Sediment resuspension and water quality during declining water levels in a shallow lake: a case study of Lake Alexandrina, South Australia

Dominic Skinner

A thesis submitted in fulfillment of the Doctor of Philosophy

School of Earth and Environmental Sciences
University of Adelaide, South Australia

November 2011
# Table of Contents

Abstract .......................................................................................................................... vi
Acknowledgements ....................................................................................................... viii
Declaration ................................................................................................................... x
List of abbreviations ................................................................................................... xi
List of figures ............................................................................................................... xiii
List of tables ............................................................................................................... xviii
Preface ...................................................................................................................... xxi

1. Introduction .......................................................................................................... 1
   1.1 Stressors in aquatic ecosystems ...................................................................... 1
   1.2 Sediment resuspension in shallow lakes ...................................................... 2
      1.2.1 Sedimentation and resuspension ............................................................ 2
      1.2.2 Impacts of sediment resuspension on aquatic ecosystems ...................... 4
      1.2.3 Water level decline and sediment resuspension ...................................... 5
   1.3 Thesis outline ................................................................................................... 8

2. The Murray-Darling Basin and Lower Lakes .................................................. 10
   2.1 The Murray-Darling Basin ......................................................................... 10
      2.1.1 Catchment characteristics .................................................................... 10
      2.1.2 Drought in the Murray-Darling Basin ................................................. 11
   2.2 The Lower Lakes – Lake Alexandrina and Lake Albert ......................... 12
      2.2.1 Site description of the Lower Lakes .................................................... 12
      2.2.2 Water Quality ...................................................................................... 13
      2.2.3 Impacts of regulation .......................................................................... 15
      2.2.4 Recent changes .................................................................................... 16

3. Water quality changes during water level drawdown in two regulated shallow lakes ........................................................................................................... 23
   3.1 Introduction ................................................................................................... 23
   3.2 Methods ....................................................................................................... 25
      3.2.1 Site description ...................................................................................... 25
      3.2.2 Monitoring water quality ..................................................................... 26
3.2.3 Water balance ................................................................. 28
3.2.4 Monthly nutrient and suspended particulate matter balance .......... 29

3.3 Results .................................................................................. 31
3.3.1 Water balance ................................................................. 31
3.3.2 Changes to water quality during water level decline .................. 32
3.3.3 Mass balance of TP, FRP and SPM ....................................... 33

3.4 Discussion ............................................................................. 34
3.4.1 Validity of water balance ................................................... 34
3.4.2 Influence of water level decline on water quality ..................... 37

4. Effects of water level decline on the composition and distribution of surface sediments in the large, shallow Lake Alexandrina, South Australia .......... 53

4.1 Introduction ........................................................................... 53
4.2 Methods ............................................................................... 55
  4.2.1 Site description ................................................................. 55
  4.2.2 Sediment analysis ............................................................ 56
  4.2.3 Water sampling ................................................................. 58
  4.2.4 Data analysis ................................................................. 58

4.3. Results ................................................................................ 59
  4.3.1 Sediment composition and bimodality .................................. 59
  4.3.2 Changes to sediment particle size distribution ....................... 61
  4.3.3 Chemical characteristics of sediments .................................. 61
  4.3.4 Calcium carbonate saturation index .................................... 62

4.4. Discussion ........................................................................... 63
  4.4.1 Climate and lake water level declines .................................. 63
  4.4.2 Changes to sediment composition and distribution during water level declines ............................................. 63
  4.4.3 Loss of inorganic carbon in peripheral sediments .................. 66

5. Fine sediment dynamics in a large, shallow lake following water level decline ......................................................................................................................... 80

5.1 Introduction ........................................................................... 80
5.2 Methods ............................................................................... 83
  5.2.1 Site description ................................................................. 83
  5.2.2 Sediment traps ................................................................. 83
  5.2.3 Calculating the depth of the wave-mixed layer ....................... 86

5.3 Results ................................................................................ 88
  5.3.1 Rate of sedimentation and resuspension ............................. 88
  5.3.2 Type of sediments undergoing resuspension ....................... 89
  5.3.3 Relationship between $D_{WML}$ and turbidity ....................... 89
  5.3.4 Rate of nutrient deposition ................................................. 90
5.4 Discussion ......................................................................................................... 90
  5.4.1 Quantifying sediment resuspension in Lake Alexandrina .................. 90
  5.4.2 Dynamics of sediment resuspension in Lake Alexandrina ................. 92

