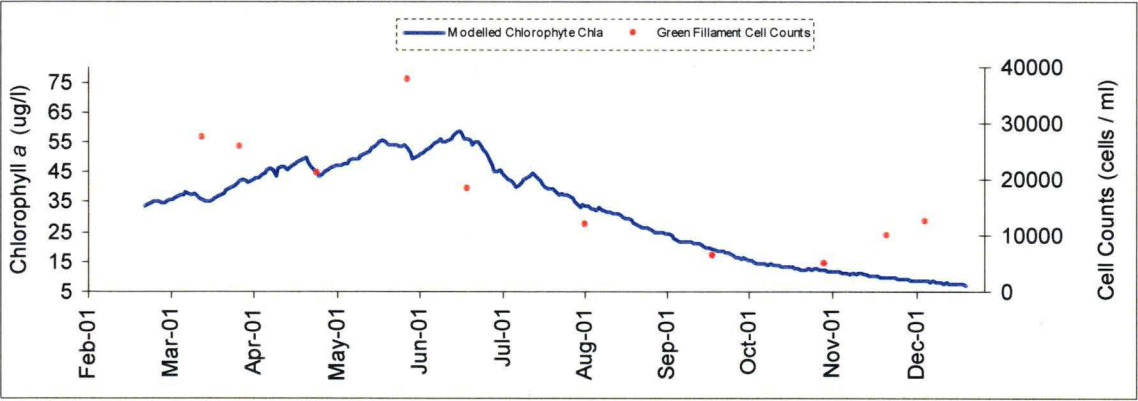


Figure 80     *Diatoma* cell counts versus modelled diatom biomass in Lake Sorell, March 2001 to December 2001.

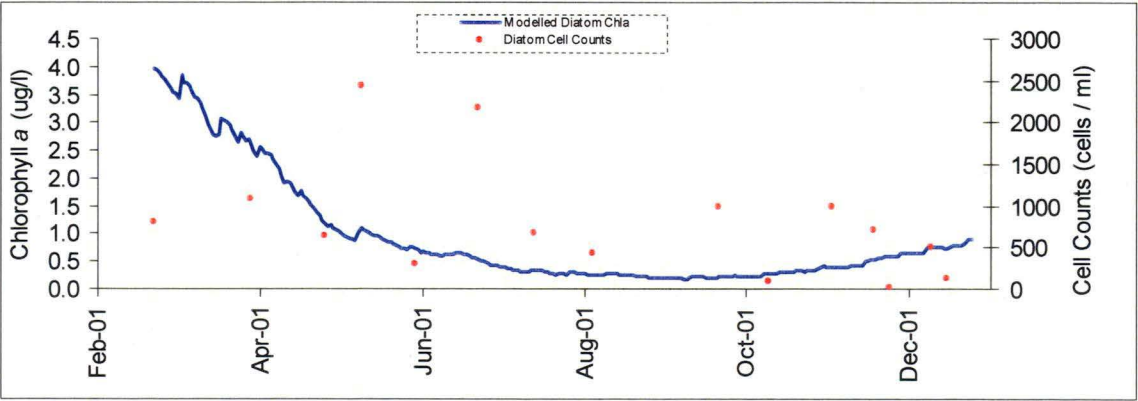
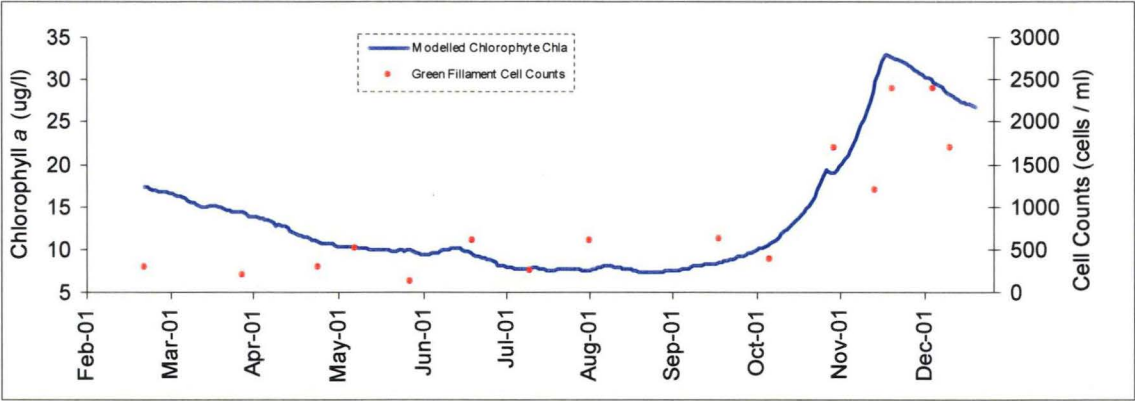


Figure 81 ‘Minute green filament’ cell counts versus modelled chlorophyte biomass in Lake Sorell, March 2001 to December 2001.



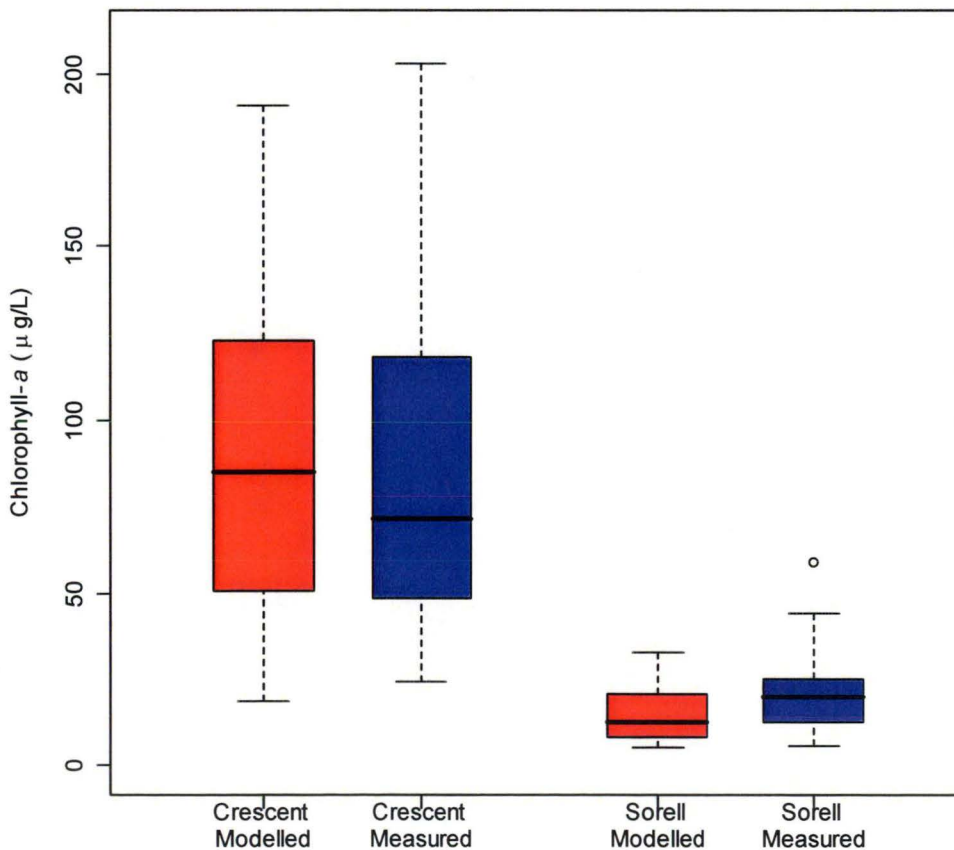
Except for the *Diatoma* group in Lake Sorell, model outputs correlated strongly and significantly with observed cell counts in both lakes (Table 37), and the variances of modelled and measured values were similar in both lakes. The time-series plots showed a good correspondence between changes in modelled biomass and observed cell counts, again with the exception of the *Diatoma* group in Lake Sorell. The poor correlation for *Diatoma* in Lake Sorell may also be influenced by low modelled biomass estimates and relatively low measured cell counts that may exacerbate errors in the data.

Table 37 Results of tests of concordance between modelled values of biomass and cell counts for the *Diatoma* and minute filamentous chlorophyte groups of algae in Lakes Crescent and Sorell. Pearson’s correlation coefficient and its *P*-value are presented with sample sizes, *n*, and the value of the Ansari-Bradley test statistic(AB), which tests for equality of dispersion, is given with *P* values; a non-significant result supports the null hypothesis of no difference between dispersions.

	<i>Diatoma</i>		Minute filamentous chlorophytes	
	Crescent	Sorell	Crescent	Sorell
<i>r</i>	0.928 (0.679, 0.984)	0.134 (-0.406, 0.604)	0.751 (0.231, 0.938)	0.856 (0.612, 0.951)
<i>p</i> value	0.0003	0.633	0.012	<0.0001
<i>n</i>	10	15	10	15
Ansari-Bradley test	AB = 65 <i>P</i> = 0.55	AB = 120 <i>P</i> > 0.9	AB = 66 <i>P</i> = 0.77	AB = 120 <i>P</i> > 0.9

Comparisons of the spread of modelled total chlorophyll-*a* values for the period from April 2000 to December 2001 compared closely within each lake (Ansari-Bradley tests both  $P > 0.8$ ) (Figure 82), nor was there any significant difference between the mean and measured values for Lake Crescent ( $t_{(29)} = 0.0712$ ,  $P = 0.943$ ) or Lake Sorell ( $t_{(31)} = 1.86$   $P = 0.072$ , ). Modelled values of chlorophyll-*a* in Lake Crescent were also about an order of magnitude greater than those in Lake Sorell. This shows the biomass estimates made by DYRESM-CAEDYM to be realistic and to fall within the range of measured field data. This coupled with the close temporal associations with cell counts of the dominant phytoplankton groups (Figure 78 to Figure 81) shows DYRESM-CAEDYM to perform satisfactorily in modelling temporal changes in the key phytoplankton groups of interest in both lakes.

Figure 82 Box-plot summary of measured and modelled total chlorophyll-*a* ( $\mu\text{g/L}$ ) for Lake Crescent and Lake Sorell, April 2000 to December 2001.





6.3.3 *Influence of sediment resuspension on driving algal productivity in lakes*  
*Sorell and Crescent*

To investigate how sediment resuspension potentially influences algal biomass in each of lakes Crescent and Sorell, plots of modelled algal biomass (as chlorophyll-*a*) and plots of modelled 1µm inorganic SPM were compared. Lake Crescent (Figure 83) showed a highly significant relationship between increasing SPM and increasing algal biomass, with concentrations of chlorophyll-*a* oscillating synchronously with changes in SPM. This demonstrates that algal productivity is intimately linked with sediment resuspension in this lake, whereas no such relationship was evident in Lake Sorell (Figure 84), which had no significant relationship between modelled algal chlorophyll-*a* and modelled SPM, with the peaks in chlorophyll-*a* being largely out of phase with peaks in SPM.

Figure 83 Comparisons of modelled 1 µm SPM concentration (mg/L) and modelled algal biomass (as chlorophyll-*a*, µg/L) for Lake Crescent, April 2000 to December 2001.

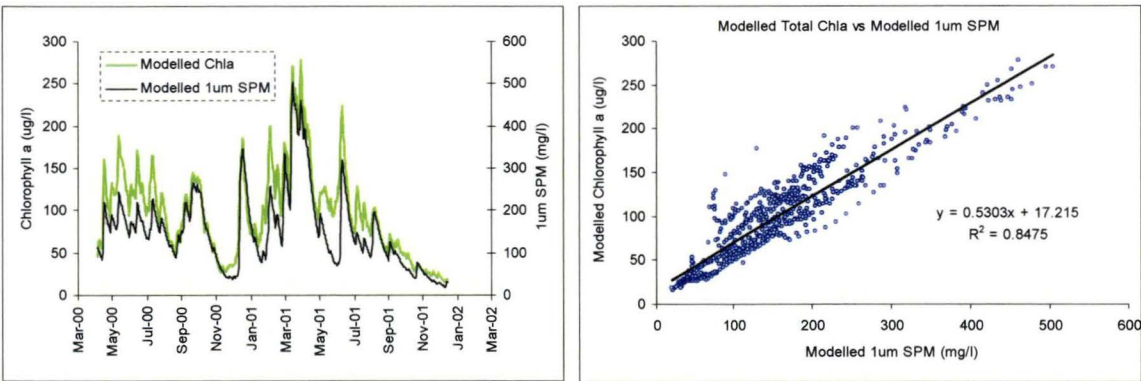
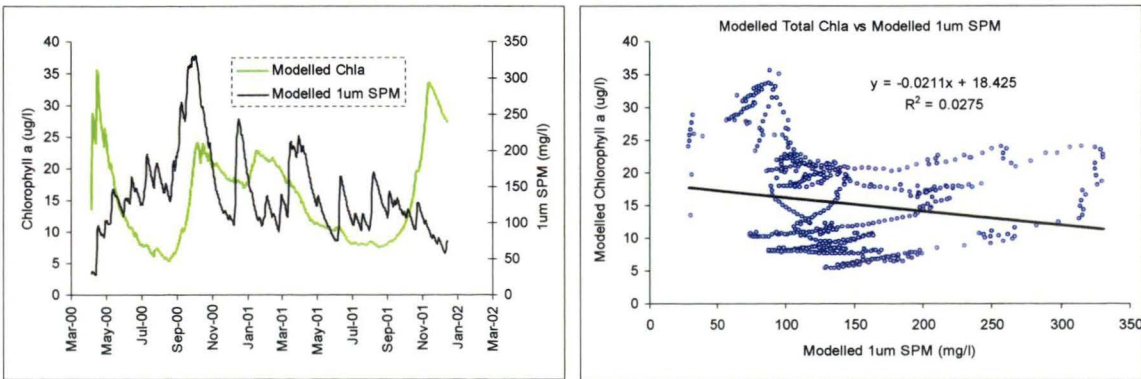
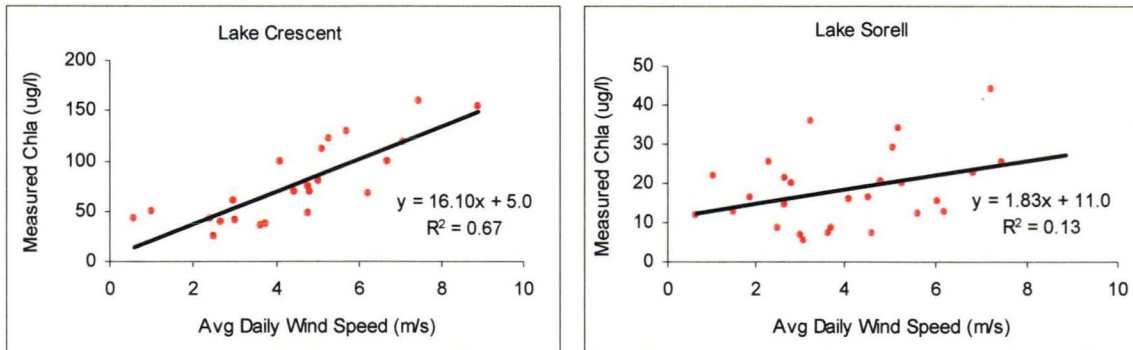


Figure 84 Comparisons of modelled 1 µm SPM concentration (mg/L) and modelled algal biomass (as chlorophyll-*a*, µg/L) for Lake Sorell, April 2000 to December 2001.



The correlation between wind speed (estimated on the day of sampling) and lake-averaged chlorophyll-*a* is summarised in Figure 85. The data includes measurements made between August 2000 and February 2002, which is the time period that wind speed measurements were available from the weather station located in the middle of Lake Sorell (Chapter 4, Section 4.2.4). Figure 85 further highlights the increased role sediment resuspension plays in driving algal productivity in Lake Crescent as demonstrated in Figure 84, with a strong, highly significant relationship between average daily wind speed and lake averaged chlorophyll-*a* in Lake Crescent ( $r^2 = 0.67$ ,  $df = 1$ ,  $t = 6.52$ ,  $P < 0.0001$ ,  $n = 23$ ) compared to the poor relationship in Lake Sorell ( $r^2 = 0.14$ ,  $df = 1$ ,  $t = 1.86$ ,  $P = 0.077$ ,  $n = 23$ ). Interestingly, analysis of the relationship between average daily wind speed and phaeophytin concentration returns an insignificant result for Lake Sorell ( $r^2 = 0.03$ ,  $df = 1$ ,  $t = 1.32$ ,  $P = 0.201$ ,  $n = 23$ ) and a slightly significant result for Lake Crescent ( $r^2 = 0.16$ ,  $df = 1$ ,  $t = 2.26$ ,  $P = 0.035$ ,  $n = 23$ ), thus reflecting a poor relationship between phaeophytin concentration and sediment resuspension in either lake.

Figure 85 Comparisons of lake averaged chlorophyll-*a*,  $\mu\text{g/L}$  versus average daily wind speed (m/s) for lakes Crescent and Sorell, August 2000 to February 2002.



#### 6.3.4 Historic modelled nutrient dynamics in lakes Sorell and Crescent

Comparisons of concentrations of modelled total phosphorus and total nitrogen in the two lakes during the current period of study ('actual' conditions: from April 2000 to December 2001), to those of total phosphorus and total nitrogen concentration under 'historic' lake level conditions (predating 1999) are presented in Figure 86 and Figure 87. For both lakes, the low lake levels typical of the period April 2000 to December



2001 caused a considerable increase in nutrient concentrations, both on average and in absolute magnitude (as realised in field measurements: Chapter 2), over the modelled estimates for increased lake levels as typical of the time period predating 1999. This strongly implicates low lake levels as being responsible for the increased eutrophication of both lakes.

Figure 86      Modelled total phosphorus and total nitrogen concentrations in Lake Sorell. – ‘Actual’ conditions (as present for the period running April 2000 to December 2001); and ‘Historic’ conditions, reflecting average lake levels typical of the period predating 1999.

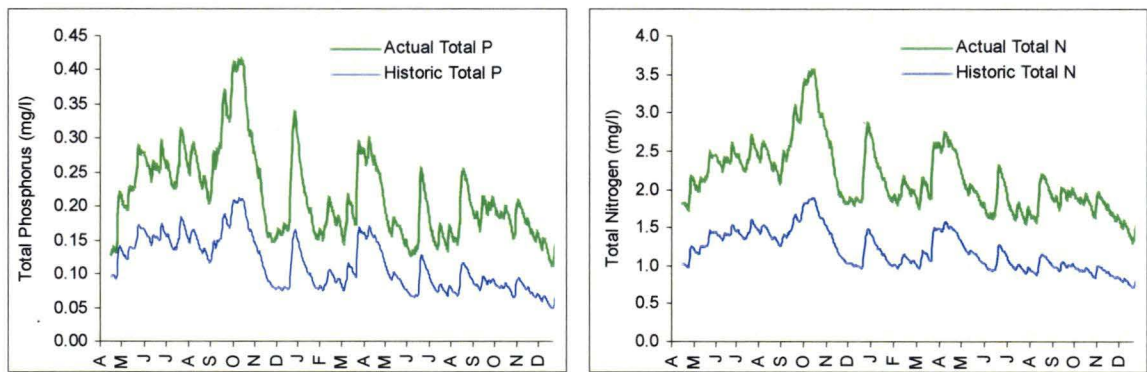
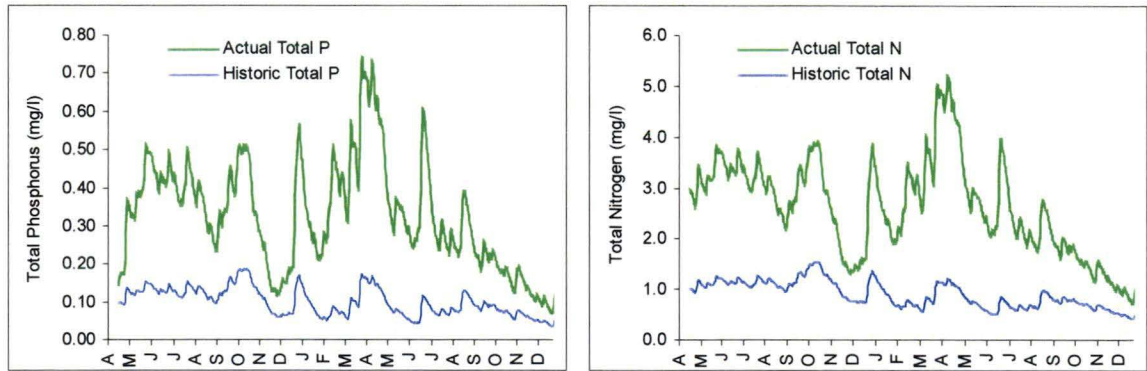


Figure 87      Modelled total phosphorus and total nitrogen concentrations in Lake Crescent. – ‘Actual’ conditions (as present for the period running April 2000 to December 2001); and ‘Historic’ conditions, reflecting average lake levels typical of the period predating 1999.



6.3.5    *Historic modelled light attenuation in lakes Sorell and Crescent*

Comparisons of changes in modelled light attenuation in Lake Sorell (Figure 88) and Lake Crescent (Figure 89) during the current period of study (‘actual’ conditions), running April 2000 to December 2001, to that of modelled light attenuation levels

under ‘historic’ lake level conditions (predating 1999) are presented below. For both lakes, the low lake levels typical of the period running April 2000 to December 2001 caused light attenuation levels to increase considerably, both on average and in absolute magnitude. An increase in lake level, typical of both lakes for times predating 1999, results in a major reduction in light attenuation levels in both lakes.

Figure 88      Modelled light attenuation in Lake Sorell. – ‘Actual’ conditions (as present for the period running April 2000 to December 2001); and ‘Historic’ conditions, reflecting average lake levels typical of the period predating 1999.

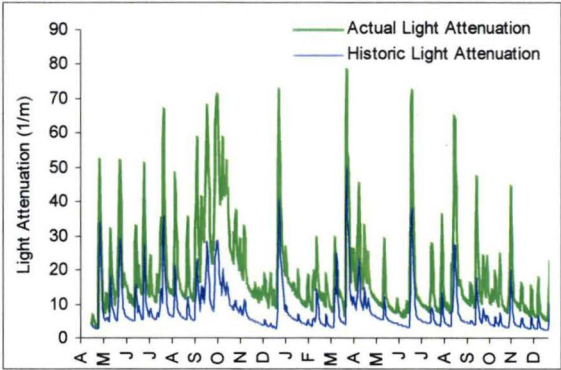
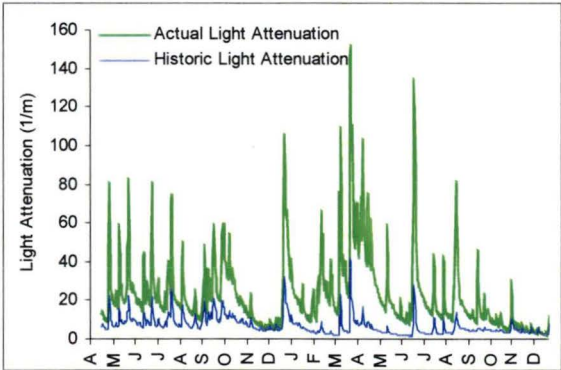


Figure 89      Modelled light attenuation in Lake Crescent. – ‘Actual’ conditions (as present for the period running April 2000 to December 2001); and ‘Historic’ conditions, reflecting average lake levels typical of the period predating 1999.



The conclusion made in Chapters 2 and 3 that low lake levels are the contributing factor leading to a severe increase in the levels of light attenuation in both lakes is further strengthened by the results of the DYRESM-CAEDYM modelling. An increase in lake levels to a ‘historic’ point typical of both lakes for the period predating 1999 results in a considerable decrease in light attenuation in both lakes, with light attenuation values dropping from  $23.4\text{ m}^{-1}$  (*St. Dev* = 20.7, *N* = 620) on average in Lake Crescent to  $7.0\text{ m}^{-1}$  (*St. Dev* = 4.9, *N* = 620), and average light

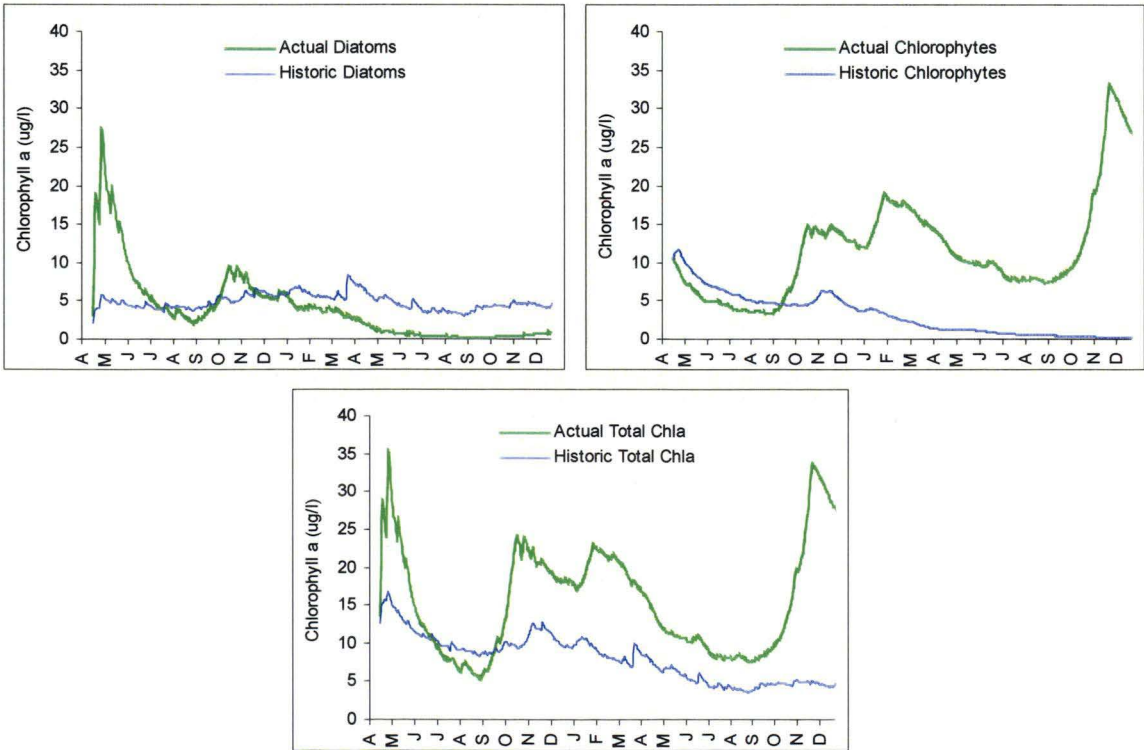
attenuation levels in lake Sorell dropping from 18.9 m<sup>-1</sup> (*St. Dev* = 13.6, *N* = 620) to 8.3 m<sup>-1</sup> (*St. Dev* = 6.6, *N* = 620).

6.3.6 *Historic modelled productivity in lakes Sorell and Crescent*

Model runs comparing changes in diatom and chlorophyte biomass under ‘historic’ and ‘actual’ conditions in Lake Sorell (Figure 90) and Lake Crescent (Figure 91) were carried out to compare the possible changes in phytoplankton biomass and species composition that might occur if lake levels were returned to conditions typical of the period predating 1999.

In the case of Lake Sorell (Figure 90), modelled chlorophyte dominance is reduced under historic conditions and the phytoplankton community is made up of an equal mix of both chlorophytes and diatoms for approximately the first half of the model run, after which diatoms out compete the chlorophytes and become the dominant plankton group. As was evident with nutrients and light attenuation, the reduction in the water level realised by the actual conditions degraded water quality: the ‘actual’ conditions stimulated algal productivity, leading to higher concentrations of total chlorophyll-*a* than under ‘historic’ conditions.

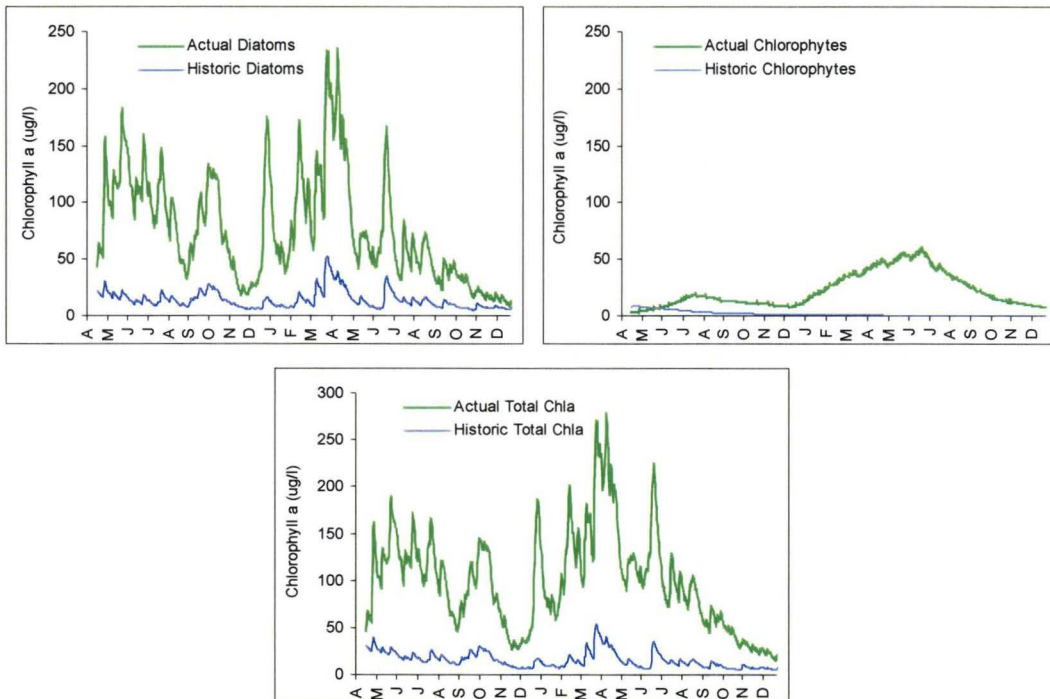
Figure 90 Comparisons of modelled plankton biomass in Lake Sorell under ‘actual’ conditions (as measured between April 2000 and December 2001) and ‘historic’ conditions.





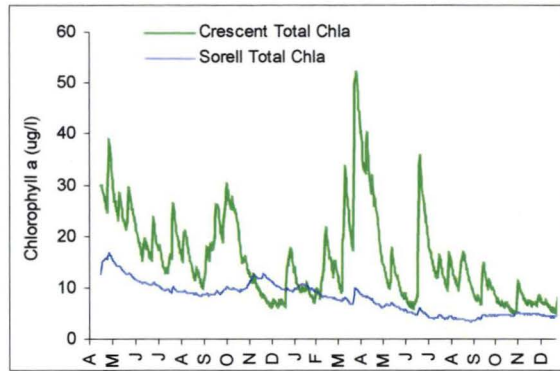
In the case of Lake Crescent, diatom dominance remains under ‘historic’ conditions and chlorophytes are all but lost from the system (Figure 91). As was evident with nutrients and light attenuation, comparisons of total chlorophyll-*a* between ‘historic’ and ‘actual’ conditions shows that reduced water levels lead to dramatically stimulated algal productivity in Lake Crescent.

Figure 91 Comparisons of modelled plankton biomass in Lake Crescent under ‘actual’ conditions (as measured between April 2000 and December 2001) and approximate ‘historic’ conditions.



Comparisons between lakes Sorell and Crescent of modelled total chlorophyll-*a* under ‘historic’ conditions shows phytoplankton productivity remained much higher in Lake Crescent than in Lake Sorell, which is a pattern that conforms with the recorded empirical record. Temporal changes in plankton biomass in Lake Crescent exhibited marked high amplitude fluctuations that would be due to the strong interaction with sediment resuspension that is characteristic of this lake. Interestingly, despite a substantial occurrence of diatoms in Lake Sorell, there is much less fluctuation in total chlorophyll-*a* under ‘historic’ conditions, further highlighting the contrast in the interaction of resuspension and algal productivity between the two lakes.

Figure 92 Comparisons of modelled total chlorophyll-*a* under ‘historic’ conditions in lakes Crescent and Sorell and Crescent.



## 6.4 Discussion

For both lakes, model output of key water quality variables compared well with measured field observations, with modelled outputs being poorest for total phosphorus and nitrogen concentrations in Lake Crescent. Overall, however, DYRESM-CAEDYM gave accurate predictions for light attenuation and SPM, and reasonable predictions for total phosphorus and total nitrogen, in that differences and similarities between the lakes were reproduced and the variation in modelled and estimated values was similar for most variables. This model thus shows promise in forecasting likely changes in key water quality variables such as light attenuation and SPM under differing scenarios for managing the water levels in these lakes, but would benefit from further calibration if accurate estimates of nutrient concentration was required.

The correlations between modelled phytoplankton biomass and measured cell counts for the two dominant algal taxa were highest for Lake Crescent and for the chlorophytes in Lake Sorell. The correlation was poorest (indeed, statistically insignificant) for the *Diatoma* group in Lake Sorell. This pattern might be expected if the postulated role of meroplanktonic diatoms is tightly linked to resuspension. Algal productivity is clearly closely linked with SPM concentrations and wind speed in Lake Crescent, whereas there is little correlation with these variables in Lake Sorell. Thus the phytoplankton dynamics in Lake Sorell appears much less linked to sediment resuspension which could result in the poor correlation between observed and modelled *Diatoma* values in this lake. To improve the performance of *Diatoma*

estimates for this lake would likely require further calibration or even invoking other mechanisms, but this is beyond the scope of the current study.

For Lake Sorell a more important correlation is that between modelled and observed chlorophytes values, because this group dominates the algal community in this lake. For example, based on cell counts (Chapter 5), diatoms contribute ~ 14% on average to the total phytoplankton cell count in lake Sorell compared to chlorophytes that contribute ~ 86%. This contrasts with Lake Crescent in which diatoms contribute ~71% on average to total phytoplankton cell counts while chlorophytes contribute ~29% .

The current modelling scenarios used identical calibration input files, with the exception of specific attenuation coefficients for SPM and algae, to model both freshwater diatoms and chlorophytes. This was justified as the dominant species within both lakes are comparable and the main aim of the exercise was to compare the influence that the physical characteristics of the systems have on limiting and stimulating primary productivity. Also, due to the very similar water chemistry of the two the lakes (Cheng and Tyler 1973a; Cheng and Tyler 1973b; Cheng and Tyler 1976a); Chapters 2 and 3), it is likely that phytoplankton in both systems are influenced by water chemistry to a similar degree.

The simulation of meroplankton by increasing the sediment survival time of diatoms increased the degree of variation described by model output when compared to measured field data in Lake Crescent significantly. This further supports the hypothesis that meroplankton play an important role in shaping and controlling primary productivity in Lake Crescent. The use of the same input files in Lake Sorell with comparable sediment survival times for diatoms and the resulting continued dominance of chlorophytes in this system reflects conditions in Lake Sorell to be less conducive to meroplankton growth and proliferation.

The importance of benthic resting stages in diatoms has been recognised in both freshwater and marine systems (McQuoid and Godhe 2004) with the resuspension of viable spores or cells from sediments having the potential to significantly influence phytoplankton community composition and biomass (Carrick, Aldridge et al. 1993).

Under optimal growth conditions, meroplankton resuspension may act as a seeding mechanism that undergoes selective growth, out-competing other phytoplankton species (Millie, Fahnenstiel et al. 2003). Carrick et al. (1993) found meroplankton resuspension under conditions of high wind significantly altered phytoplankton community composition, biomass and size.

Interestingly, the time resting spores spend in the sediments isolated from the euphotic zone does not decrease the time taken for germination but does decrease the percent of spores able to germinate (McQuoid and Hobson 1996). It is therefore possible that the more frequently resting cells and spores are resuspended into the water column the higher the survival and potential growth rates. Consequently, in Lake Crescent, the increased prevalence of sediment resuspension under typical 'historic' conditions may have selected for meroplanktonic dominance by favouring this set of adaptations.

The strong evidence of meroplankton dominance discussed in Chapter 5 is further supported by the results presented here in Chapter 6. Of real importance when making comparisons of the conditions of the lakes during the current study to that of studies undertaken in previous years, such as Cheng and Tyler's work during the early seventies, is that patterns of broad algal division dominance, such as the contrasting dominance of Chlorophyta in Lake Sorell compared to the dominance of Bacillariophyta in Lake Crescent (Figure 66 through to Figure 69, Table 30; Cheng and Tyler (1973a)) and relative biomass differences between the lakes has not changed significantly between then and now. It would therefore be possible to conclude that the mechanisms structuring community characteristics within each lake, and the contrast between lakes, still hold under present conditions, even in the face of significant changes in the physical and chemical water quality characteristics in the lakes observed between 2000 and 2002.

Interestingly, Cheng and Tyler (1973a) found surface production rates in Lake Crescent greatly exceeded those of Lake Sorell; by contrast, rates at 1 m depth were often greater in Sorell than in Crescent. This resulted in production per unit of surface per day and per year in Crescent to be only  $2.6 \times$  that of Sorell (Cheng and Tyler 1976b) despite Lake Crescent maintaining ten times the standing crop of plankton of

Lake Sorell. This may further highlight the important role meroplankton plays in Lake Crescent with the increased turbidity that is typical of Lake Crescent restricting light penetration and in turn limiting primary production (Cheng and Tyler 1976b). The meroplanktonic community of Lake Crescent would be well adapted to such an environment with the development of resting cells and spores allowing a viable plankton community to be maintained even during conditions unfavourable for growth (Schelske 2002).

In both lakes the low lake levels encountered during the current study have increased light attenuation considerably. However, the potential impact on reducing algal productivity (and hence biomass) (Hellstrom 1991; Arfi, Guiral et al. 1993) has not been realised; in fact both modelled and measured algal productivity increased. Thus sediment resuspension in these lakes appears crucial to increased overall algal productivity in both lakes. Accordingly, minimising the frequency and severity of resuspension episodes by increasing lake water levels appears to be the only promising management option for improving their water quality.

There are other examples of lakes elsewhere that share some of these key features. For example, Schelske et al. (1995) described the surface sediments of Lake Apopka, Florida, as consisting of a nepheloid layer made up of meroplanktonic diatoms intermixed with the non-living organic and inorganic materials that are readily resuspended – which appears similar to Lake Crescent. Schelske et al. (1995) also concluded that the water column chlorophyll-*a* concentration was capable of doubling during the course of a wind event. The available data for Lake Crescent is unable to support this, but the strong correlation between SPM and chlorophyll-*a* strongly supports the contention that direct resuspension of meroplanktonic diatoms contributes to the chlorophyll-*a* concentrations in Lake Crescent.

Ogilvie and Mitchell (1998) also found meroplanktonic species of algae to be important in dominating the phytoplankton community, contributing as high as 90% of total plankton community composition 6 hours after artificial resuspension events in enclosure experiments in Hawksbury, Tomahawk and Tuakitoto Lagoons in the South Island of New Zealand. Interestingly, they found that this effect had largely disappeared after 24 hours.

Lowe et al. (2001) documented declines in concentrations of TP, TN and chlorophyll-*a* as being roughly proportional to reductions in the concentration of SPM and concluded that these constituents are more influential on SPM. In the case of lakes Sorell and Crescent the opposite mechanism seems more plausible: the mechanisms driving SPM are causing the changes in the rest of the species. In this chapter, the modelling strongly suggests that increased lake levels result in increased light availability by reducing the amount of sediment resuspension; this, in turn, reduces the levels all of the above mentioned variables. These changes, which ignore the impact of zooplankton grazing, still do not increase algal biomass, despite improved water clarity and nutrient levels remaining high, even under historic conditions. Thus in the case of lakes Sorell and Crescent, it seems that sediment resuspension is a direct stimulus driving increased algal productivity.

This is similar to Bachmann et al.'s (2005) interpretations; they proposed that wind-driven waves resuspended the upper portion of the fluid mud in Lake Apopka, Florida, resulting in high levels of SPM, total phosphorus and chlorophyll-*a* (Carrick, Aldridge et al. 1993; Bachmann, Hoyer et al. 1999). This seems directly comparable with Lake Crescent, with sediment resuspension increasing levels of these variables substantially.

Carrick et al. (1993) used their strong correlations between chlorophyll-*a* and average wind speed as supporting the hypothesis that direct inoculation of meroplankton exists in Lake Apopka, Florida. However, despite the high positive correlation obtained between increased wind speed and increased algal biomass in Lake Crescent, the result does not unequivocally prove resuspension and direct inoculation of meroplankton from the benthos as being responsible for the observed increases in algal productivity. To adequately address this issue, rigorous experiments, such as those described and carried out by Schallenberg and Burns (2004) would be required to tease apart the interplay between nutrients, light and sediment and meroplankton resuspension. Also, detailed real-time logging of chlorophyll-*a* concentration in the lakes would give some indication as to the importance direct inoculation of the water column by meroplankton plays in increasing primary productivity in Lake Crescent, as estimates of natural maximum growth rates are available in the literature (Harris

1986) and measured changes in chlorophyll-*a* greater than about  $1.5 \text{ day}^{-1}$  would strongly suggest that biomass increases could not be due to plankton growth alone. Additional short-interval sampling of phytoplankton community structure before, during and after an event would also be beneficial (Schelske 2002).

Overall, the results of the modelling exercise strongly support the hypothesis that sediment and meroplankton resuspension in Lake Crescent is a fundamental factor driving increased algal productivity in this lake over and above that of Lake Sorell. This is further supported by the contrasting correlations between measured average daily wind speed and algal biomass in the lakes

## **6.5 Conclusion**

Overall, this chapter presents evidence that supports the contention that increased algal productivity in Lake Crescent relies on and is stimulated by higher turbulence and a greater susceptibility to sediment resuspension when compared to Lake Sorell.

Furthermore, the modelling results strongly support the hypothesis that observed increases in plankton biomass in Lake Crescent, on average and during times of increased wind stress, result from the inoculation of the water column by epibenthic meroplankton coupled with increases in both phosphorus and nitrogen, which would further enhance phytoplankton growth. The opposite holds for Lake Sorell with algal biomass showing little reliance on sediment resuspension as a stimulus for increasing productivity.

It may therefore be concluded that the morphometry of Lake Crescent which is more susceptible to sediment resuspension than Lake Sorell, is conducive to maintaining an increased algal standing crop and provides an environment suitable for the establishment and maintenance of meroplanktonic species of diatom that drive increases in algal biomass during sediment resuspension events.

Carrick et al. (1993) concluded that the extent to which meroplankton develop in a lake and become resuspended by wind is likely determined by specific lake characteristics such as local topography, sediment type, lake depth, lake surface area and the presence or absence of macrophytes. The results of the modelling exercise in

this chapter support this conclusion and demonstrate that the factors leading to a meroplanktonic dominated plankton community are likely quite subtle when the closely comparable characteristics of lakes Crescent and Sorell are taken into consideration.

Additionally, modelling of SPM, light and nutrient dynamics have highlighted the importance lake level changes play as a key management initiative in the restoration of lakes Sorell and Crescent. Maintenance of increased lake level in both lakes will be necessary to reduce the level of light attenuation and increase the fraction of the lake bed with sufficient light to allow macrophyte colonisation and growth.

It is worth noting that the assumptions made in the application of DYRESM-CAEDYM to model phytoplankton dynamics presented here in Chapter 6, have simplified the complex, complicated biological interactions that occur in lakes Sorell and Crescent. However, for the purpose of preliminary assessment of the influence turbulence may play in stimulating algal biomass in the lakes, it was believed these assumptions necessary, and to have little bearing on the final conclusions of the chapter.

As the model currently stands, DYRESM-CAEDYM provides an effective tool in modelling key water quality components in lakes Sorell and Crescent capable of investigating management options and forecasting the likely effect on key water quality variables such as SPM and light attenuation levels. However, for more detailed applications that would look further into biological associations within the lakes, along with more comprehensive, applied modelling of phytoplankton and nutrient dynamics, additional calibration of the model would be required.



## Chapter 7 General Discussion

### 7.1 Recapitulation of context and aims

The rapid decline in water quality and the extreme change that occurred in water clarity and light availability during the 1990s prompted this investigation. These changes in water quality were associated with the decline of a valuable trout fishery, while their impacts on endemic and listed flora and fauna remained uncertain. There were additional pressures to manipulate water levels further to attempt to eradicate introduced European Carp, and increased demand for irrigation water coinciding with several consecutive drought years. Therefore, the aims of this research project were twofold. Firstly, to identify the likely driving factors for the decline in water quality, and, secondly, to investigate the basis for the strong contrast in trophic state that exists between the two lakes, and evaluate whether “top-down” trophic cascades could explain Lake Sorell’s differences from Lake Crescent.

Early limnological work by Cheng and Tyler (1973a; 1973b; 1976a; 1976b) found the trophic characteristics of lakes Sorell and Crescent to be considerably different, despite the lakes laying adjacent to one another within the same catchment and sharing similar geological and morphological characteristics. Historically, each lake exhibited strongly contrasting ‘stable-states’: Lake Sorell was a macrophyte-dominated clear water system, while Lake Crescent was a turbid system dominated by phytoplankton. Cheng and Tyler dubbed this a “limnological paradox”.

Lakes Sorell and Crescent share the same trophic structure with similar species assemblages at all trophic levels, but they differ in the densities of individual species. The well documented history of lakes Sorell and Crescent and the management aims of the recreational trout fishery pointed strongly towards a top-down trophic cascade as being a possible causal factor leading to the differences in primary productivity of the two systems.

Furthermore, that the degradation in water quality in both lakes coincided with record low lake levels, pointed towards physical factors playing a role in the rapid decline in water clarity. The aims of the project required a multidisciplinary approach to

investigate both biological and physical processes influencing the ecology of the lakes, in order to establish causation behind the lakes exhibiting characteristics synonymous to alternative stable states in times predating the massive decline in water quality observed in the late 1990s, along with identifying the cause for the current degradation in water quality.

## **7.2 Summary of results**

The historical water quality data assembled in Chapter 2 detailed temporal trends in nutrients, algal biomass and turbidity, and quantified the extremes to which levels of the major plant nutrients, chlorophyll-*a* and turbidity have reached in recent years. Alarmingly, the relationships developed between euphotic depth and turbidity demonstrated that, at turbidity and SPM levels typical of the period between 2000 and 2002, the maximum euphotic depth would have been little more than 20 cm. This effectively eliminates macrophyte growth at the sediment surface throughout both lakes. Strong correlations between algal biomass, nutrients, turbidity and the light attenuation coefficient demonstrated how closely these components were linked. The marked increase in major plant nutrients and algal biomass in both lakes suggested increased eutrophication. However, quantifying nutrient and sediment inputs from external sources demonstrated that nutrient and sediment loading from tributaries were inconsequential, despite agriculture and forestry activity in the catchment. It was therefore concluded that observed increases in nutrients and SPM were chiefly derived from internal sources.

Multiple linear regression analysis carried out in Chapter 3 demonstrated inorganic SPM to be the main component responsible for the degradation in water clarity. Reductions in lake level were found to be highly significant, leading to increases in SPM and turbidity which resulted from increased exposure of the lake bed sediment to wind driven waves. This prompted a comparison between modelled energy levels exerted on the lake bed prior to 1999 and after 1999 which showed that reduced water levels after 1999 dramatically increased the area of the lake beds in contact with the base of wind driven waves.

Concomitant increases in colloid concentrations alongside those of water column SPM suggested strongly that sediment erosion had increased beyond levels typically

experienced by the lakes in the past, and contributed to the severe degradation in water quality. Furthermore, the colloidal material in the lakes was shown to behave largely as a conservative substance, with reductions in concentration depending on dilution via inflow and rainfall that, in turn, were offset by evaporative concentration during dry periods. Chapter 3 concluded colloids to have the capacity to maintain high background levels of turbidity that would limit the euphotic depth in both lakes Sorell and Crescent and retard macrophyte re-establishment and growth.