6. The influence of sediment resuspension on light availability in a shallow lake following extreme water level decline: implications for lake productivity ... 102
   6.1 Introduction .................................................................................................... 102
   6.2 Methods ....................................................................................................... 104
     6.2.1 Study site ................................................................................................. 104
     6.2.2 Instrument deployment ........................................................................... 105
     6.2.3 Water samples ......................................................................................... 108
     6.2.4 Determining critical shear stress .............................................................. 108
   6.3 Results ........................................................................................................... 111
     6.3.1 Wind and wave induced sediment resuspension .................................. 111
     6.3.2 Estimates of critical shear stress .............................................................. 111
     6.3.3 Sediment resuspension and particle size characteristics ....................... 112
     6.3.4 Influence of sediment resuspension on light attenuation ...................... 113
     6.3.5 Dissolved oxygen profiles ....................................................................... 114
   6.4 Discussion ..................................................................................................... 114
     6.4.1 Physical conditions ................................................................................. 114
     6.4.2 Sediment resuspension ........................................................................... 115
     6.4.3 Light availability for primary productivity ............................................. 117
     6.4.4 Light limitation in the pelagic ................................................................. 118
     6.4.5 Proportion of littoral sediments after water level declines .................. 121

7. The frequency of wind-driven sediment resuspension under different water levels in Lake Alexandrina ................................................................. 135
   7.1 Introduction ................................................................................................... 135
   7.2 Methods ....................................................................................................... 138
     7.2.1 Magnitude, frequency and periodicity of wind on Lake Alexandrina ....... 138
     7.2.2 Hydrodynamic modeling (ELCOM-CAEDYM) ....................................... 139
     7.2.3 Modelling sedimentation in shallow lakes – a variant of the Courant-Friedrichs-Lewy constraint ................................................................. 141
     7.2.4 Statistical analysis ................................................................................... 142
   7.3 Results ........................................................................................................... 142
     7.3.1 Wind conditions and temperature stratification ...................................... 142
     7.3.2 Modeled orbital velocity at different water levels .................................... 143
     7.3.3 Critical wind speed, sediment resuspension and water level ............... 144
     7.3.4 Frequency of sediment resuspension ...................................................... 145
   7.4 Discussion ..................................................................................................... 145
     7.4.1 Temperature stratification and wind driven sediment resuspension ..... 145
     7.4.2 Diel periodicity ......................................................................................... 146
7.4.3 Changing water levels ................................................................. 148
7.4.4 Water level declines and water clarity restoration ..................... 148

8. Discussion .......................................................................................... 163

8.1 Effects of water level decline in shallow lakes ................................ 163
  8.1.1 Water use, climate change and water level decline ...................... 163
  8.1.2 Water level decline influences stable ecosystem states ................ 164
  8.1.3 Influence of drawdown on the species distribution of primary producers in shallow lakes .................................................... 167

8.2 Resuspension and redistribution of sediments in large, shallow lakes .... 168

8.3 Implications for the management of Lake Alexandrina .................... 171

A.1. Method selection when estimating the proportion of settling particles attributable to sediment resuspension ................................... 178
  A.1.1 The label method ................................................................. 178
  A.1.2 The TIPM/TPM method ....................................................... 179
  A.1.3 Correcting with long-term accumulation rates ......................... 180
  A.1.4 Method comparison and optimization for Lake Alexandrina .......... 181

A.2. The effect of salinity on flocculation and sedimentation rates ........ 186
  A.2.1 The SETCOL technique ...................................................... 186
  A.2.2 Settling velocity and floc size .............................................. 187

References ............................................................................................. 193
Abstract

Shallow lakes have a tendency to be present in one of two broad alternative states: clear or turbid. Increased demand for water by humans, drought and climatic shifts may reduce water availability and increase the frequency or magnitude of lake water-level drawdown. However, there is conflicting evidence as to how shallow lakes respond to water level decline. Many lakes become clearer due to reduced nutrient inputs from inflowing rivers. Alternatively, internal processes, such as the resuspension of sediments, can exacerbate the turbid state as lower water levels attenuate less wind-induced turbulence before it reaches the sediment surface. A better understanding of responses of lake water quality to drawdown will greatly improve the ability of water managers to maximise the ecological benefits of drawdowns, while minimising the adverse consequences.