The sediment resuspension investigation initiated in Chapter 4 began by quantifying gradients of sediment shear stress across the lake basins under varying water level and wind speed scenarios. It was demonstrated that high shear stress levels capable of resuspending sediments were realised at low to moderate wind speeds at the low water levels attained by both lakes during the period 1999 to 2002. A detailed sediment resuspension model in DYRESM-CAEDYM was successfully calibrated and validated to run on lakes Sorell and Crescent for the period running April 2000 to December 2001. The low water levels typical of the lakes for this period resulted in massive increases in modelled sediment entrainment. Model manipulations investigating the influence increased lake levels had on sediment entrainment showed that raising water levels can substantially reduce the amount of sediment resuspension.

Chapter 4 confirmed the conclusions in Chapter 2 about the importance of internal loading processes and further reinforced the conclusions made in Chapter 3, that reductions in lake level between 1999 and 2002 dramatically increased the energy exerted at the lake bed thus increasing the rate and extent of sediment erosion. Lake level management was, therefore, identified as the most promising effective means to limit the impacts of sediment resuspension.

Recognising and quantifying the processes responsible for the decline in water quality failed to account for the contrasting levels of primary production that exists between the two systems. The need to increase our understanding of the ecology of both lakes and provide the best information possible to adequately manage the lakes initiated the study into top-down trophic interactions. The aim of the work was to assess the degree to which trophic interactions defined ecosystem function and determine if

biomanipulation is a realistic management option for limiting algal standing crop in either lake Sorell or Crescent.

Thus far the importance of biotic interactions and the potential for top-down interactions (and, by implication, the potential utility of biomanipulation) remained unevaluated. Chapter 5 presented a conceptual model of the major trophic pathways present in both lakes. Investigation of the top order consumers concluded that differing recreational fishery management practices between the two lakes increased the abundance and density of the predatory brown trout in Lake Sorell. This increased predation pressure on the golden galaxiid in Lake Sorell. Further, the greater galaxiid density in Lake Crescent reduced the occurrence of large bodied zooplankton and resulted in a zooplankton community dominated by small-bodied cladocerans and copepods. In contrast, zooplankton grazing rates by galaxiids were significantly reduced in Lake Sorell, resulting in large bodied cladocerans being present with increased frequency. However, estimates of potential cladoceran clearance rates of phytoplankton in Lake Sorell failed to account for its lesser algal standing crop. Thus, despite strong top-down trophic interactions occurring in both lakes, the trophic cascade appears effectively decoupled at the zooplankton / phytoplankton link in Lake Sorell and was therefore insufficient to explain Cheng and Tyler's "limnological paradox".

Further investigation of the phytoplankton communities of the lakes, past and present, showed Lake Crescent to be dominated by meroplanktonic taxa of diatoms. This prompted the formation of an alternative hypothesis based on the presence of differing energy and turbulence regimes between the two lakes, as quantified in Chapter 3, favouring the formation of a more viable, productive meroplankton community in Lake Crescent over Lake Sorell.

Chapter 6 attempted to determine if differences in the morphometry of the two lakes alone was capable of stimulating meroplanktonic primary productivity in Lake Crescent as opposed to Lake Sorell. A more detailed application of DYRESM-CAEDYM was employed that incorporated nutrient, light and phytoplankton dynamics. Once the models were calibrated, model runs were undertaken with lake

levels representative of conditions encountered during the present study, with additional runs made with lake levels typical of those prior to 1999.

The results showed that phytoplankton productivity was linked to basin morphometry, lake level management and sediment resuspension in Lake Crescent, that stimulated and favoured the meroplanktonic diatom community both under historic conditions and conditions encountered in the lake during the current study. By contrast Lake Sorell showed little if any correlation between algal productivity and sediment resuspension, with the physical environment of Lake Sorell favouring a mixed algal community of chlorophytes and diatoms, with a marked reduction in algal standing crop overall. The results of the DYRESM-CAEDYM modelling exercise were further validated by correlating average daily wind speed with field measured algal biomass in both lakes; Lake Crescent returned a strong correlation compared to a weak, insignificant correlation for Lake Sorell.

The results strongly support the hypothesis that the increased primary productivity of Lake Crescent is driven by physical processes that are more pronounced than in Lake Sorell. Furthermore, the dominance of meroplanktonic forms of phytoplankton (Chapter 5) in Lake Crescent predisposes it to reaching high levels of phytoplankton productivity when under the influence of sediment resuspension. Padisak and Reynolds (2003) discuss the evolution of meroplanktonic diatoms in shallow lakes and concluded that it occurred under sufficiently frequent or continuous wind mixing and sediment entrainment. The hydrodynamic conditions of Lake Crescent favour the establishment and maintenance of a meroplanktonic phytoplankton community, in contrast to Lake Sorell where conditions are more favourable for a mixed plankton assemblage of reduced productivity.

It is likely, therefore, that the characteristic clear-water, macrophyte- dominated state of Lake Sorell and the turbid, phytoplankton-dominated state of Lake Crescent results from differences in their hydrological environment rather than differing top-down trophic influences. Furthermore, increasing lake level is likely to dramatically improve the trophic status of the lakes resulting in reductions in nitrogen and phosphorus concentration, increased light availability, and potentially reduced algal productivity.

Lakes Sorell and Crescent provide an example of two lakes in which water level fluctuations are fundamental in driving ecosystem process and defining the trophic status of the water body. Suspended sediment, phosphorus, nitrogen, light attenuation and algal biomass have all been shown to dramatically increase with falling water level. Investigation into the effects water level fluctuations have on key water quality variables showed that returning lake levels to average levels typical of the period predating 1999 would have dramatic effects on improving water quality.

In a management context, it must be remembered that natural seasonal fluctuations in water level are important in structuring wetland communities and maintaining ecosystem diversity and the health of shallow lakes (Heffer 2003). However, when the amplitude of change increases significantly, the water level fluctuation becomes a strong disturbance that has the potential to destabilise the water body, as evidenced with lakes Sorell and Crescent. Similar examples are available overseas with Noges et al. (2003) finding water level to be a leading factor controlling nutrient cycling, light availability and the phytoplankton community composition in Lake Vortsjarv, Estonia. Additionally, Bachmann et al. (2000) note low water levels reached in Lake Apopka in the 1950s to be coincident with massive fish kills that occurred in the lake during extreme wind events. Coops et al. (2003) also identify water level management to be extremely important in lake restoration.

The persistence of high colloid concentrations in both lakes is of particular concern. It appears this material will limit the euphotic depth in both lakes and retard the re-establishment of aquatic macrophytes for at least another 5 years given the current low lake levels. The reliance on dilution and flushing to reduce the concentration of colloids, coupled with the potential for climate change to further limit rainfall in the catchment, may increase the longevity of this material in the water column. Additionally, a sustained reduction in lake level below the points reached in 2000 and 2001 would increase exposure of sediments that have been little disturbed in the past and result in additional release of colloids.

If climate change reduces average annual rainfall in the catchment, then long-term reductions in water levels will likely result. Under such conditions, the frequency and

extent of sediment resuspension would increase and Lake Sorell may evolve in a similar way as Lake Crescent, with the establishment of a dominant meroplanktonic plankton community that reduces light availability and out competes aquatic macrophytes. It is also possible that the entrainment of long-undisturbed, consolidated sediments in Lake Sorell has resulted in a new pool of more easily resuspended material that may increase the susceptibility of this lake to sediment entrainment, along with retarding the growth of macrophytes due to poor substrate composition (Bachmann, Hoyer et al. 1999).

It is possible that the release of colloids from the largely undisturbed, well consolidated sediments of Lake Sorell reflects a 'worst case scenario' in that this material, once released in the water column, will remain in suspension indefinitely until it is exported and lost from the system. It therefore follows that, provided lake levels remain above the record low levels reached in March of 2000 and 2001 (Appendix 10), a comparable release of colloidal material should not eventuate, as this material has been lost from the sediment secondary layer, and over time, lost from the system.

An additional factor that should be considered in the future management of the lakes, particularly if lake levels remain low, is the potential for fluid mud to develop through wave induced liquefaction of once consolidated sediments (Bachmann, Hoyer et al. 2005). The frequent resuspension of sediments in lakes Sorell and Crescent during periods of extremely low lake level, may give rise to the development of fluid mud. Furthermore, it has been postulated that the existence of fluid mud seriously retards the establishment of aquatic macrophytes due to unstable sediments and the inability of plant roots to find secure anchorage against wave shear (Bachmann, Hoyer et al. 1999; Lowe, Battoe et al. 2001).

### **7.3 Synthesis and implications**

The theory of alternative stable-states as described by Scheffer (1998) and its application to account for shallow lakes alternating between clear and turbid conditions has been widely accepted (Jeppesen, Jensen et al. 1999). Lakes Sorell and Crescent provided an opportunity to study trophic dynamics and investigate the applicability of alternative stable-state theory, as both systems reflect lakes that have

been in contrasting alternative ‘stable states’ in the past, with Lake Sorell maintaining a state of relatively clear water, with an expansive cover of macrophytes while Lake Crescent had a greater algal biomass that led to elevated turbidities (Cheng and Tyler 1973a), and a limited cover of macrophytes.

The severe decline in water levels experienced by both lakes in the late 1990s introduced a major disturbance that eroded the relatively stable equilibrium states of both lakes, with the lakes becoming increasingly turbid due to severe increases in the magnitude and extent of sediment resuspension. Lake Sorell, in particular, seemed to be following the model of a lake ‘flipping’ from a clear to turbid state.

A similar change has been documented for Lake Apopka, Florida. Bachmann et al. (2005) felt that Lake Apopka was a classic example of a lake switching between alternative stable states. One hypothesis for the switch from a macrophyte dominated, clear water state to a turbid, phytoplankton dominated state in Lake Apopka was the passage of a category 4 hurricane in 1947 that passed through the area, causing extreme winds that destabilised sediments and uprooted macrophytes (Schelske, Carrick et al. 1995; Bachmann, Hoyer et al. 2001). This switch may be likened to Lake Sorell, with record low lake levels resulting in severe increases in bed shear stress, akin to increased storm-induced turbulence experienced by Lake Apopka.

Interestingly, two mechanisms have been proposed for the maintenance of the turbid, phytoplankton dominated state of Lake Apopka, both of which are relevant to lakes Sorell and Crescent. One school of thought, proposed by Schelske and Lowe, is that chlorophyll-*a* and increased algal production largely drives light limitation, and retards the re-establishment of rooted aquatic macrophytes (Schelske, Carrick et al. 1995; Schelske, Aldridge et al. 1999; Schelske, Coveney et al. 2000; Lowe, Battoe et al. 2001; Schelske and Kenney 2001). By contrast, Bachmann, Canfield and Hoyer (Canfield, Shireman et al. 1984; Bachmann, Hoyer et al. 1999; Bachmann, Hoyer et al. 2000; Canfield, Bachman et al. 2000; Bachmann, Hoyer et al. 2001; Bachmann, Hoyer et al. 2001; Bachmann, Hoyer et al. 2005) maintain that high levels of loosely flocculated sediments (termed ‘fluid mud’, a mixture of fine sediments and water that has virtually no shear strength (Bachmann, Hoyer et al. 2005)) has developed in the lakes, increasing the frequency and extent of sediment resuspension. It is these



resuspension events which increase the internal loading of nutrients to the water column that lead to the observed eutrophication (Bachmann, Hoyer et al. 1999; Bachmann, Hoyer et al. 2000; Bachmann, Hoyer et al. 2000).

Lakes Crescent and Sorell seem to conform to Bachmann et al.'s (1999; 2000; 2000) hypothesis. There is little evidence that the increased nutrient concentrations of the late 1990s have been derived from external sources, and the contribution of algae to total SPM and the influence on the light climate is minor. The empirical data suggest increased productivity to be driven by internal loading exacerbated by resuspension, and the modelling with DYRESM-CAEDYM strongly supports this mechanism to be plausible. This, coupled with the inorganic and colloidal material being more easily resuspended as lake levels drop, resembles the role played by Bachmann et al.'s 'fluid mud' and mimics an ecosystem in which sediment resuspension rather than increased algal productivity limits light and retards macrophyte growth.

Failure to recognise the importance sediment resuspension has on influencing productivity and trophic status may result in invalid conclusions and mismanagement. Canfield et al. (2000) document how, in recent years, it has become common practice for lake managers to try to control phosphorus whenever faced with problematic levels of planktonic algae, regardless of what other management actions may be considered. This point is supported by Kairesalo et al. (1999) who state that generally, a reduction in the external nutrient load is always the primary restoration measure followed to provide a basis for the recovery of a eutrophic lake. It is obvious that, in lakes Crescent and Sorell, management focussed on reducing nutrient inputs alone would have little impact on controlling algal productivity. Additionally, the application of biomanipulation by some managers to control algal productivity is deemed equally applicable (Auer, Storey et al. 1990; Hanson and Butler 1990; Hosper and Jagtman 1990; Jeppesen, Jensen et al. 1999; Meijer, de Boois et al. 1999; Sondergaard, Jeppesen et al. 2000). However, work presented in Chapter 5 demonstrated the weak associations between zooplankton and algal standing crop in lakes Sorell and Crescent despite the presence of significant trophic interactions at higher trophic levels, and brings into question the applicability of biomanipulation as a management tool in these two systems.

Although changes in lake water levels appear fundamental to the recent changes in these lakes, restoration of 'historically typical' levels may not restore Lake Sorell, in particular, to its former state. As asserted by Hosper (1998), a comprehensive ecosystem approach may be required to adequately understand ecosystem function and provide the necessary information required for effective restoration and management of a shallow lake.

Although the evidence presented in Chapter 5 does not support a classic, top-down trophic cascade for Lake Sorell (cf. Carpenter, Kitchell et al. 1985), completely disregarding the roles of higher trophic levels may be premature. For example, under conditions of increased algal and sediment resuspension, the influence of biotic top-down processes such as grazing by zooplankton, and bottom-up processes such as nutrient and light limitation in influencing algal standing crop may be altered, and further research may be needed to forecast the likely trajectories of these lakes should the low water levels persist if the prolonged drought continues in eastern Tasmania. Increased sediment resuspension may become a more frequent and persistent feature of these lakes. How will the biota respond?

Currently, the importance sediment resuspension plays to higher trophic interactions in shallow lake communities, and its potential to affect top-down control of productivity is poorly researched, especially in Australia. Nonetheless, resuspension may have important effects. For example, Levine et al. (2005) quantified the effect sediment resuspension had on *Daphnia* clearance rates and found an increase in turbidity from 2.2 NTU to 15 NTU reduced threshold phytoplankton feeding rates by 25%. Levine et al. (2005) went on to conclude that sediment resuspension has the potential to affect energy flow through food webs by reducing the significance of zooplankton - phytoplankton interactions. The importance of cladoceran grazing on limiting phytoplankton productivity in Lake Sorell would therefore increase during times of reduced turbidity, and may have the potential to limit phytoplankton. Information on zooplankton - phytoplankton interactions in Lake Sorell during times of improved water clarity would be necessary to fully evaluate the importance cladoceran grazing has in limiting algal standing crop, as the increased prevalence of sediment resuspension in lakes Sorell and Crescent experienced during the current

study may confound the degree to which trophic interactions influence primary productivity.

The results of the modelling exercises presented in Chapter 3 and Chapter 6 proved the physical environments of the lakes to differ, with the morphology and typical water level management scenarios of Lake Crescent leading to a higher susceptibility to sediment resuspension that stimulates algal productivity. This result is similar to studies elsewhere that found significant interactions between sediment entrainment and primary productivity through either direct resuspension of chlorophyll or indirectly through increased resuspension and release of nitrogen and phosphorus (Hamilton and Mitchell 1988). A further example of the positive effect sediment resuspension may have on algal productivity is the work of Carrick et al. (1993) who found direct entrainment of meroplankton to be highly significant in increasing algal standing crop in Lake Apopka, Florida. But in direct contrast to the situation described by Hellstrom (1991) who found sediment resuspension to reduce mean light intensity and limit phytoplankton productivity by up to 85% in a shallow lake in Sweden.

This is in contrast to Lake Sorell, where sediment resuspension is less pronounced under historic conditions (Chapter 3) and appears to favour a phytoplankton community with a reduced dominance of meroplanktonic forms (Chapter 5). The increase in sediment ‘accumulation’ areas (Hakanson 1982) or areas of reduced turbulence that would occur in Lake Sorell compared to Lake Crescent (Chapter 3), may act as sinks for algal cells and provide a mechanism for the long-term loss of meroplankton from the water column. This coupled with a reduction in areal extent to which the lake bed is influenced by wind driven waves may help to limit the establishment of a meroplanktonic dominated phytoplankton community in Lake Sorell.

Field-based mapping of accumulation, transportation and erosion areas across both lakes would provide further information to interpret the importance loss processes may have in limiting meroplankton dominance in Lake Sorell compared to Lake Crescent, along with providing detailed information to ground truth and validate a more sophisticated, spatially explicit 3-dimensional hydrodynamic mixing model.

Such a model would have the capacity to resolve spatial gradients in turbulence and sediment exposure throughout the lakes. The successful application of a 3-dimensional model would help to spatially map the hydrodynamic environments of the lakes as well as providing a means to study the interplay between sediment resuspension and phytoplankton growth dynamics.

Additionally, to improve the capacity of DYRESM-CAEDYM to simulate phytoplankton growth dynamics in lakes Sorell and Crescent, in-situ and laboratory based quantification of parameters specific to modelling phytoplankton growth dynamics in the lakes, such as the maximum potential growth rate; the respiration rate; settling velocity ; and the half saturation constants for nutrient uptake of the phytoplankton group / species of interest would be beneficial (Lewis, Brookes et al. 2004). Such information would be necessary if a more detailed and applied investigation of plankton growth dynamics was pursued.

To utilise the capacity of DYRESM-CAEDYM to study more complex interactions of ecosystem function in either lake, such as higher order trophic interactions, additional data to calibrate DYRESM-CAEDYM would be required.

Quantifying and modelling the relationships between turbidity and cladoceran clearance rates would be of particular value in modelling zooplankton - phytoplankton interactions in systems with frequent and highly variable changes in turbidity and SPM. Such an algorithm could be readily integrated into the modelling framework, and coupled with information on the possible impacts increased SPM loading has on retarding cladoceran population growth would be a valuable addition to CAEDYM and allow for a more realistic representation of zooplankton dynamics in shallow, turbulent lentic systems such as lakes Sorell and Crescent.

Additionally, accurate data on the concentration of golden galaxiids in both lakes would be required to determine the potential for limitation of Cladocera through predation. It is also likely that the feeding rates of golden galaxiids and the brown trout (Stuart-Smith, Richardson et al. 2004) are adversely affected by increased turbidity. In a similar fashion as proposed for cladocerans, developing and integrating relationships into CAEDYM describing the effects of turbidity on foraging efficiency

of these two key fish species should increase the ability of the model to simulate higher order trophic interactions in shallow systems such as lakes Sorell and Crescent

Also, at present, there is little capacity for DYRESM-CAEDYM to model the influence of macrophytes on complex ecosystem processes such as sediment resuspension, competition with phytoplankton for nutrient uptake or the ability to provide refuge for zooplankton. Processes that are important to the functioning of a shallow lake ecosystem (Scheffer 1998). For example, developing CAEDYM algorithms that model freshwater macrophyte growth and cover and feed back into zooplankton - zooplanktivorous fish interactions would be valuable. Also, quantifying the level to which macrophytes influence sediment resuspension (e.g. effects of growth form and density) would be beneficial for modelling the role of increased macrophytes in improving water quality in shallow lakes.

DYRESM-CAEDYM also has the capacity to model a 'tracer' with a set decay rate that could be employed to model reductions in colloid levels in the lakes. There are no management options readily available for reducing the concentration of colloids in water bodies the size of lakes Sorell and Crescent, so being able to detail the longevity of such substances would be beneficial for predicting the long term impacts colloids may have on reducing light availability and effecting macrophyte productivity in the lakes. Also, sediment analysis of cores to identify the horizon between the sediment primary and secondary layers and the amount of colloidal material present in each would help to quantify the potential for further release of colloids from the sediments under differing lake level management scenarios relative to those experienced in recent years.

Finally, further, more detailed work would be required to determine the exact process that drives increased algal productivity in Lake Crescent and the explicit role resuspension plays on influencing primary productivity. The present study failed to determine if increased algal productivity was via sediment resuspension and release of nutrients occluded to the SPM or present in the interstitial pore water stimulating algal productivity, or whether it was via direct water column inoculation of meroplankton from the sediments. In order to answer confidently the question of the origin of the meroplankton that increase the short term productivity of Lake Crescent, additional

experimental and field based trials are required. Experiments such as those conducted by Schallenberg and Burns (2004) would have the capacity to identify the source of the meroplankton, and determine if increased algal productivity in Lake Crescent is due to direct water column inoculation of meroplankton from the sediments; or if increased productivity is due to resuspension and release of nutrients that stimulates algal growth. Also, additional field trials with real-time chlorophyll and turbidity logging equipment deployed during sediment resuspension events, coupled with frequent sampling of water column plankton before, during and after the event, would further aid our understanding of the complex processes that limit algal productivity in Lake Sorell and stimulate algal productivity in Lake Crescent. There also would be merit in running algal bioassays to assess the degree of nutrient limitation in the lakes, and to see if this has changed significantly since the work of Cheng and Tyler.

The inaccuracies inherent in the spectrophotometric determination of phaeopigment limited accurate determination of the contribution made by Pha to resuspended algal biomass (Webb, Burnison et al. 1992; Axler and Owen 1994). In future research, particularly with respect to algal resuspension, a more advanced method of chlorophyll and phaeopigment measurement, such as high performance liquid chromatography (HPLC) (Humphrey and Jeffrey 1997; Mantoura, Jeffrey et al. 1997), would be valuable to determine the precise contribution phaeopigment makes to resuspended algal samples.

## **7.4 Conclusions**

In light of all available data collected during the current project, it is most likely that the process controlling turbidity, nutrient cycling, light attenuation and lake productivity in lakes Sorell and Crescent is sediment resuspension.

Management options for reducing sediment resuspension include increasing lake water levels and fetch manipulation. Fetch manipulation for water bodies the size of lakes Sorell and Crescent would involve extensive earthworks and have the potential to do more harm than good, particularly in the face of ensuring the continued survival of the endemics of these two lakes. Therefore, manipulating water levels is the most feasible management option, with the modelling showing that it has considerable promise.

Furthermore, it has been shown in lakes Sorell and Crescent that increasing light availability through a reduction in sediment resuspension will result in a reduction in algal biomass, this is despite the possibility of an improved light climate favouring algal dominance, particularly after a stable equilibrium state has been destabilised (Hellstrom 1991; Scheffer and Jeppesen 1998). The reason for algal production declining despite increased water clarity is that internal loading by resuspended nutrients should decrease as lake levels increase, because higher water levels mean less of the lake beds are exposed to above-critical shear stresses.

At the low lake levels experienced by the lakes in recent times, resuspension of sediments occurs at very low wind speeds across large areas of the lake basin. Such winds occur frequently and under even relatively calm conditions, there is sufficient turbulence to keep sediment in suspension. With resuspension happening so frequently, consolidation of sediments is not possible, further compounding the problem. The nature of the relationships show turbidity to increase logarithmically with decreasing lake level, therefore, a small drop in lake level at low levels can cause extreme increases in turbidity.

Ecologically, it is likely that the contrast in algal productivity between lakes Sorell and Crescent is driven by contrasts in the hydrodynamic environments of the lakes. Furthermore, a more detailed study on the significance of trophic interactions in the lakes may be warranted, particularly at times when water clarity and SPM concentrations improve, as the relationships between zooplankton and phytoplankton grazing may be confounded by high turbidity levels (Levine, Zehrer et al. 2005). Expanding the current application of DYRESM-CAEDYM to include zooplankton, galaxiid and trout interactions would further add to our current understanding of the ecosystem processes in lakes Sorell and Crescent and help with evaluating other scenarios such as what might happen to the higher trophic levels if prolonged drought keeps the water levels in these lakes low.

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Appendices

Appendix 1      Hypsographic curves of lakes Sorell and Crescent.

Hypsographic curves of lakes Sorell and Crescent were produced by digitising and analysing contour areas of the bathymetric maps (Chapter 2, Figure 1 and Figure 2) using MapInfo V6. In the case of Lake Sorell, bathymetric data extended to a maximum depth of 801.4 m (ASL). It was therefore not possible to extend the hypsographic analysis beyond this point.

Figure 1      Hypsographic curves for Lake Sorell.

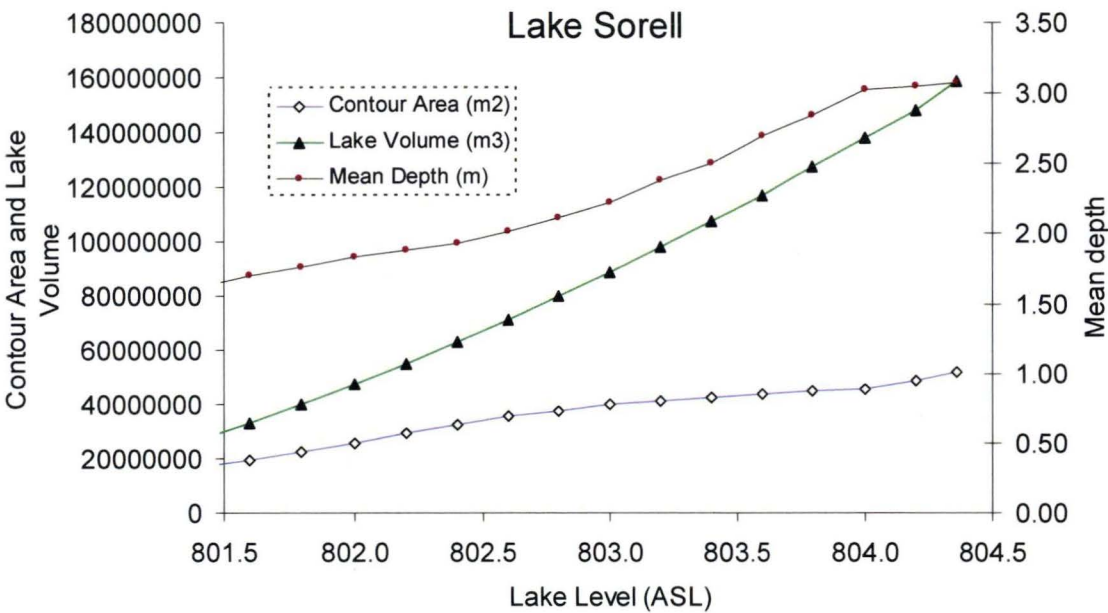
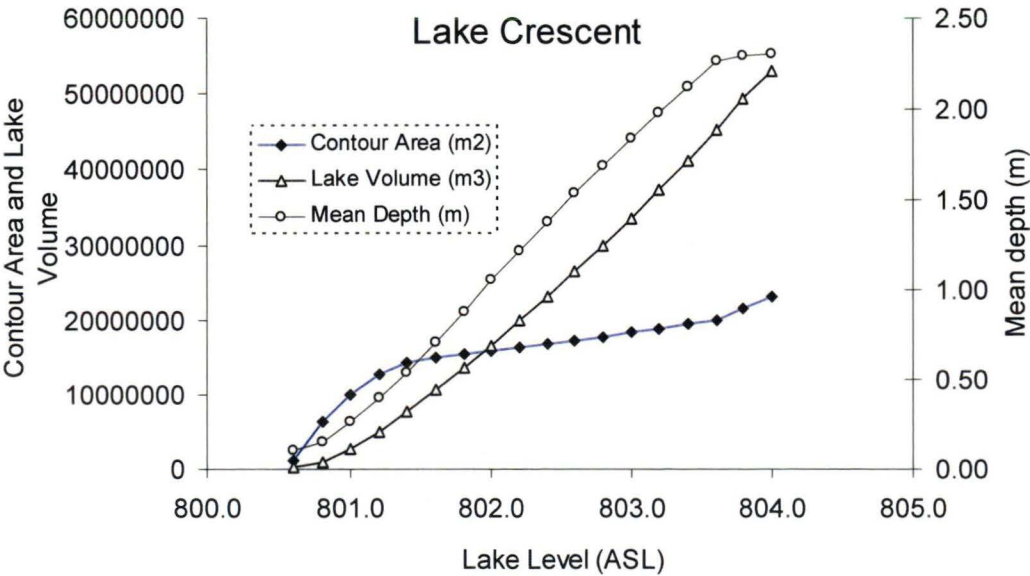


Figure 2      Hypsographic curves for Lake Crescent.



## **Appendix 2      Comparisons of measurements made by four independent turbidity meters on Formazin and field samples.**

During the course of the project, four separate turbidity meters were used for in-situ measurement and logging of turbidity. Instruments used included:

Hach 2100P Portable Turbidimeter  
McVann Analite NEP 160 turbidity meter with an Analite NEP 290 probe  
Analite Model 156 Portable Nephelometer  
Greenspan TS 300 0-500NTU turbidity logger

Different meters were used for several reasons. The Hach 2100P was used for measuring filtered turbidity of 1µm filtrate, and was the only instrument capable of analysing small sample volumes. The 2100P was also used for turbidity measurements made pre-April 2000. Access to the 2100P was limited during the course of the project, so the McVann NEP 160 was used as a dedicated field instrument measuring 'in-situ' turbidity for all routine monitoring trips. The Analite 156 and the Greenspan TS 300 were both used as dedicated turbidity loggers, and were deployed for several weeks at a time, monitoring changes in suspended sediment concentration with changing environmental conditions.

By definition, each turbidity meter should return comparable values when field measurements are made, providing meters are calibrated correctly. Comparisons were made between total and filtered turbidity, between historic and current turbidity levels, and also using turbidity logger measurements to approximate suspended sediment concentration. Therefore, it was critical to ensure that all turbidity meters returned comparable measurements.

To verify this, each meters calibration was checked against Formazin standards according to the manufacturers specifications, and adjusted if necessary. The Formazin was prepared according to Standard Methods (A.P.H.A, 1992). Measurements were made on a variety of different Formazin concentrations and each meters response noted. The meters returned comparable measurements when made on Formazin suspensions (refer figure 1).

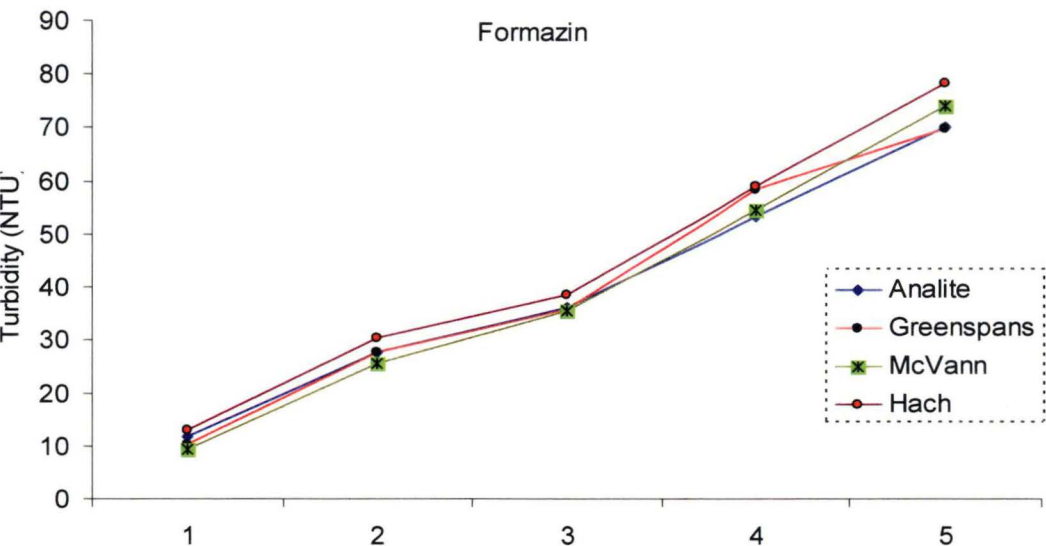
However, large discrepancies existed when made on samples of lake water from both lakes Sorell and Crescent (refer figures 2 and 3). This posed a significant problem, as measurements made by independent turbidity meters were not comparable with each other. This is similar to work carried out by (Gippel 1988; Gippel 1995) who demonstrated that two turbidity meters, when calibrated against a Formazin standard returned comparable readings but when measurements were made on a latex suspension, the discrepancy in readings was up to 3 times in comparison.

Following this, it was hoped to establish accurate relationships between each meter and the Hach 2100P in regards to measurements made on field samples. If comparative measurements between meters could be accurately modelled, then each meter's measurments could be standardised so as to be comparable to the Hach 2100P. This would enable accurate turbidity comparisons between measurements

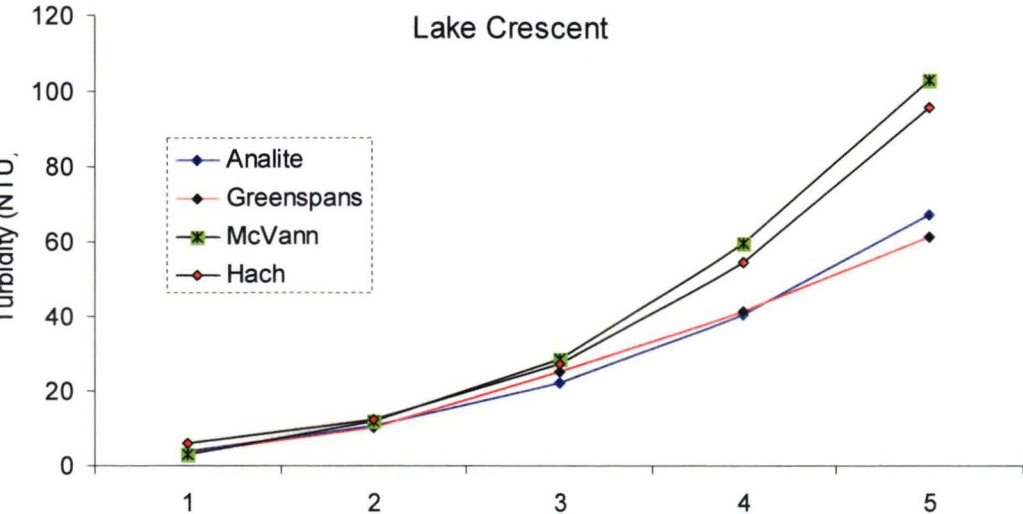
made with different meters and allow comparisons to be made to turbidity readings from previous studies.

Figure 4 illustrates the relationship found between measurements made with the McVann NEP160 and the Hach 2100P. The relationship was almost perfectly linear and the resulting regression equation was used to standardise all field measurements made with the McVann NEP160 so as to be comparable to the Hach 2100P. Analysis of data for the remaining two meters returned similar results (refer table 1). From these relationships, all measurements made were adjusted to be comparable to the Hach 2100P.

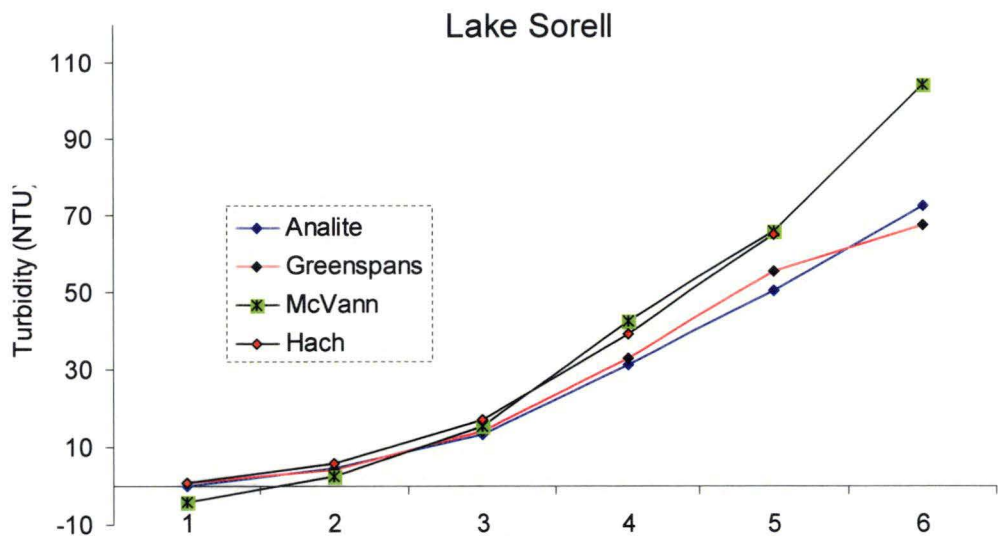
**Figure 1** Comparison of Formazin readings from the four turbidity meters used during the project.



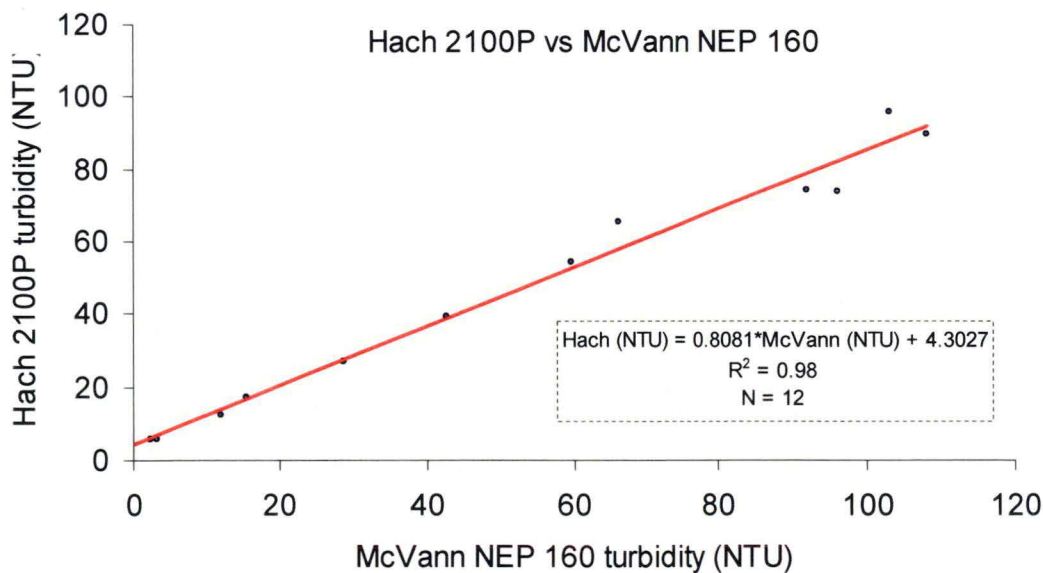
**Figure 2** Comparison of meter readings made on a water sample from Lake Crescent.



**Figure 3** Comparisons of readings between meters made on a Lake Sorell water sample.



**Figure 4** Comparison of measurements made with the McVann NEP 160 turbidity probe and the Hach 2100P nephelometer.



**Table 1** Linear regression coefficients comparing the Hach 2100P and the turbidity meters utilised during the project. The relationships were used to standardise all turbidity measurements so as to be comparable to the Hach 2100P.

Meter	Coefficient	Intercept	r-squ	P
Analite	1.375	-1.146	0.99	<<0.001
McVann	0.8081	4.3027	0.98	<<0.001
Greenspans	1.378	-1.942	0.96	<<0.001



### Appendix 3      Comparisons of two spectrophotometers for g440 analysis.

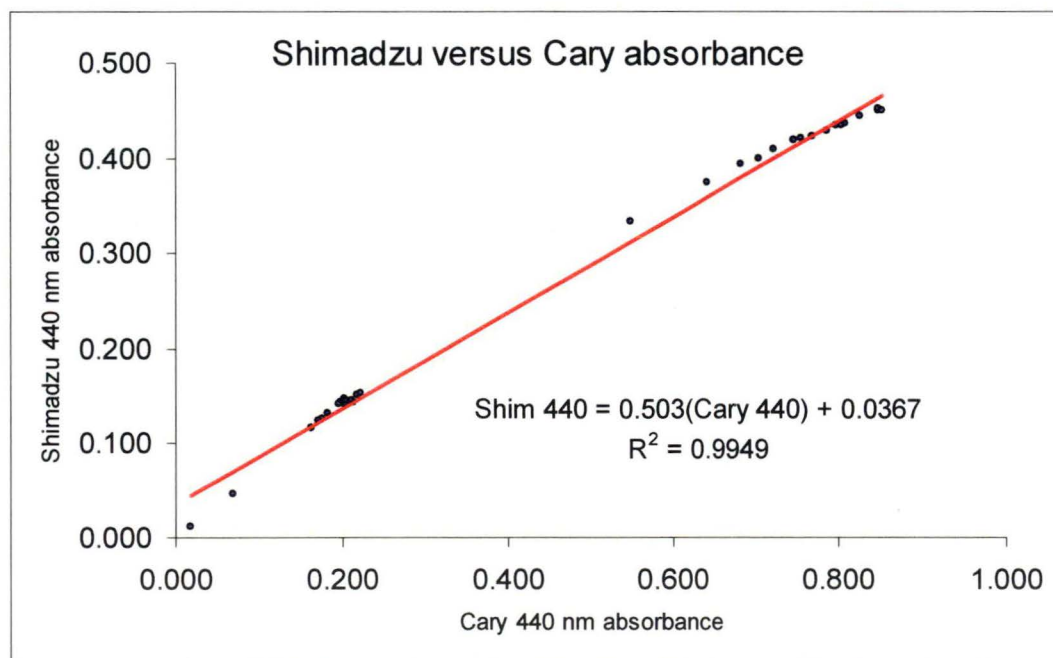
Dissolved colour measured as g440, was determined spectrophotometrically according to the method outlined by Kirk (1976). The method involved measuring the absorbance at 440nm of filtrate passed through a 0.45µm membrane filter paper.

The analysis was initially carried out on a Shimadzu UV-120-02 spectrophotometer in a 4cm cuvette. Later, g440's were measured on a Varian Cary 50 CONC spectrophotometer in a 10 cm cuvette. This was due to the Shimadzu being replaced during the course of the project. Theoretically, the absorbance of a sample at 440nm should be comparable between machines, however, it was found that the Varian measured significantly higher absorbance than the Shimadzu. The reasons for this are unknown, but to gain comparative readings between sample dates for the course of the project, it was necessary to correct readings so that they were comparable between instruments.

#### Method outline:

Comparisons of absorbance measurements were made between the two machines for a sweet of samples from both lakes. It was found that the comparative response of both machines to increasing levels of dissolved colour was almost perfectly linear (refer figure 1). Using the linear regression equation outlined in figure 1, all field sample measurements made on the Varian were converted as to be comparable to readings made on the Shimadzu.

**Figure 1**      Graphical and linear regression analysis of the relationship between absorbance of field samples at 440nm for the two spectrophotometers.



## **Appendix 4      Colloidal solids analysis.**

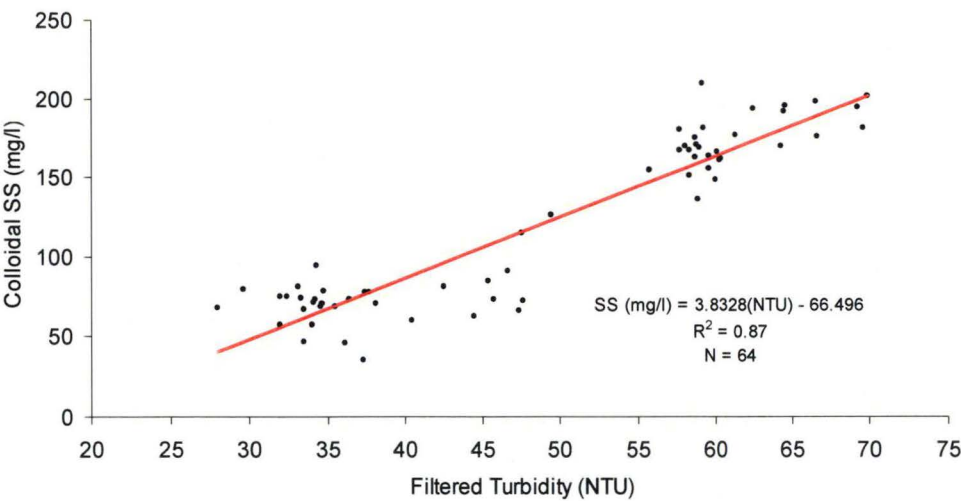
It was evident that the colloidal component present in both lakes Sorell and Crescent had the potential to significantly contribute to the total mass of particulate material in suspension. The increase in levels of filtered turbidity was highly significant, and understanding the relationship between the mass of colloidal solids and filtered turbidity would help in understanding the extent of erosion which occurred in both lakes. Also the potential contribution, by mass, of colloids to background turbidity could be assessed.

Standard methods for determining suspended solids (A.P.H.A.1992) did not sample the colloidal material, as it passed through the filter paper and was not retained. It was necessary to devise a methodology that would effectively measure the mass contribution of colloids to total suspended sediment.

It was found that freezing of water samples resulted in almost complete flocculation of the colloidal material. Floc formation appeared to occur within the ice crystal lattice as the water samples froze. On thawing, the flocculated colloidal material, having a greater particle size was easily retained on a standard 1  $\mu\text{m}$  (nominal pore size) glass-fibre filter paper. By comparing the mass of suspended solids on unfrozen samples to the mass of suspended solids of samples that had been frozen and thawed, it was possible to accurately determine the mass of colloidal material.

Figure 1 outlines the relationship between filtered turbidity and the mass of colloidal solids. Using the relationship, it was possible to approximate the mass of colloidal solids from previous filtered turbidity readings for the duration of the project. Table 1 is a summary of the ability of the technique to remove colloids, and also illustrates the significance of colloids in contributing to the total mass of suspended sediment. What is notable is the clarity of the filtrate after the frozen samples were filtered, dropping from 46 NTU and 66 NTU for lakes Crescent and Sorell respectively, to below 1.4 NTU. Water of this clarity is comparable to zero blanks prepared from analytical grade deionised water and measured on the Hach 2100P.

**Figure 1** Linear regression analysis of the mass of colloidal solids and filtered turbidity for 64 samples taken over several dates from both lakes.



**Table 1** Overview of the contribution of colloidal and particulate solids to the total measured solids and the changes in turbidity before and after freezing.