Severe drought in the Murray-Darling Basin, Australia’s largest river system, resulted in extreme water level decline in the two end of system lakes, Lake Alexandrina and Lake Albert (the Lower Lakes). Water depth in Lake Alexandrina dropped from a mean of 2.9 metres in summer 2006/07 to 1.3 metres two years later, where water levels remained until April 2010 when floods refilled the lakes. This provided a unique opportunity to assess the influence of water level decline on lake characteristics. The aims of this work were to firstly elucidate the effects of water level decline on sediment resuspension and redistribution, accompanying nutrient changes and light availability. Secondly, the study sought to understand the interactions between water level decline and sediment resuspension in the context of alternative stable states during periods of drought.

As water level fell in the Lower Lakes, the water quality became increasingly brackish as mean salinity rose from 0.6 g L\(^{-1}\) to 3.7 g L\(^{-1}\). The concentration of suspended particulate matter (SPM) and associated nitrogen and phosphorus concentrations increased as water levels declined, whereas soluble phosphorus did not. A mass
balance showed that month to month variation in SPM could not be explained by the sum of inputs and outputs, suggesting that sediment resuspension was an important process in Lake Alexandrina at all water levels. Surveys of the size distribution of surface sediments suggested that focusing of fine sediments was more prevalent when water levels were higher. Analysis of gross sedimentation using sediment traps supported the hypothesis that the resuspension and redistribution of sediments had become spatially homogeneous as water levels declined.

High-resolution data collected from the centre of Lake Alexandrina suggested that sediment resuspension frequently occurred below even the deepest water. At bed shear stresses greater than 0.031 N m$^{-2}$ particles with a nominal diameter of 26 µm were resuspended. Under low water levels, sediment resuspension strongly influenced the depth of light penetration into the water column (euphotic depth ranged between 0.68 and 0.91 metres). However, lower water levels led to an increase in the average irradiance through the water column compared with previous studies at higher water levels.

Hydrodynamic modeling was used to quantify the influence of wind speed on sediment resuspension. This showed that wind speeds of 7.7 m s$^{-1}$ were required to resuspend particles with a nominal diameter of 26 µm when water levels were high. However, at low water levels, the wind speed imparting critical shear velocity at the lake-bed decreased to 2.4 m s$^{-1}$. The frequency distribution of wind speed suggested that sediments were being resuspended only 27% of the time when water levels were high, but over 87% of the time at the lowest water levels studied.

The shallow morphology of Lake Alexandrina, as well as the high wind-induced turbulent energy at the sediment surface due to its large surface area, increased the resuspension and redistribution of fine sediments during water level drawdown. For Lake Alexandrina, the internal process of sediment resuspension overcame the drought-induced reduction in nutrient inputs to result in an increasingly turbid system. These results imply that lake management should centre on the reduction of external nutrients, as well as biomanipulation of the food webs to increase the water clarity of Lake Alexandrina and improve conditions for macrophyte development.
My path prior to undertaking this thesis was in no way linear. Fortunately, limnology is a diverse field, suited as much to a generalist scientist willing to acquire diverse skills, as it is to a specialist. Throughout this journey, many people graciously donated their time and expertise to assist my development of the skills I needed to engage usefully in this new field. To all, I am immensely thankful.

I would like to sincerely thank my primary supervisor, Justin Brookes, for his constant support and guidance; for seeing the breadth of my research interests as an asset, rather than a liability; for directing my wandering inquisition; and, for taking the time to introduce me to further ideas with immense enthusiasm. Thank you to Kane Aldridge, whose continued support was invaluable. Kane generously provided raw data for comparison and kept an open door to happily answer any and every question, regardless of how trivial. I would like to extend my sincere appreciation to Rod Oliver, for his unwavering commitment to scientific rigor, endless cross-examination and tireless determination to analyse and understand every last data point.