Lake	n	Total Turbidity before freezing (NTU)	Filtered Turbidity before freezing (NTU)	SS before freezing (mg/L)	SS after freezing (mg/L)	Filtered Turbidity after freezing (NTU)
Crescent	8	168	46	167.3	248.2	1.34
Sorell	8	152	66	90.8	282.3	1.09

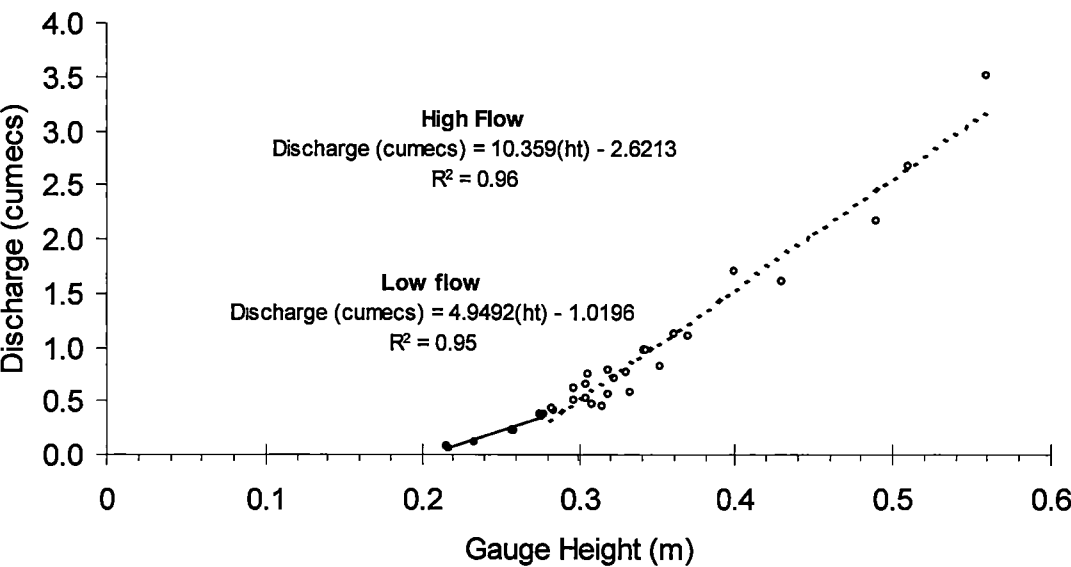
Appendix 5 Stream discharge ratings.

Table 1 is a summary of the location of each of the logged weir pools and sample sites for each of the tributaries sampled during 2000 and 2001. Figures 1 through 4 are the gauge ratings of measured instantaneous discharge and stage height. The relationships were used to produce the hydrographs (figures 5 through 12) for each of the tributaries and approximate discharge for 2000 and 2001. The high level of hysteresis present for the drain leaving Kemps Marsh is due to wind forcing occurring on the marsh that influenced the degree of discharge independent to inflows. The hydrograph for the Kemps Marsh drain for 2000 does not include discharge pre September 2000 which may influence the overall estimate of the volume entering the lake from this tributary. The extent of flow for this period is thought to be minimal due to the low level of water that had accumulated in the marsh.

Table 1 Location of gauged sites and sampling stations for each of the tributaries.

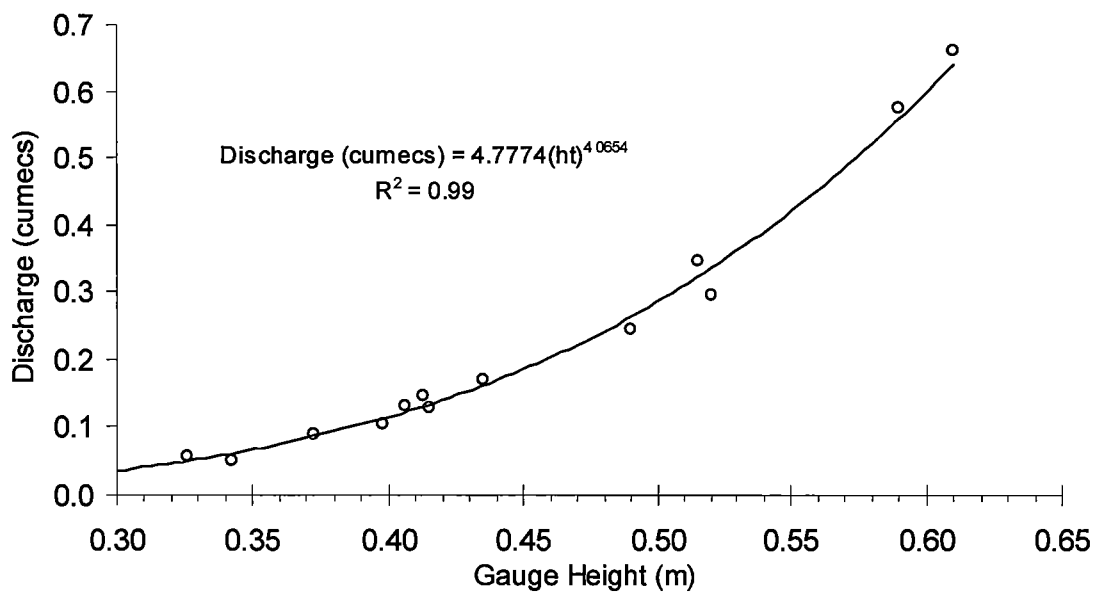
	Easting	Northing
Mountain Creek	515 740	5 342 100
Silver Plains Creek	508 470	5 338 020
North Silver Plains Creek	508 620	5 339 160
Kemps Marsh Drain	511 660	5 335 580
Above Kemps Marsh	509 450	5 334 990
Agnews Creek	505 930	5 332 260

Figure 1 Mountain Creek gauge rating.

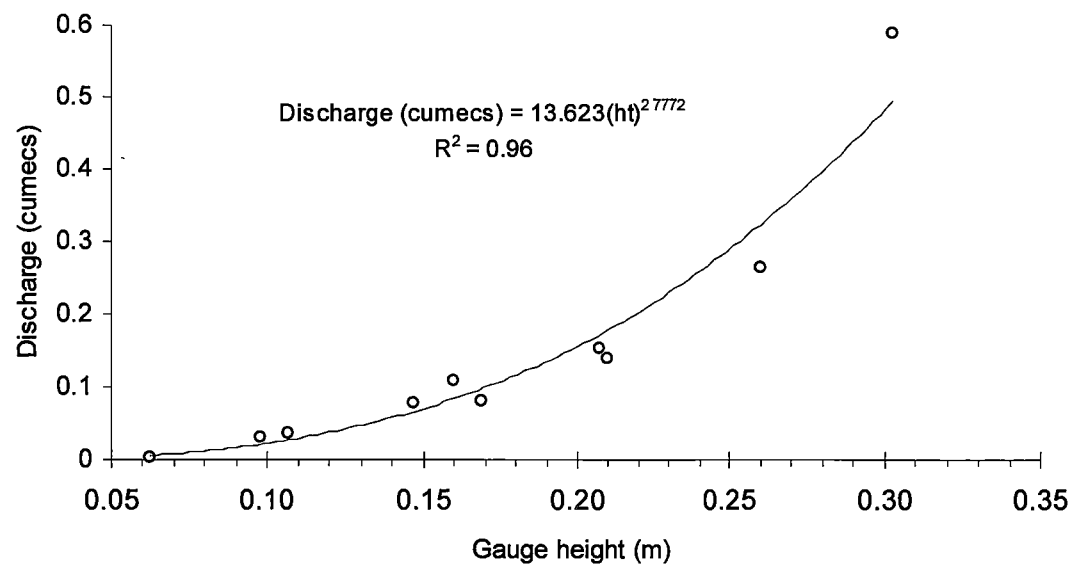




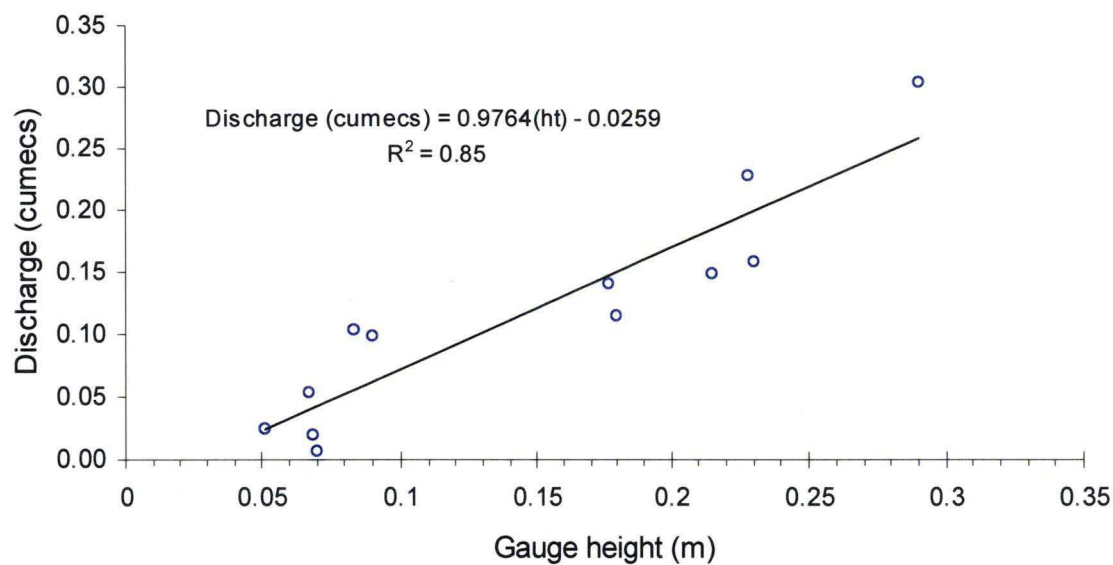
**Figure 2** Silver Plains Creek gauge rating.



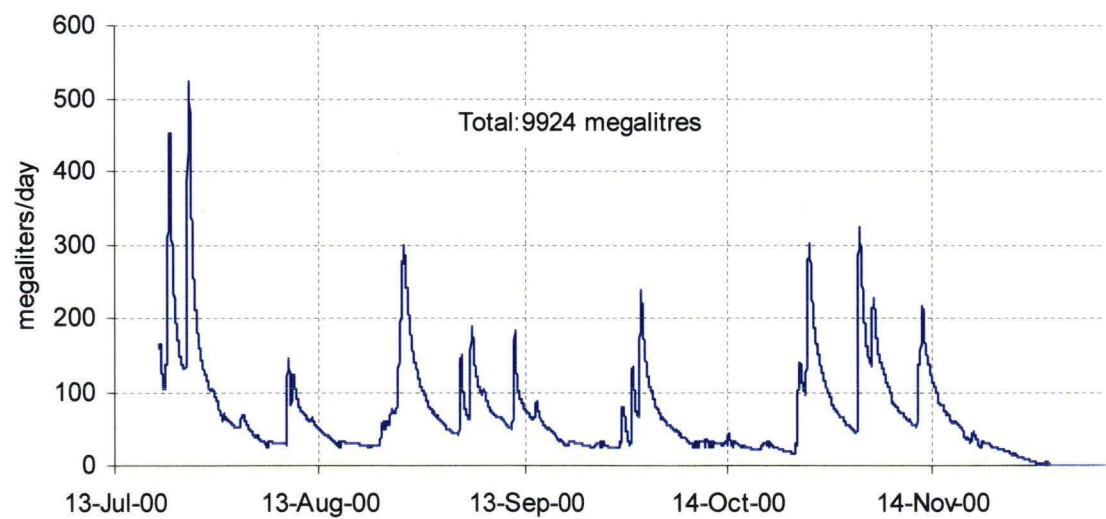
**Figure 3** North Silver Plains Creek gauge rating.



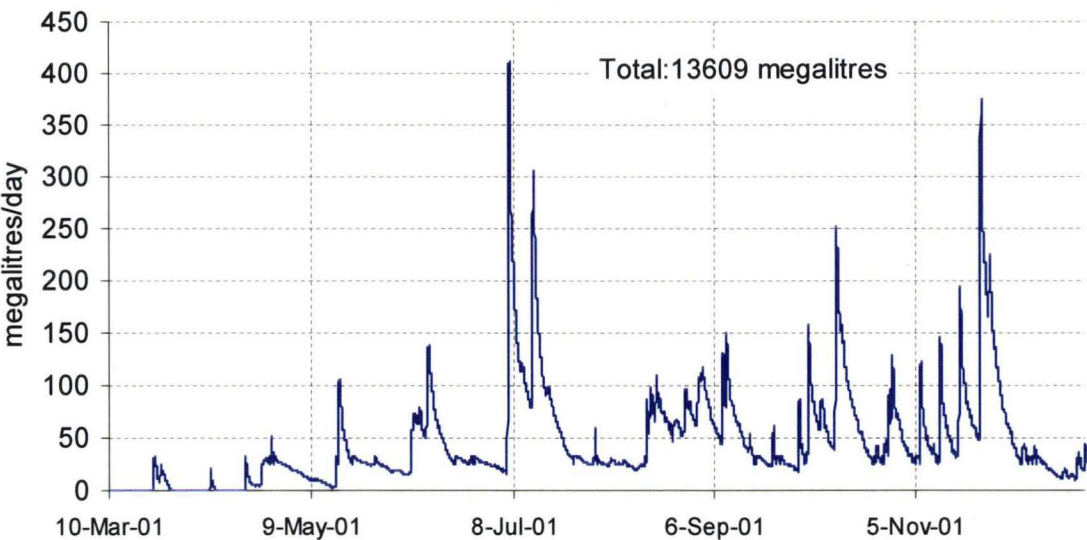
**Figure 4**      Kemps Marsh drain gauge rating.



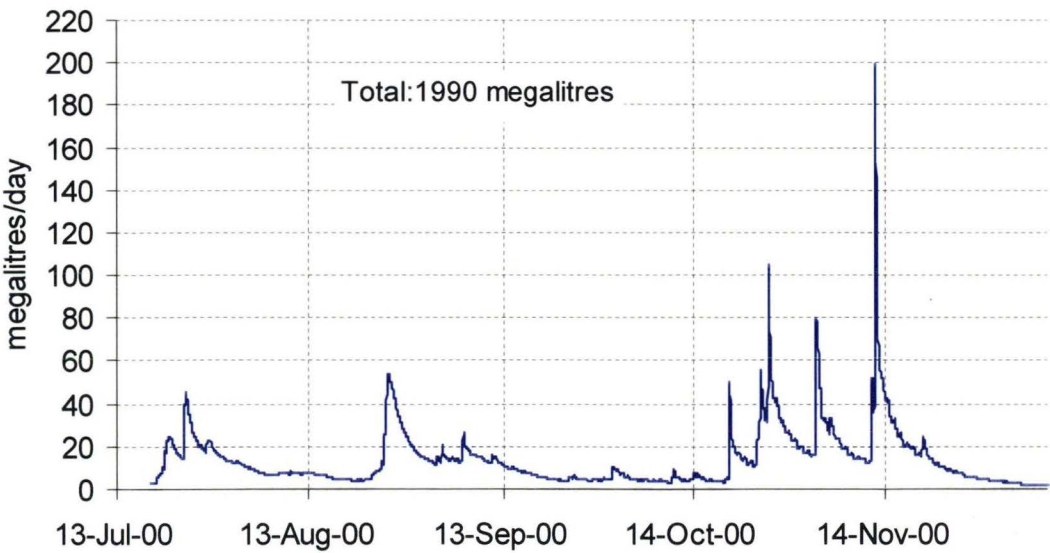
**Figure 5**      Mountain Creek hydrograph for 2000.



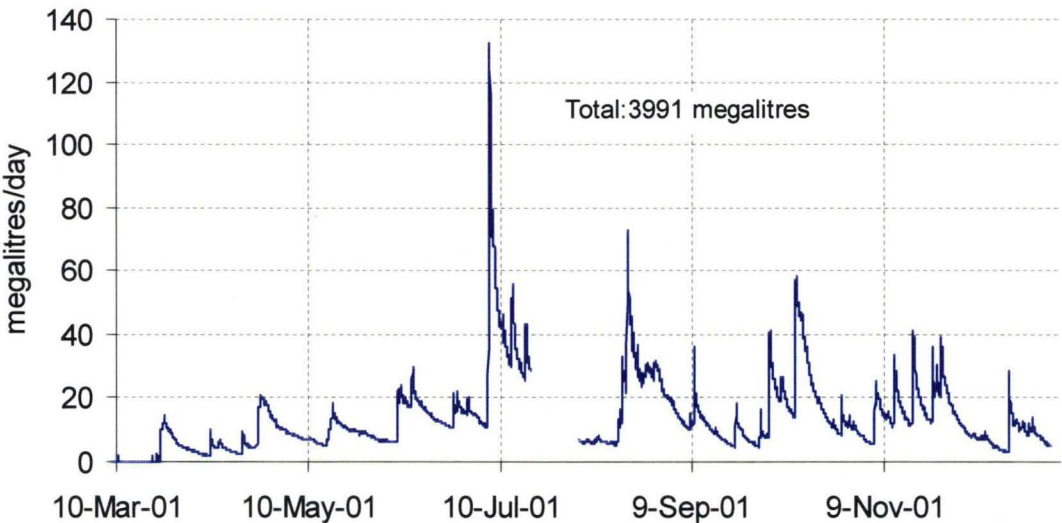
**Figure 6** Mountain Creek hydrograph for 2001.



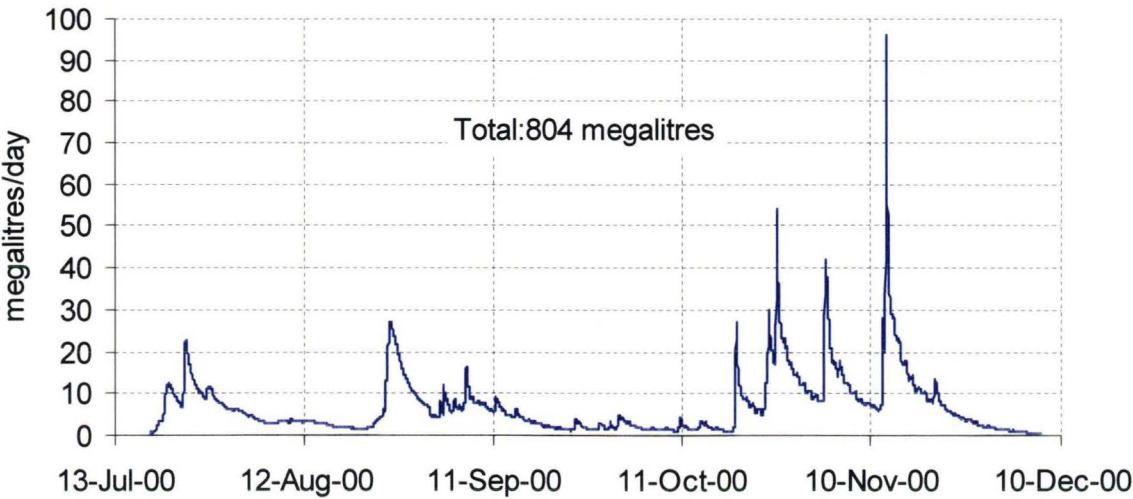
**Figure 7** Silver Plains Creek hydrograph for 2000.



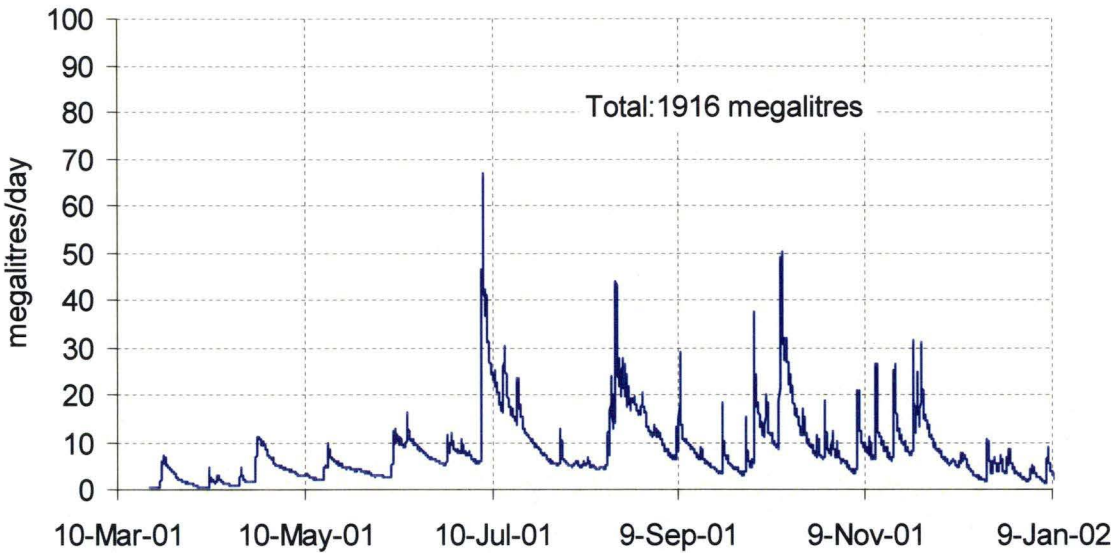
**Figure 8** Silver Plains Creek hydrograph for 2001.



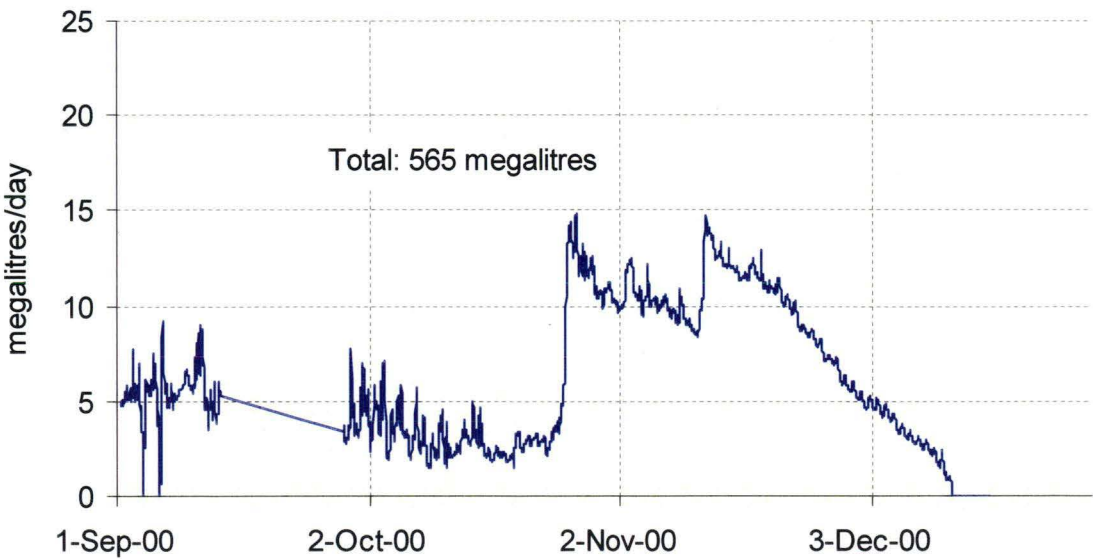
**Figure 9** North Silver Plains Creek hydrograph for 2000.



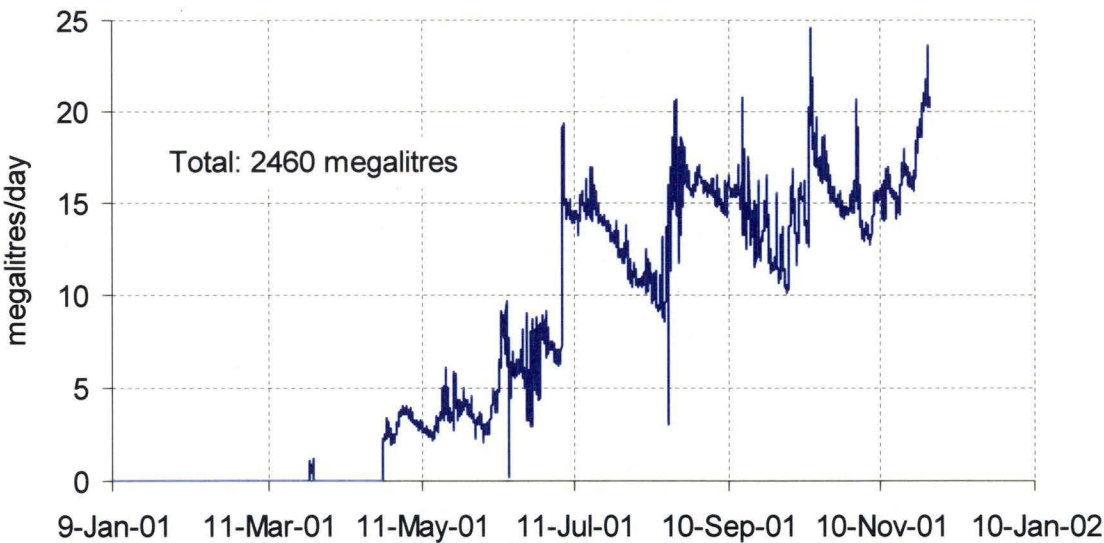
**Figure 10** North Silver Plains Creek hydrograph for 2001.



**Figure 11** Kemps Marsh drain hydrograph for 2000, excluding flows previous to September 1, 2000.



**Figure 12**      Kemps Marsh drain hydrograph 2001.



**Discharge correlations comparing Agnews Creek to other gauged sites**

Agnews Creek was recognised as the main tributary (excluding the Interlaken canal) entering Lake Crescent. It was not possible to find a suitable site for the placement of a logger in Agnews Creek for reasons of security and weir pool suitability. In light of this, it was thought that a significant correlation between instantaneous flow from Agnews Creek and one of the other logged catchments would be found, and flow from Agnews Creek could be approximated from the hydrograph of one of the other sites.

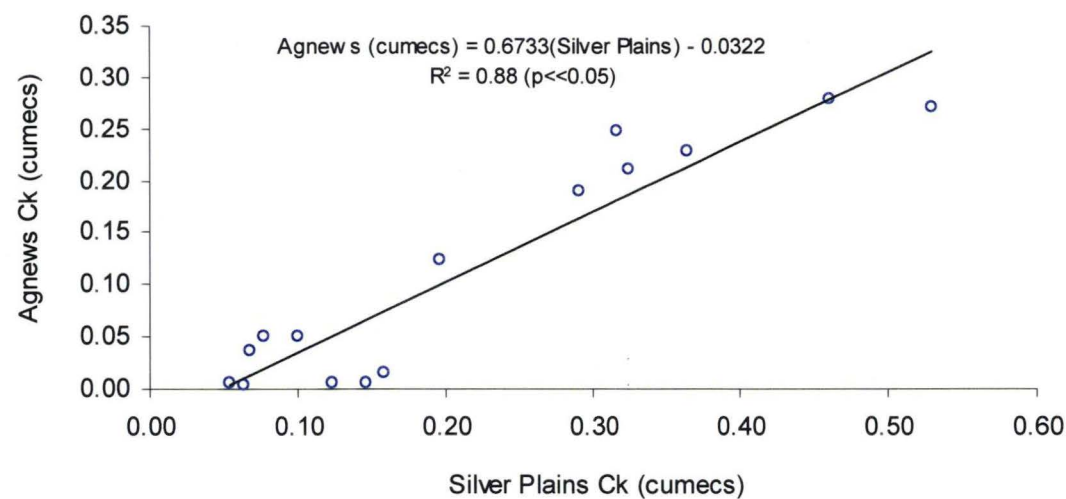
Table 2 outlines the correlations between instantaneous discharge of Agnews Creek and discharge from each of the other monitored tributaries. All correlations are highly significant. The most suitable for approximating Agnews Creek flow was found to be Silver Plains Creek, as it had the highest correlation coefficient. Figure 13 outlines the regression analysis relating discharge of Silver Plains Creek and Agnews Creek. This relationship was used to produce the hydrographs for 2000 and 2001 for Agnews Creek (refer figures 14 and 15) and approximate total stream discharge for these time periods.

**Table 2**      Pearson correlation matrix comparing measured instantaneous discharge at Agnews Creek, with instantaneous discharge from the four logged sites.

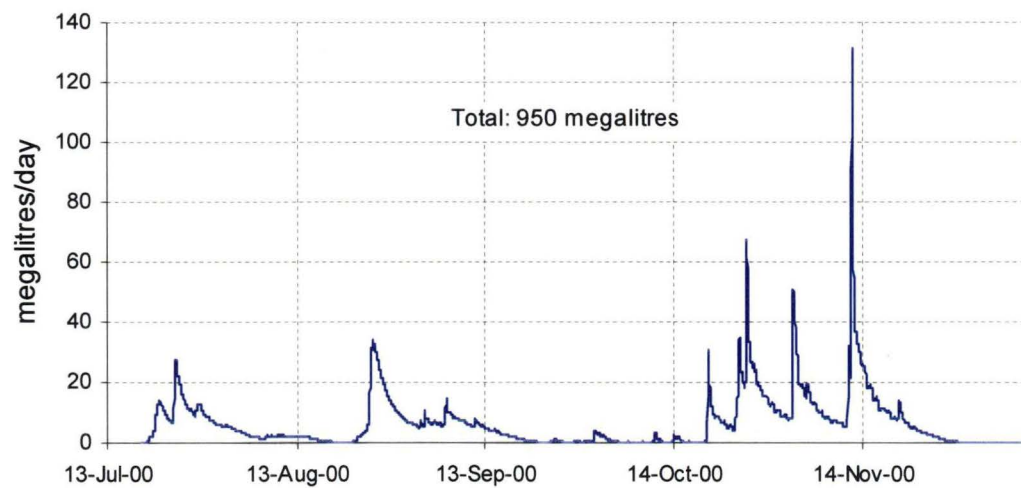
	Agnews Creek
Silver Plains Creek	0.940
Mountain Creek	0.751
North Silver Plains Creek	0.939
Kemps Marsh Drain	0.873



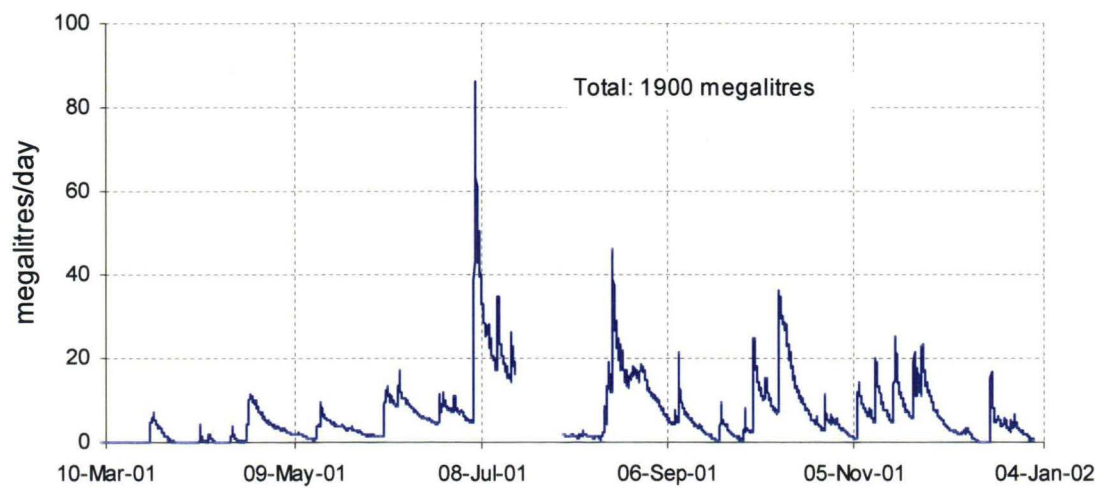
**Figure 13** Regression relationship comparing logged instantaneous discharge measured at Silver Plains Creek to instantaneous discharge measured at Agnews Creek.



**Figure 14** Agnews Creek hydrograph for 2000.



**Figure 15** Agnews Creek hydrograph for 2001.

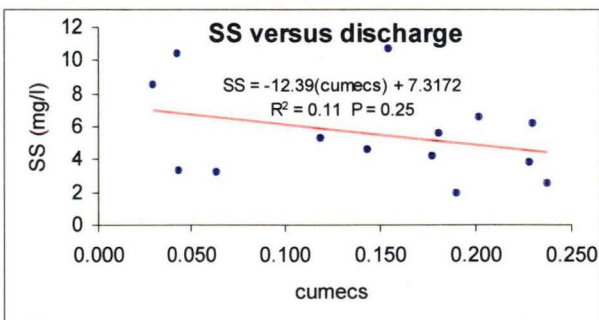
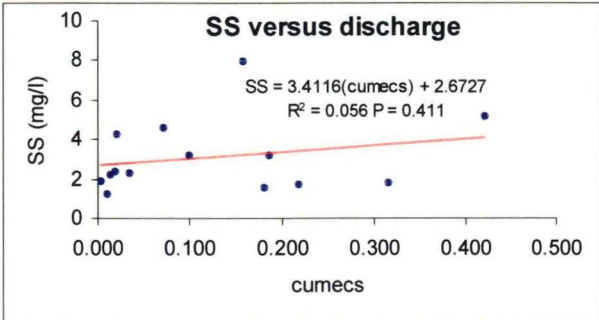
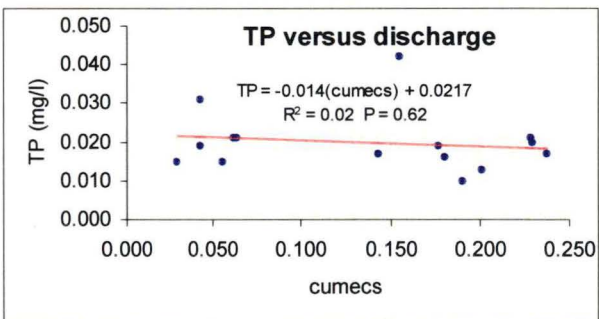
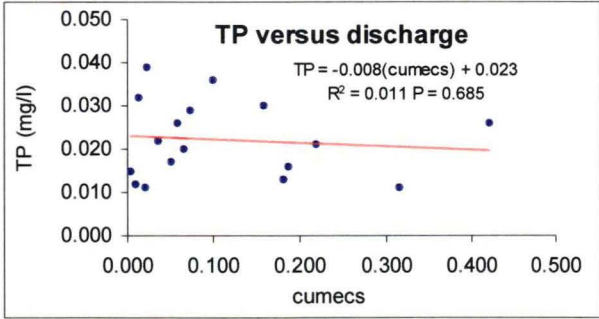
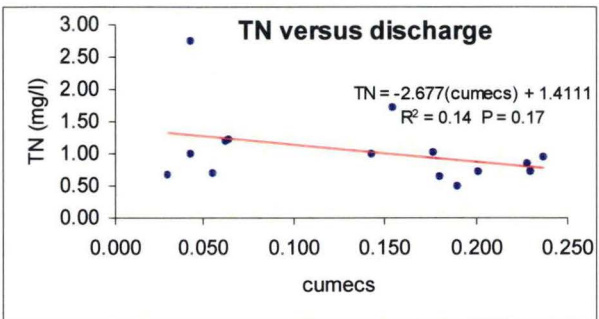
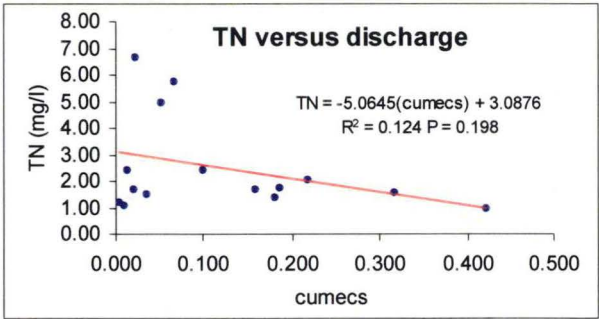


**Appendix 6      Stream discharge versus nutrient and sediment concentration.**

The following figures outline the regression relationships between total nitrogen (TN), total phosphorus (TP), suspended solids (SS), and instantaneous discharge (cumecs) for all the monitored tributaries of lakes Sorell and Crescent. The regression equations for North Silver Plains Creek and Silver Plains Creek for TN, TP and SS were used to approximate nutrient and sediment concentration from instantaneous discharge. For Mountain Creek, the relationship for TN was used to approximate TN concentration from instantaneous discharge. For the remainder, it was assumed a linear change in concentration occurred between sample dates.

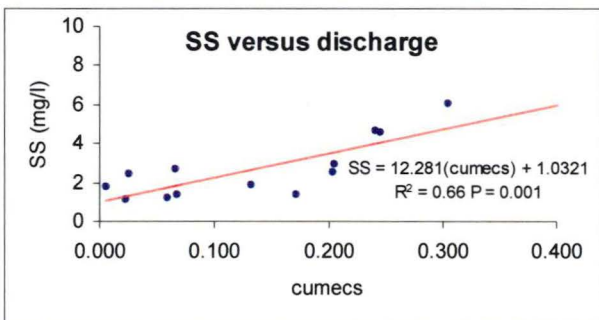
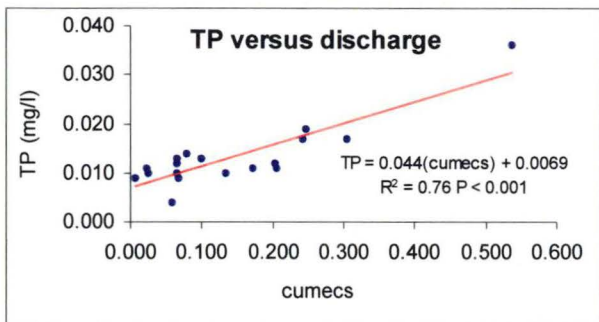
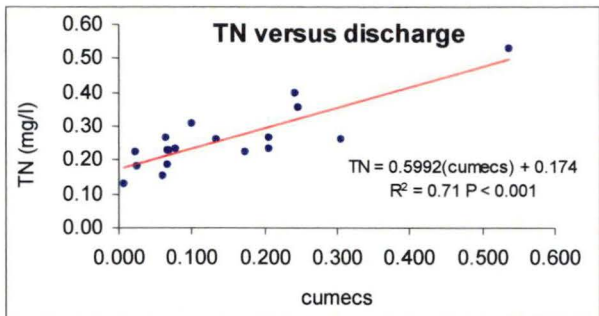
**Agnews Creek**

**Kemps Marsh Drain**

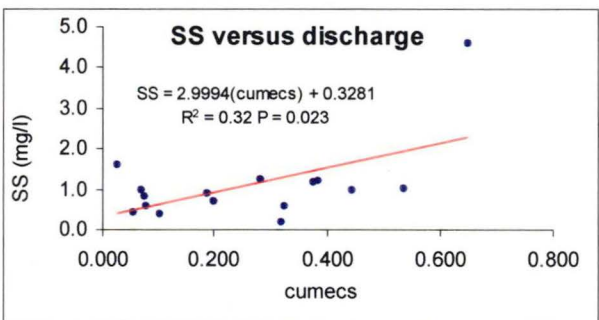
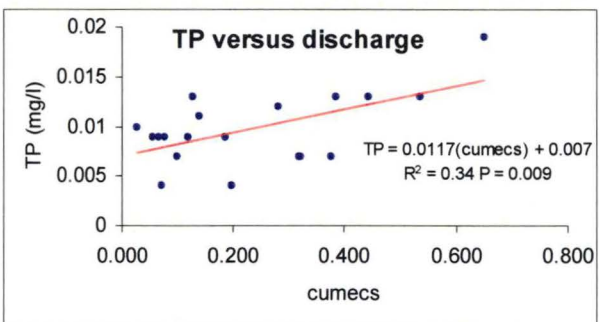
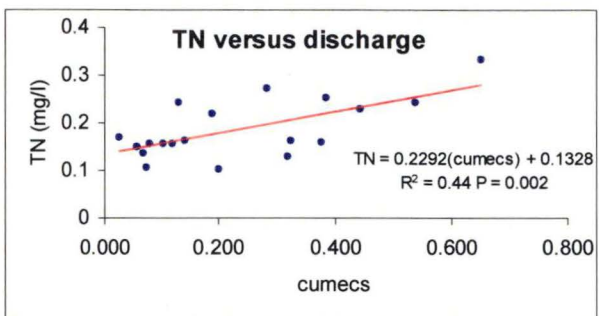




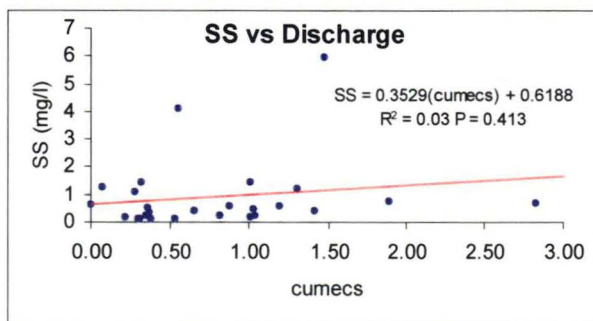
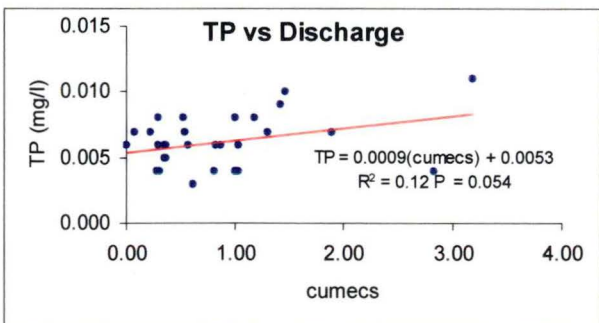
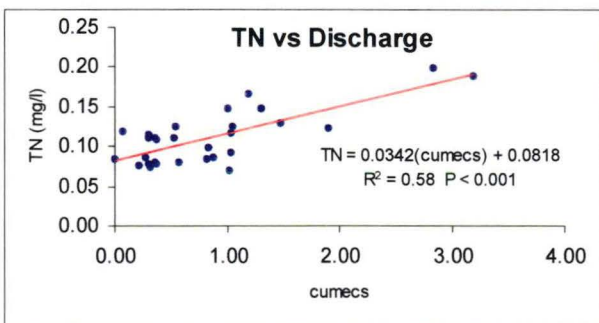
### North Silver Plains Creek



### Silver Plains Creek



### Mountain Creek

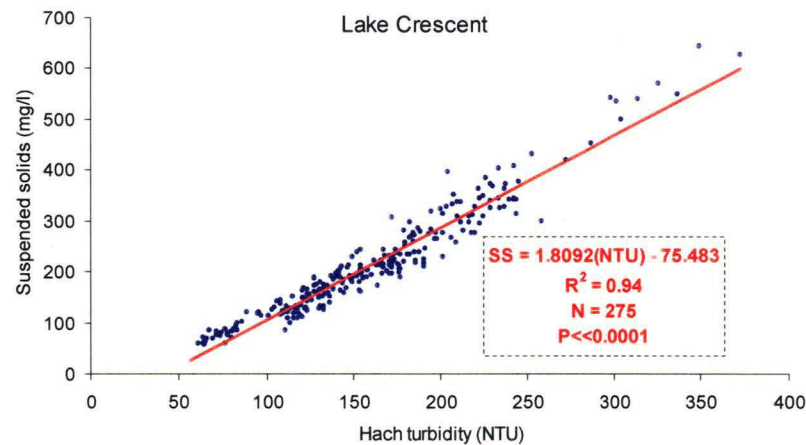


**Appendix 7      Turbidity versus suspended solids concentration of routine monitoring samples.**

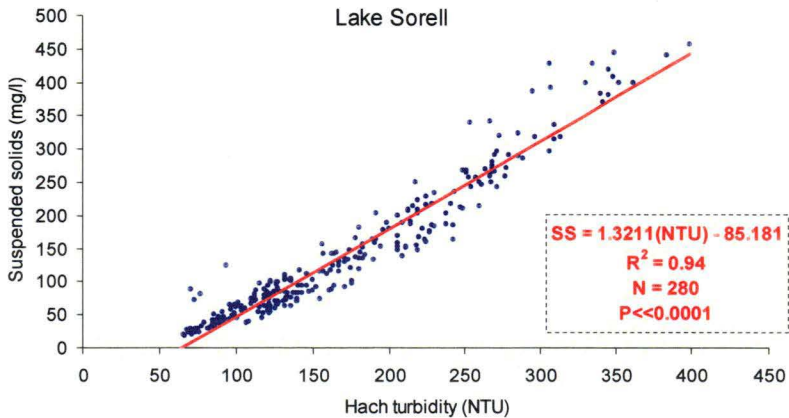
Turbidity is a measure of light scattering, and is largely dependent on the amount of suspended particulate material present in the water column. Due to this, turbidity has been used extensively to approximate suspended solids concentration, and offers an extremely quick and effective tool for monitoring changes in levels of suspended sediment.(Gippel 1989) During the project, turbidity loggers were used to monitor changes in suspended sediment levels. This was dependent on developing a robust relationship between turbidity and suspended solids concentration.

Linear regression analysis was carried out on all turbidity and suspended solids samples from trips made between April 2000 to January 2002. The resulting regression relationships (refer figures 1 and 2) were found to be highly significant, and were used to approximate suspended sediment concentration from turbidity readings made by the turbidity loggers.

**Figure 1**      Graphical representation of the linear relationship between measured ‘in-situ’ turbidity and suspended solids for Lake Crescent.

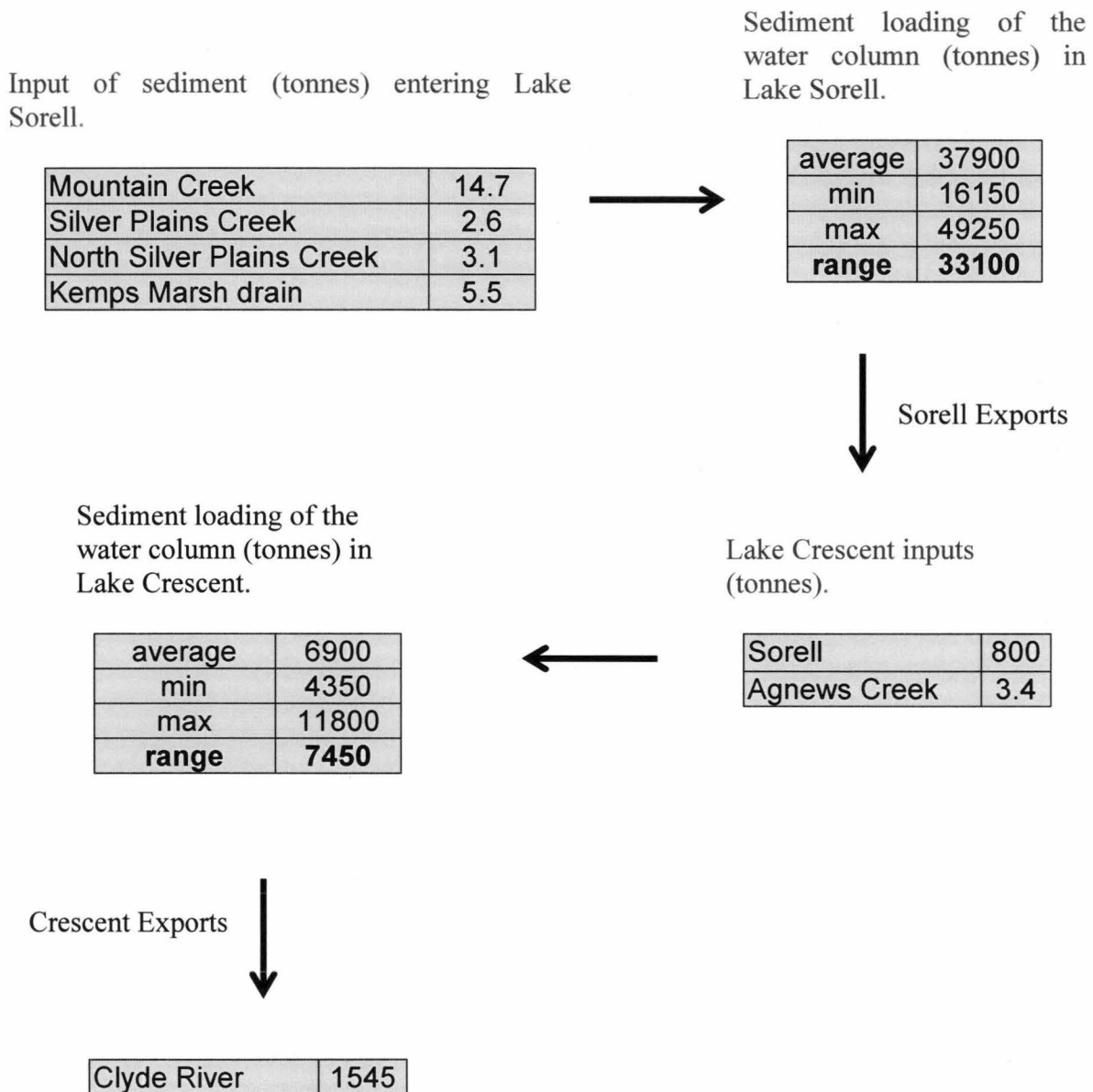


**Figure 2**      Graphical representation of the linear relationship between measured ‘in-situ’ turbidity and suspended solids for Lake Sorell.