Throughout my doctoral studies, I received additional support and technical assistance from numerous others. I would firstly like to thank Rob Daly, for timely, logical answers to questions regarding computing, MatLab, data acquisition and limnology in general. I would also like to express my appreciation to Jason Antenucci, Brendan Busch, George Ganf, Matt Hipsey, Sebastien Lamontagne, Andrew Metcalfe, Luke Mosley, Ian Webster and Richard White for detailed answers to vague questions on a variety of topics.
The collection of data for this thesis took a large amount of time in the field and the lab, as well as a significant amount of encouragement and assistance from many others. Most notably, my thanks go to John and Alex Payne. John provided easy and affordable access to a boat and car in the crucial stage of my research, and helped lug equipment, collect samples, wash and label copious numbers of bottles with a smile. Alex contributed long hours in the field and helped with laboratory analysis promptly and thoroughly. I would like to particularly acknowledge Robert Brookes, for sharing extensive knowledge of Lake Albert and the Lower Lakes region, and for making cold, wet, early mornings entertaining, as only a fisherman can. Thank you to Abby, Alex, Annie, Dae Heui, Grace, John, Justin, Kane, Kelly, Malcolm, Nathan, Nick and Steve for help on various field trips and in the lab. I greatly appreciate all those that made the often tedious process of data collection, field trips and lab work entertaining.

My work was also supported through collaboration with people from external organizations. In particular, I greatly appreciate Mike Burch, Peter Hobson and Tim Kildea, for the generous loan of equipment and for giving me the opportunity to learn a great deal in the months preceding this PhD at the Australian Water Quality Centre. I would also like to thank Andrew Skinner and the folks at Measurement Engineering Australia, whose expertise and generosity with data, equipment and information were invaluable.

Lastly, and most importantly, I turn to those who have allowed me to continue to enjoy life outside of this thesis. My sincere thanks and love go to Kelly, Claudia, Andrew, Malcolm, Claire, Annie, Robert and Brendon. Finally, to Tim Minchin – hopefully, sanity prevailed!
Declaration

I, Dominic Skinner certify that this work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text.

I give consent to this copy of my thesis, when deposited in the University Library, being made available for loan and photocopying, subject to the provisions of the Copyright Act 1968.

I also give permission for the digital version of my thesis to be made available on the web, via the University’s digital research repository, the Library catalogue and also through web search engines, unless permission has been granted by the University to restrict access for a period of time.

Dominic Skinner 18/11/2011
List of abbreviations

AHD: Australian Height Datum, where zero equals the mean sea level recorded between 1966 and 1968
ANCOVA: Analysis of Covariance
ANOVA: Analysis of Variance
ASS: Acid sulfate soils
Chla: Chlorophyll a
CSIRO: Commonwealth Scientific and Industrial Research Organisation
DOC: Dissolved organic carbon
DWLBC: Department for Water, Land and Biodiversity Conservation
EC: Electrical conductivity
FRP: Filterable reactive phosphorus
GDP: Gross Domestic Product
Gt: Gigatonne (equal to 1000 Megatonnes)
IC: Inorganic carbon
ICP-AES: Inductively coupled plasma atomic emission spectrophotometry
ICP-OES: Inductively coupled plasma optical emission spectrophotometry
IGo: Loss-on-ignition (reported as a percentage of fresh weight)
LISST: Laser in-situ scattering and transmissometry
LOI: Loss-on-ignition (reported as a percentage of dry weight)
MDBA: Murray Darling Basin Authority
ML: Megalitre (equal to 1 000 000 litres)
Mt: Megatonne (equal to 1 000 000 tonnes)
NATA: National Association of Testing Authorities
OC: Organic carbon
pCO2: Partial pressure of carbon dioxide
PSD: Particle size distribution
PVC: Poly-vinyl chloride
RMSE: Root mean square error
SIPM: Suspended inorganic particulate matter
SOPM: Suspended organic particulate matter
SPM: Suspended particulate matter
TC: Total carbon
TIPM: Trapped inorganic particulate matter
TN: Total nitrogen
TOPM: Trapped organic particulate matter
TP: Total phosphorus
TPM: Trapped particulate matter
WC: Water content
\( Z_{\text{mean}} \): Mean depth (metres)
\( Z_{\text{max}} \): Maximum depth (metres)
List of figures

Figure 1.1: Timing of experiments relative to water level drawdown in Lake Alexandrina.................................................................9

Figure 2.1: The location of Lake Alexandrina (35.42784°S 139.16142°E) and Lake Albert within the Murray-Darling Basin.........................................................20

Figure 2.2: Timeline of significant management events during the drought-induced decline in water level of Lake Alexandrina (inset). ........................................21

Figure 2.3: Bathymetry of Lake Alexandrina, Lake Albert and the northern end of the Coorong.................................................................22

Figure 3.1: Water sampling sites and significant geographical features in the Lower Lakes. .........................................................................43