## Appendix 8      Sediment and nutrient budgets for both lakes for 2000 and 2001.

Particulate and colloidal (total) sediment budget (tonnes) for 2000:



### Overview:

Total sediment	Sorell	Crescent
inputs (tonnes)	26	803.4
internal change (+/-)	33100	7450
exports (tonnes)	800	1545

## Particulate sediment budget (tonnes) 2000:

Input of sediment (tonnes) entering Lake Sorell.

Mountain Creek	14.7
Silver Plains Creek	2.6
North Silver Plains Creek	3.1
Kemps Marsh drain	5.5

Sediment loading of the water column (tonnes) in Lake Sorell.

average	18100
min	3400
max	26550
<b>range</b>	<b>23150</b>



Sorell Exports

Sediment loading of the water column (tonnes) in Lake Crescent.

average	4450
min	2550
max	8900
<b>range</b>	<b>6350</b>

Lake Crescent inputs (tonnes).

Sorell	242.4
Agnews Creek	3.4



Crescent Exports

Clyde River	1061.6
-------------	--------

## Overview:

Particulate Sediment	Sorell	Crescent
inputs (tonnes)	25.9	245.8
internal change (+/-)	23150	6350
exports (tonnes)	242.4	1061.6

Colloidal sediment budget (tonnes) 2000:

Input of colloids (tonnes) entering Lake Sorell.

Mountain Creek	Negligible
Silver Plains Creek	Negligible
North Silver Plains Creek	Negligible
Kemps Marsh drain	Negligible



Colloid loading of the water column (tonnes) in Lake Sorell.

average	19750
min	12800
max	22750
range	9950



Sorell Exports

Colloid loading of the water column (tonnes) in Lake Crescent.

average	2450
min	1800
max	2900
range	1100



Lake Crescent inputs (tonnes).

Sorell	563.8
Agnews Creek	negligible

Crescent Exports

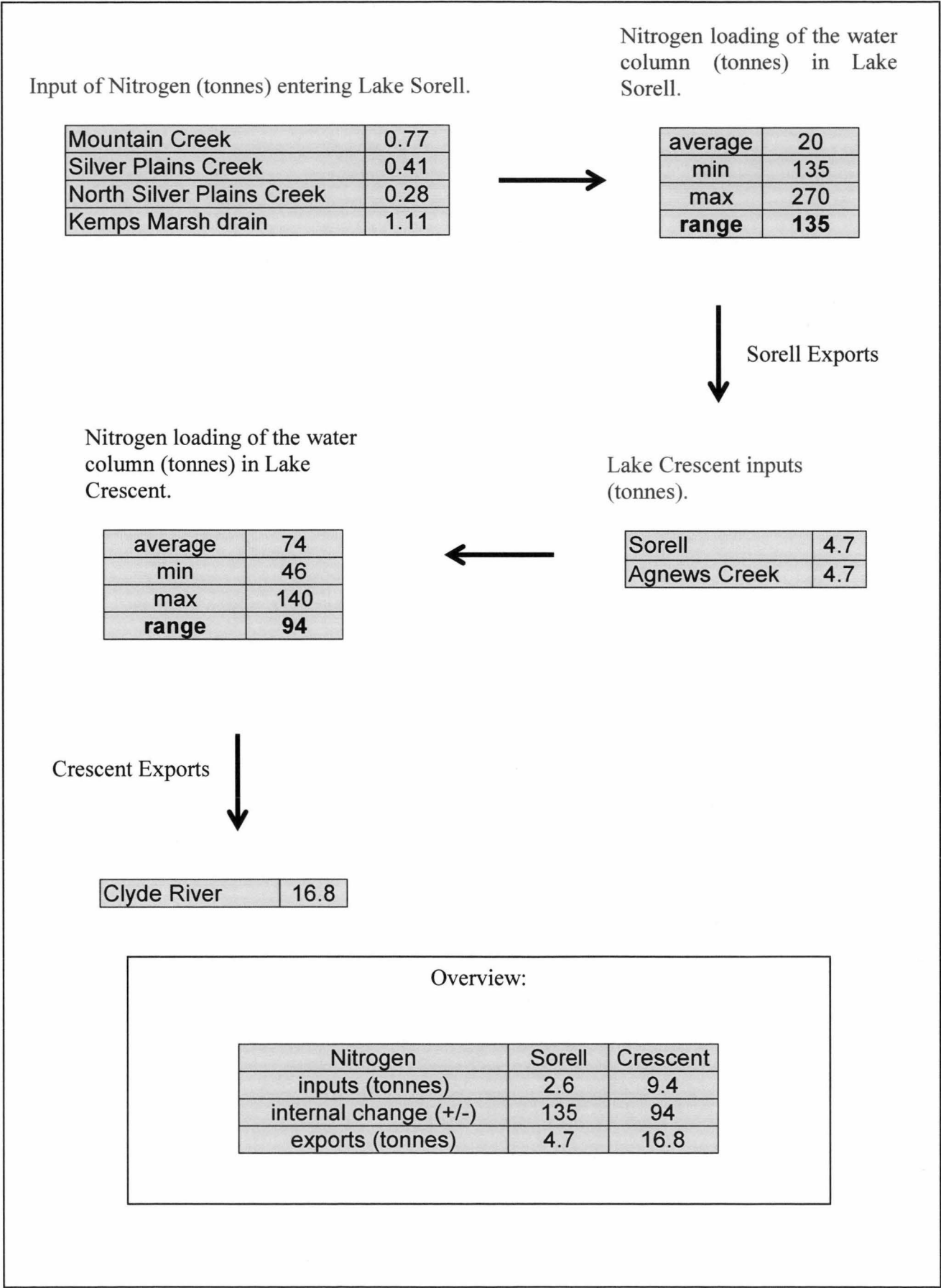


Clyde River	483.1
-------------	-------

Overview:

Colloids	Sorell	Crescent
inputs (tonnes)	N/A	563.8
internal change (+/-)	9950	1100
exports (tonnes)	563.8	483.1

Nitrogen budget (tonnes) 2000:



## Phosphorus budget (kgs) 2000:

Input of Phosphorus (kgs) entering Lake Sorell.

Mountain Creek	76
Silver Plains Creek	21
North Silver Plains Creek	14
Kemps Marsh drain	20



Phosphorus loading of the water column (kgs) in Lake Sorell.

average	16200
min	9400
max	23350
<b>range</b>	<b>13950</b>

Sorell Exports



Phosphorus loading of the water column (kgs) in Lake Crescent.

average	3550
min	2050
max	7700
<b>range</b>	<b>5650</b>

Lake Crescent inputs (kgs).

Sorell	350
Agnews Creek	29



Crescent Exports



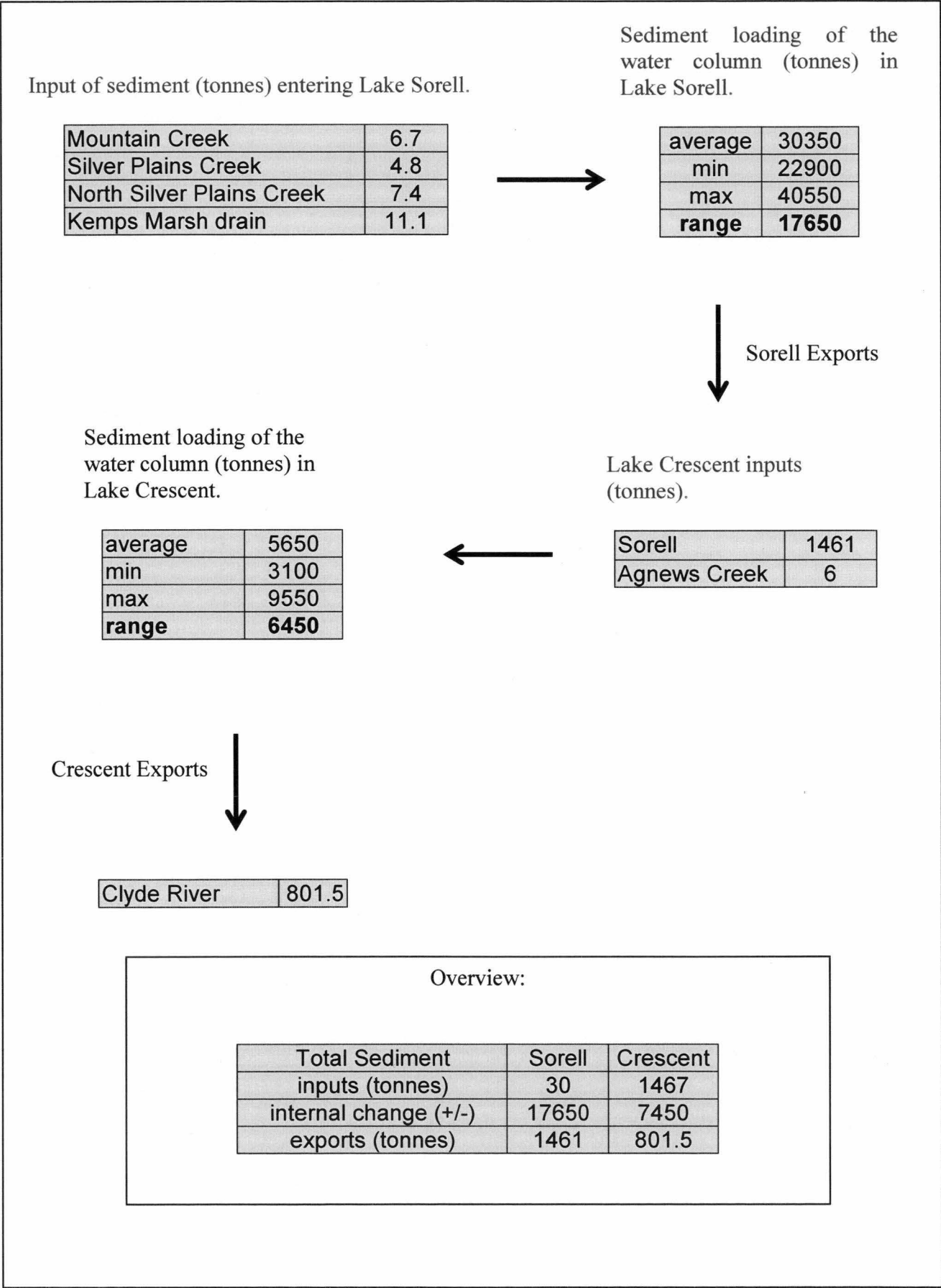
Clyde River	820
-------------	-----

Overview:

Phosphorus	Sorell	Crescent
inputs (kg)	131	379
internal change (+/-)	13950	5650
exports (kg)	350	820



Particulate and colloidal (total) sediment budget (tonnes) 2001:





Particulate sediment budget (tonnes) 2001:

Input of sediment (tonnes) entering Lake Sorell.

Mountain Creek	6.7
Silver Plains Creek	4.8
North Silver Plains Creek	7.4
Kemps Marsh drain	11.1



Sediment loading of the water column (tonnes) in Lake Sorell.

average	7900
min	3050
max	15300
range	12250



Sorell Exports

Sediment loading of the water column (tonnes) in Lake Crescent.

average	3650
min	1700
max	6800
range	5100



Lake Crescent inputs (tonnes).

Sorell	280
Agnews Creek	6

Crescent Exports

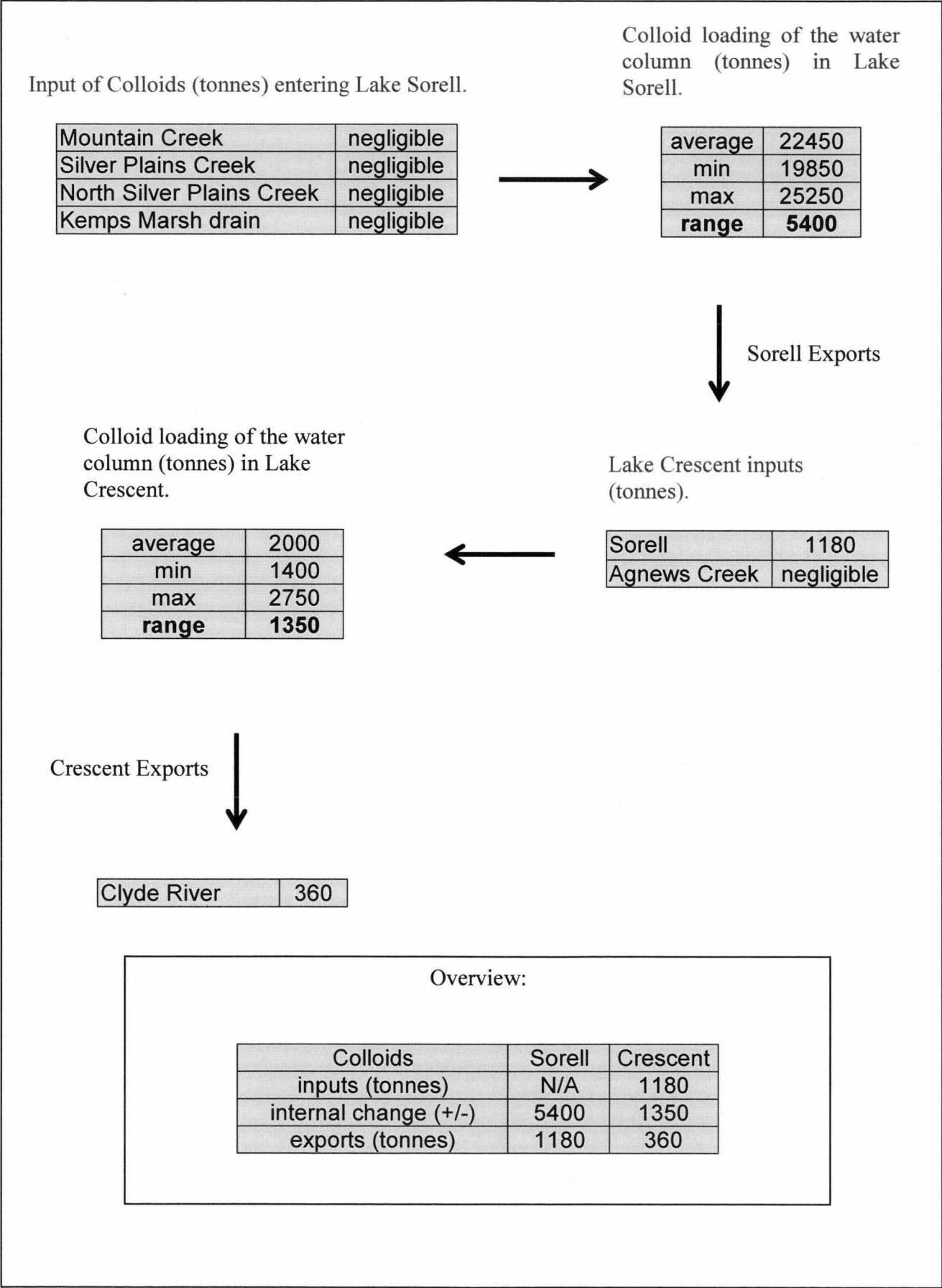


Clyde River	441.2
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Overview:

Particulate Sediment	Sorell	Crescent
inputs (tonnes)	30.0	286
internal change (+/-)	12250	5100
exports (tonnes)	280	441.2

Colloidal sediment budget (tonnes) 2001:



Nitrogen budget (tonnes) 2001:

Input of Nitrogen (tonnes) entering Lake Sorell.

Mountain Creek	1.33
Silver Plains Creek	0.82
North Silver Plains Creek	0.66
Kemps Marsh drain	2.00

Nitrogen loading of the water column (tonnes) in Lake Sorell.

average	187
min	134
max	228
range	94



Sorell Exports



Nitrogen loading of the water column (tonnes) in Lake Crescent.

average	69
min	39
max	121
range	82

Lake Crescent inputs (tonnes).

Sorell	12.6
Agnews Creek	4.5



Crescent Exports



Clyde River	12.3
-------------	------

Overview:

Nitrogen	Sorell	Crescent
inputs (tonnes)	4.8	17.1
internal change (+/-)	94	82
exports (tonnes)	12.6	12.3

Phosphorus (kgs) budget 2001:

Input of Phosphorus (kgs) entering Lake Sorell.

Mountain Creek	67
Silver Plains Creek	43
North Silver Plains Creek	34
Kemps Marsh drain	41



Phosphorus loading of the water column (kgs) in Lake Sorell.

average	14150
min	6900
max	19550
<b>range</b>	<b>12650</b>



Sorell Exports

Sediment loading of the water column (kgs) in Lake Crescent.

average	3350
min	1700
max	6700
<b>range</b>	<b>5000</b>



Lake Crescent inputs (kgs).

Sorell	349
Agnews Creek	48

Crescent Exports



Clyde River	536
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Overview:

Phosphorus	Sorell	Crescent
inputs (kg)	185	397
internal change (+/-)	12650	5000
exports (kg)	349	536

## Appendix 9      Colloidal nutrient contribution to the total nutrient pool.

Lake Sorell Total Nitrogen						
Date	Turbidity (NTU)	Filtered Turbidity (NTU)	Total N (mg/L)	Colloidal TN (mg/L)	Particulate TN (mg/L)	Colloidal TN/Total TN
11/04/02	94	63.6	2.72	1.38	1.34	51%
16/06/02	142	64.6	4.17	1.32	2.85	32%
10/07/02	158	69.2	3.46	1.23	2.23	36%
25/07/02	147	67.2	2.95	1.28	1.67	43%
Average	135	66.2	3.33	1.30	2.02	
St. Dev	28	2.5	0.64	0.06	0.66	

Lake Sorell Total Phosphorus							
Date	Turbidity (NTU)	Filtered Turbidity (NTU)	Total P (mg/L)	Colloidal TP (mg/L)	Particulate TP (mg/L)	Soluble Reactive P (mg/L)	Colloidal TP/Total TP
11/04/02	94	63.6	0.128	0.033	0.095	0.004	26%
16/06/02	142	64.6	0.199	0.042	0.157	0.003	21%
10/07/02	158	69.2	0.122	0.037	0.085	0.008	30%
25/07/02	147	67.2	0.101	0.036	0.065	0.003	36%
Average	135	66.2	0.138	0.037	0.101	0.005	
St. Dev	28	2.5	0.043	0.004	0.040	0.002	

Lake Crescent Total Nitrogen						
Date	Turbidity (NTU)	Filtered Turbidity (NTU)	Total N (mg/L)	Colloidal TN (mg/L)	Particulate TN (mg/L)	Colloidal TN/Total TN
11/04/02	119	44.6	2.54	1.42	1.12	56%
16/06/02	173	44.5	2.22	1.45	0.77	65%
11/07/02	167	46.6	2.04	1.53	0.51	75%
26/07/02	169	47.2	1.73	1.32	0.41	76%
Average	157	45.7	2.13	1.43	0.70	
St. Dev	25	1.4	0.34	0.09	0.32	

Lake Crescent Total Phosphorus							
Date	Turbidity (NTU)	Filtered Turbidity (NTU)	Total P (mg/L)	Colloidal TP (mg/L)	Particulate TP (mg/L)	Soluble Reactive P (mg/L)	Colloidal TP/Total TP
11/04/02	119	44.6	0.128	0.069	0.059	0.003	54%
16/06/02	173	44.5	0.189	0.080	0.109	0.005	42%
11/07/02	167	46.6	0.155	0.088	0.067	0.006	57%
26/07/02	169	47.2	0.142	0.080	0.062	0.004	56%
Average	157	45.7	0.154	0.079	0.074	0.005	
St. Dev	25	1.4	0.026	0.008	0.023	0.001	

Appendix 10      Lake Level Changes for Lake Sorell and Lake Crescent, 1970 – 2004.

Figure 1: Lake level variation (mASL) in Lake Sorell, 1970 to July 2004.

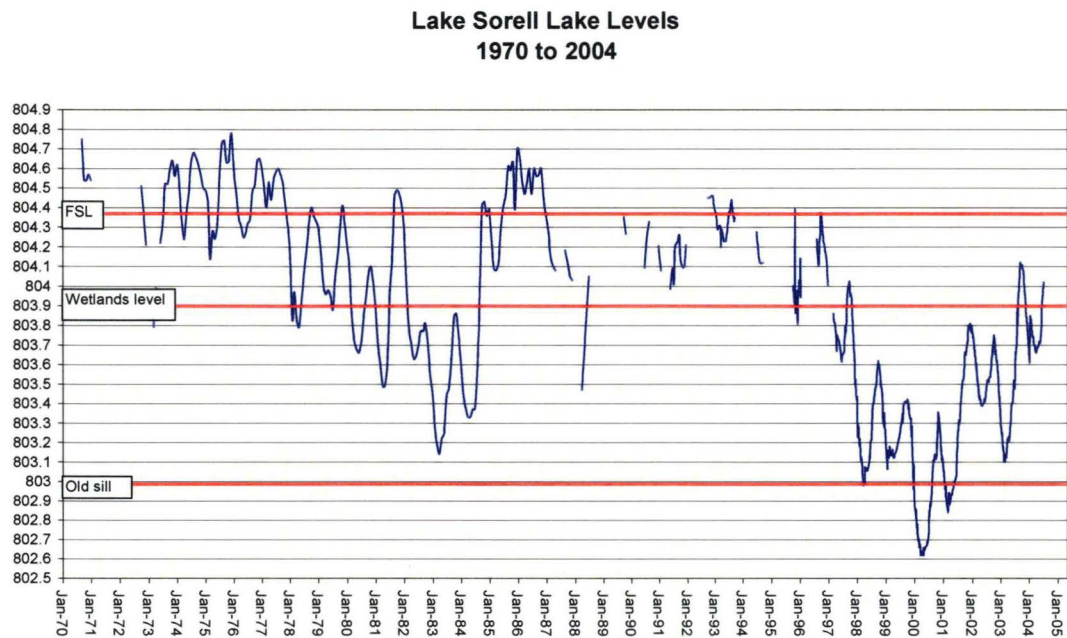


Figure 2: Lake level variation (mASL) in Lake Crescent, 1970 to July 2004.

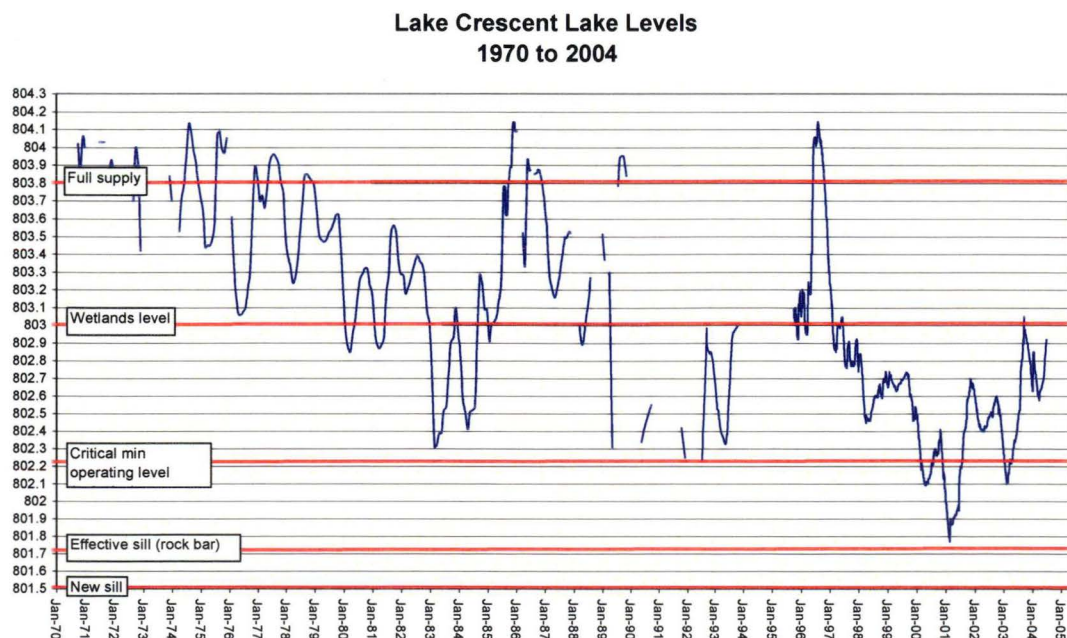




Figure 3: Lake level variation (mASL) in Lake Sorell, 1997 to 2004.

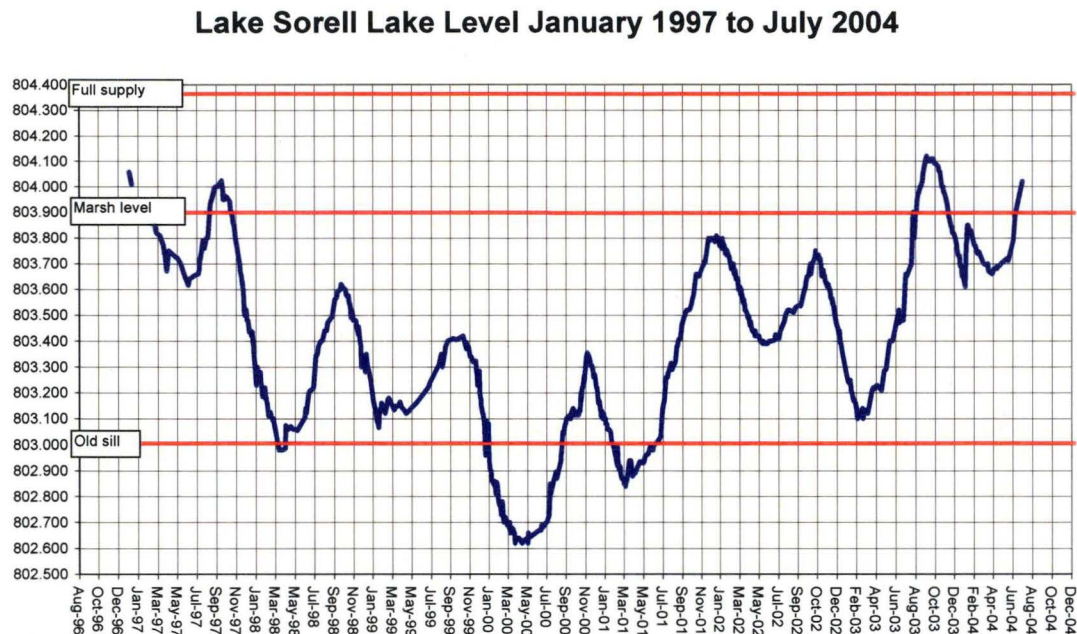


Figure 4: Lake level variation (mASL) in Lake Crescent, 1997 to 2004.

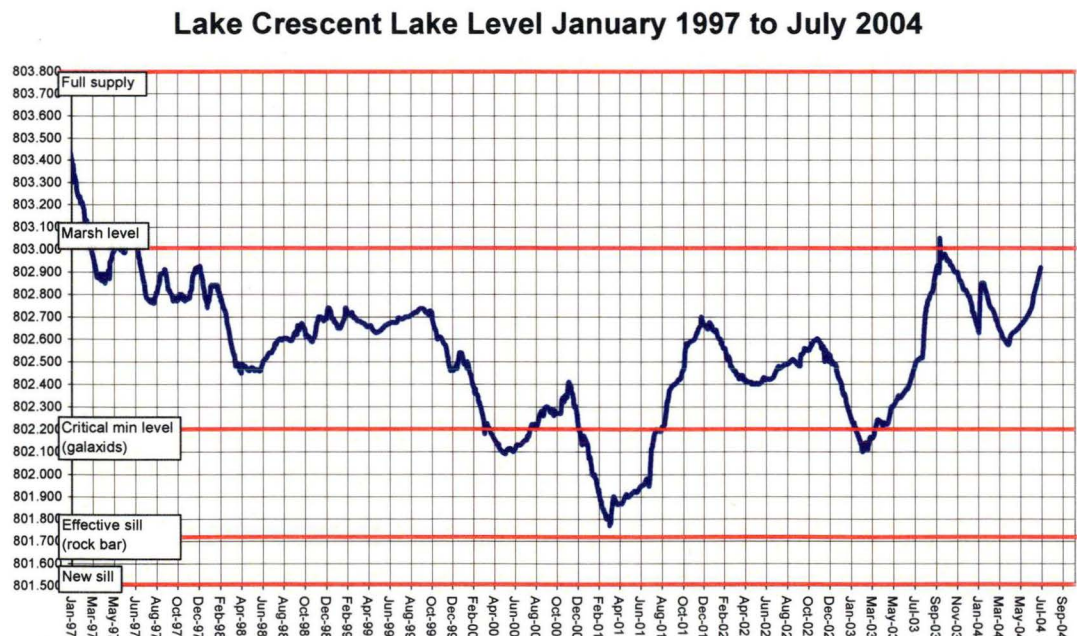


Figure 5: Dot-density and box plots depicting the spread in recorded lake level from January 1970 to December 1997 (Pre 1998); and January 1998 to July 2004 in lakes Crescent and Sorell.

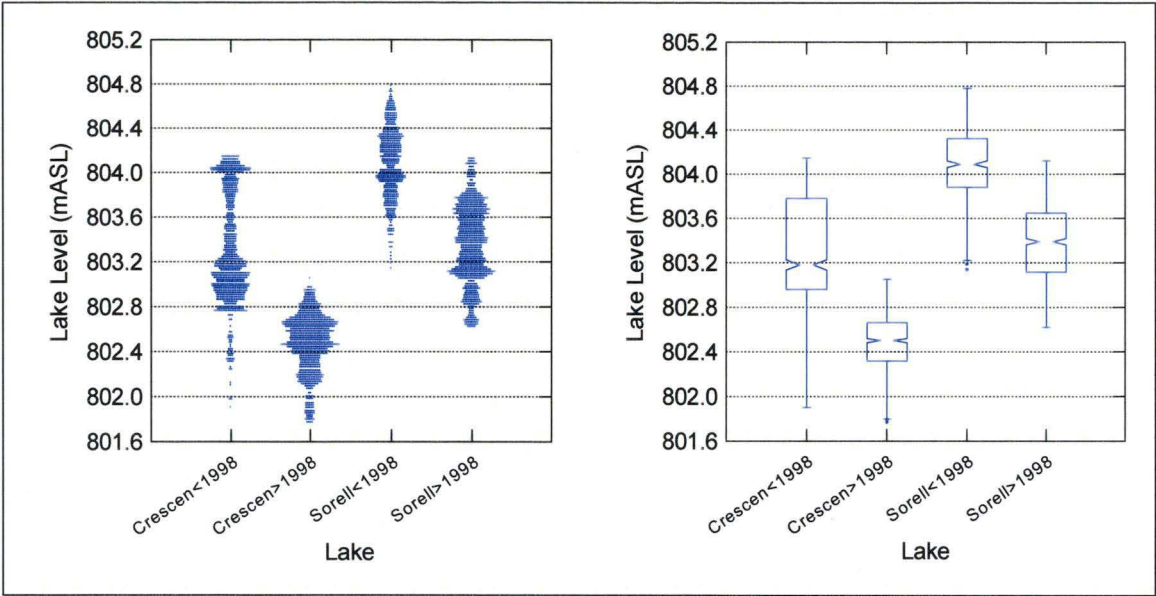


Table 1: Descriptive statistics summarising water levels in lakes Sorell and Crescent from January 1970 to December 1997 (Pre 1998); and January 1998 to July 2004.

Lake	Sorell Pre 1998	Sorell Post 1998	Crescent Pre 1998	Crescent Post 1998
Minimum	803.14	802.62	801.9	801.77
Lower hinge	803.88	803.12	802.96	802.32
Median	804.09	803.39	803.18	802.5
Upper hinge	804.32	803.65	803.78	802.66
Maximum	804.78	804.12	804.15	803.05
N	503	833	804	781
Mean	804.09	803.36	803.30	802.48
St. Dev	0.31	0.35	0.49	0.26



## Appendix 11      DYRESM-CAEDYM input files.

### DYRESM-CAEDYMMeteorological file .met

<#3>

Interlaken meteorological data. SW [ $\text{W m}^{-2}$ ], CC (fraction cloud cover)., Tair [Celsius], Pvpap [hPa], Uwind ( $\text{ms}^{-1}$ )., Rainfall (m).

86400 # Input met data time step (in seconds). Daily.

CLOUD\_COVER # Cloud cover option for LW

FIXED\_HT 14 # wind speed sensor type and height above lake bed (assuming metres).

YrDayNum	SW	CC(frac).	Tair	Pvpap	Uwind	Rainfall
2000001	388.65	0.2	9.5	8.2	4.6	0
2000002	334.49	0.5	11.6	9.42	5.41	0
2000003	282.38	0.5	12.2	10.46	7.34	0
2000004	35.13	0.8	9.8	10.26	6.11	0
2000005	239.89	0.7	7.2	8.35	5.97	0
2000006	407.94	0.2	10.6	8.14	4.53	0
2000007	229.64	0.7	11.6	9.25	4.36	0
2000008	256.87	0.5	13.3	10.32	4.55	0
2000009	388.32	0.1	12.8	9.77	6.34	0
2000010	299.34	0	15.6	11.68	5.43	0
2000011	352.36	0.3	17.1	12.01	4.24	0
2000012	407.64	0	14.7	11.21	5.84	0
2000013	396.49	0.1	15.2	10.74	6.29	0
2000014	348.98	0.5	14.7	10.86	6.81	0
2000015	271.15	0.3	15.5	12.63	4.74	0
2000016	399.11	0.1	13.9	11.05	5.05	0
2000017	391.04	0.1	14.1	8.53	4.2	0
2000018	187.36	0.8	14.4	10.12	5.68	0.0002
2000019	283.89	0.6	14	12.37	5.77	0.0002
2000020	232.98	0.8	12.4	11.46	7.72	0.0002
2000021	216.04	0.7	9.1	9.75	5.98	0.0002
2000022	242.26	0.7	6.4	7.75	5.54	0.0018
2000023	238.51	0.7	5.2	7.33	5.19	0.0010
2000024	322.47	0.4	8.5	7.41	4.6	0
2000025	173.36	0.7	10.1	8.91	4.34	0.0004
2000026	95.96	0.8	8.5	9.49	3.98	0.018
2000027	209.55	0.7	9.8	10.93	4.94	0.0024
2000028	288.15	0.5	10.9	9.91	4.7	0
2000029	210.4	0.8	7.6	9.17	6.09	0
2000030	356.32	0.3	10.1	7.92	4.31	0
2000031	323.19	0.7	11.5	9.63	4.18	0

**DYRESM-CAEDYM morphometry file .stg**

```

<#3>
Lake Crescent morphometry file
-42                                # latitude
800.2                             # height of base? above MSL
2                                 # number of streams
SURF 70      0.13  0.016 Inter_Canal # info of stream (surf/sub-surf,
                                half-ang, slope, drag coef., name).
SURF 70      0.13  0.016 Agnews      # info of stream (surf/sub-surf,
                                half-ang, slope, drag coef., name).

0      # base elevation (m). or elevation of lake bottom
3.6    # full supply elevation (m). or crest elevation
1      # number of outlets
1.3    # outlet heights
18     # number of bathymetry records
Height(m).  Area(m2).
0.4        1058000
0.6        6383000
0.8        9889000
1.0        12680000
1.2        14180000
1.4        14840000
1.6        15441500
1.8        15770000
2.4        16250000
2.6        16730000
2.8        17225000
3.0        17720000
3.2        18285000
3.4        18850000
3.6        19430000
3.8        20010000
4.0        21540000
4.2        23070000

```

**DYRESM-CAEDYM inflow file .inf**

Lake Sorell Inflow file - 2 years from day 1, 2000 to day 365, 2001

```

4          # of inflows
MtCk       # Name of inflow 1
SlvrPlns   # Name of inflow 2
NslvrPlns  # Name of inflow 3
Kemps      # Name of inflow 4
YrDayNum   InfNum  VOLUME    TEMP    SAL    TN    TP    SSOL1  SSOL2
2000240    1      133570.92    3.88    0.00    0.13  0.01  0.90  0.00
2000240    2      31551.30     3.88    0.00    0.22  0.01  1.42  0.00
2000240    3      16006.00     3.88    0.00    0.29  0.02  3.31  0.00
2000240    4        0.00     3.88    0.00    0.00  0.00  0.00  0.00
2000241    1     101817.21     4.39    0.00    0.12  0.01  0.90  0.00
2000241    2     24290.19     4.39    0.00    0.20  0.01  1.17  0.00
2000241    3     12321.63     4.39    0.00    0.26  0.01  2.78  0.00
2000241    4        0.00     4.39    0.00    0.00  0.00  0.00  0.00
2000242    1     80070.40     4.66    0.00    0.11  0.01  0.90  0.00
2000242    2     19607.12     4.66    0.00    0.18  0.01  1.01  0.00
2000242    3     9883.16      4.66    0.00    0.24  0.01  2.44  0.00
2000242    4        0.00     4.66    0.00    0.00  0.00  0.00  0.00
2000243    1     67755.43     4.82    0.00    0.11  0.01  0.90  0.00
2000243    2     16215.08     4.82    0.00    0.18  0.01  0.89  0.00
2000243    3     8084.29      4.82    0.00    0.23  0.01  2.18  0.00
2000243    4        0.00     4.82    0.00    0.00  0.00  0.00  0.00
2000244    1     57165.64     4.85    0.00    0.10  0.01  0.90  0.00
2000244    2     13657.30     4.85    0.00    0.17  0.01  0.82  0.00
2000244    3     6263.24      4.85    0.00    0.22  0.01  1.96  0.00
2000244    4        0.00     4.85    0.00    0.00  0.00  0.00  0.00
2000245    1     45464.02     4.88    0.00    0.10  0.01  0.90  0.00
2000245    2     12645.96     4.88    0.00    0.17  0.01  0.77  0.00
2000245    3     4584.51      4.88    0.00    0.21  0.01  1.68  0.00
2000245    4     2681.75      4.88    0.00    0.72  0.02  5.48  0.00

```

**DYRESM-CAEDYM outflow file .wdr**

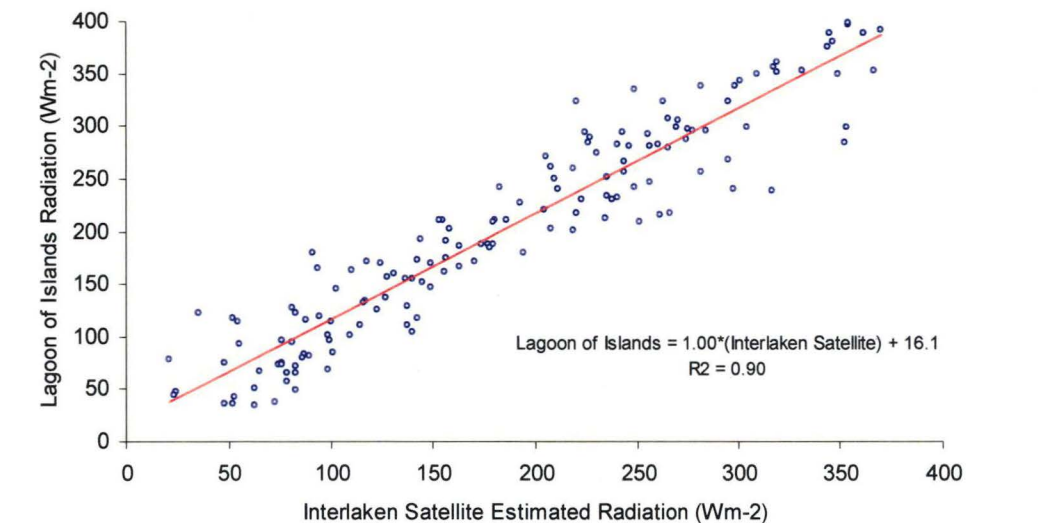
Lake Crescent withdrawal file

1 # Number of outlets. Summer release, 2yrs.

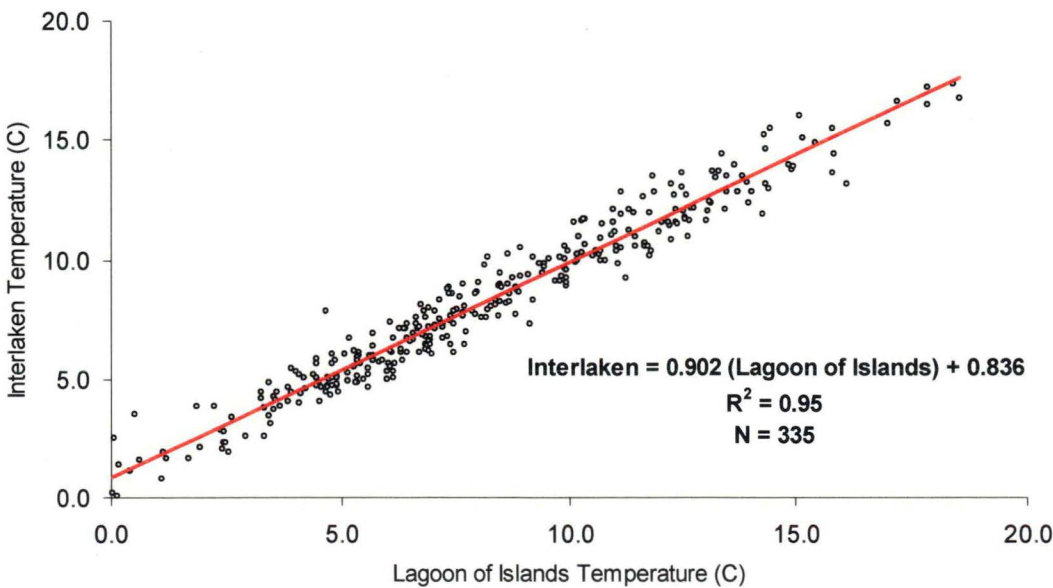
YrDayNum	Outflow
2000320	0
2000321	0
2000322	0
2000323	0
2000324	0
2000325	0
2000326	0
2000327	0
2000328	0
2000329	7782.3936
2000330	31540.1472
2000331	66625.8048
2000332	73361.7216
2000333	51545.2032
2000334	36676.368
2000335	43543.1808
2000336	47253.1968
2000337	46614.7008
2000338	45749.7504
2000339	54065.232
2000340	55883.3472

**Appendix 12      Interlaken and Lagoon of Islands radiation, temperature and wind speed regressions.**

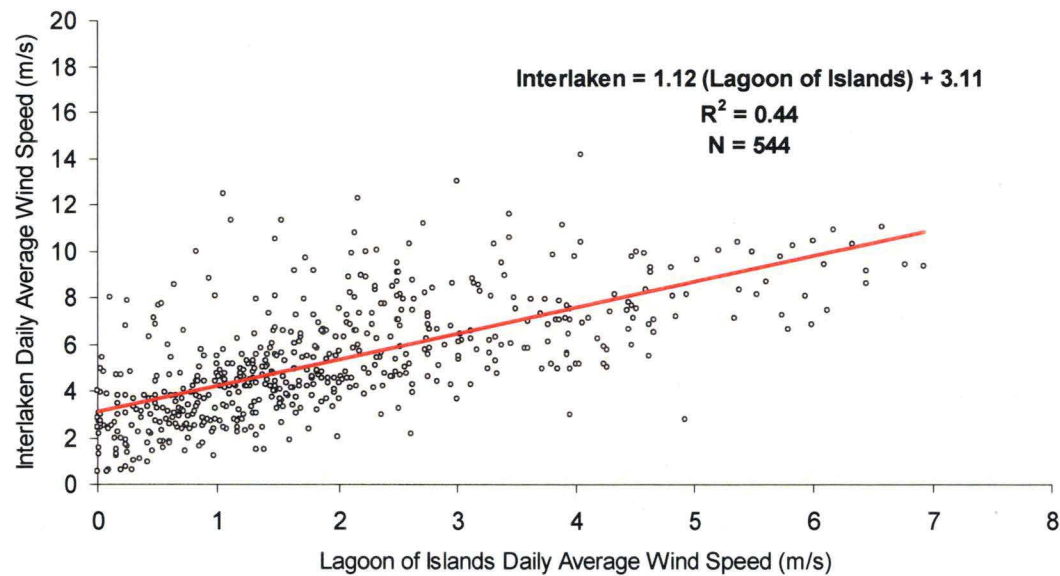
**Figure 1:** Linear regression analysis of daily average Lagoon of Islands radiation versus daily average radiation estimated from satellite at Interlaken.



**Figure 2:** Linear regression analysis of daily average Interlaken air temperature versus daily average Lagoon of Islands air temperature.

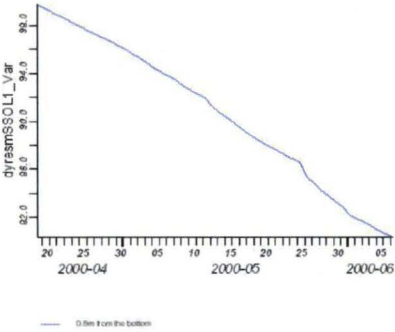
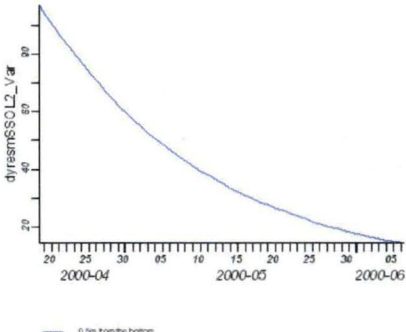
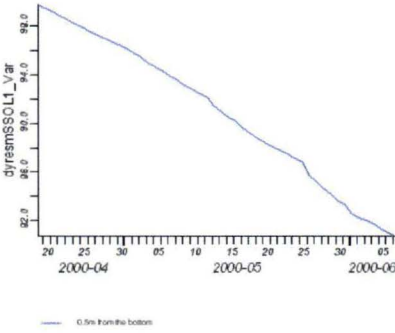
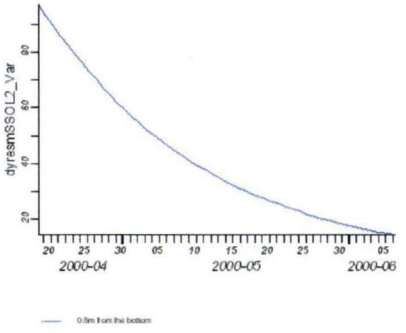


**Figure 3:** Linear regression analysis of daily average Interlaken wind speed versus daily average Lagoon of Islands wind speed.



## Appendix 13 Calibration of particle size and resuspension rate.

**Table 1** Overview of the influence of changing particle size and Alpha S (resuspension rate constant) on the flux of sediments. The modelling run is for a period from April 19<sup>th</sup> 2000 to June 6<sup>th</sup> 2000 in Lake Crescent.

AlpS ( $\text{gm}^{-2}\text{s}^{-1}$ )	0.35 $\mu\text{m}$	1.0 $\mu\text{m}$
$0.5 * 10^{-6}$		
$0.5 * 10^{-5}$		
$0.5 * 10^{-4}$	