Figure 3.2: Changes in water level (—) and salinity (---) of Lake Alexandrina, January 2007-April 2010 .................................................................................44

Figure 3.3: Monthly water balance for Lake Alexandrina.............................................45

Figure 3.4: Monthly water balance for Lake Albert .....................................................46

Figure 3.5: Comparison between calculated water balance and measured water volume of both lakes. .................................................................47

Figure 3.6: The relationship between water quality parameters and water level in Lake Alexandrina (closed circles) and Lake Albert (open circles)..............48
Figure 3.7: Time series of chlorophyll $a$ data for the duration of the study........49

Figure 3.8: Mass balance of TP (top panel), FRP (middle panel) and SPM (bottom panel) in Lake Alexandrina...............................................................50

Figure 3.9: Mass balance of TP (top panel), FRP (middle panel) and SPM (bottom panel) in Lake Albert.................................................................51

Figure 3.10: Relationship between TP and SPM measured from integrated water samples in Lake Alexandrina.........................................................52

Figure 4.1: Sediment sampling sites in Lake Alexandrina........................................74

Figure 4.2: Sediment sampling dates relative to water level drawdown and increasing salinity in Lake Alexandrina........................................75

Figure 4.3: Relationship of water depth with water content (A) and organic matter as loss-on-ignition (B) of surface sediments in Lake Alexandrina. ..........76

Figure 4.4: Frequency distribution of sediment water content in 2007 (—) and in 2009 (- - -). ........................................................................................................77

Figure 4.5: Comparison between the mean particle size distribution in 2007 (circles) and 2009 (squares) of mud (closed symbols) and sand (open symbols) in Lake Alexandrina.................................................................78

Figure 4.6: Relationship between total carbon (TC) and loss-on-ignition (LOI) for sand and mud in 2007 and in 2009..................................................79

Figure 5.1: Location of deployed sediment traps within Lake Alexandrina.............97

Figure 5.2: Variation of gross sedimentation rate and depth of wave-mixed layer ($D_{WML}$).............................................................................................98
Figure 5.3: Particle size distribution of sediments, trap contents (TPM) and suspended particles (SPM) averaged across all deployment periods. 99

Figure 5.4: Time series of turbidity and the depth of the wave mixed layer calculated using linear wave theory for deep water. 100

Figure 5.5: Distribution of sedimentation with water depth in Lake Alexandrina as measured using all sites across changing water depths for the duration of the experiment. 101

Figure 6.1: Location of instrument deployment within Lake Alexandrina. 124

Figure 6.2: Period of measurements recorded using different instruments during the 70 hr intensive field campaign. 125

Figure 6.3: The influence of temperature stratification and wind speed on changes to turbidity. 126

Figure 6.4: Wind speed and direction recorded on Lake Alexandrina for the duration of the experiment. 127

Figure 6.5: Frequency spectra of three components of water velocity (u, v and w) measured by the Acoustic Doppler Velocimeter. 128

Figure 6.6: Rate of change in turbidity and the logarithm of the shear stress calculated from orbital velocity measurements with an Acoustic Doppler Velocimeter. 129

Figure 6.7: In situ particle size distribution of suspended sediments between 1 pm 21 March and 11 am 24 March 2010. 130

Figure 6.8: Time series of changes to suspended particles and the euphotic depth. 131
Figure 6.9: Vertical gradients of dissolved oxygen (top panel) compared with changes to wind speed and temperature stratification (bottom panel).........132

Figure 6.10: Average irradiance in the middle of Lake Alexandrina calculated using $z_{mean} = 1.3$ m (black diamonds) and $z_{max} = 2.2$ m (white squares).............134

Figure 7.1: Location of input parameters used to set up a three-dimensional hydrodynamic model (ELCOM-CAEDYM) for Lake Alexandrina.........152

Figure 7.2: Wavelet analysis of long-term wind data from Langhorne Creek......153

Figure 7.3: Cumulative frequency distribution of long-term wind record at Langhorne Creek..................................................................................................................154

Figure 7.4: Wavelet analysis of wind data from the northeast of Lake Alexandrina......................................................................................................................155

Figure 7.5: Cumulative frequency distribution of wind speed measured on the northeast of Lake Alexandrina..........................156

Figure 7.6: Periodicity in temperature stratification in the middle of Lake Alexandrina..........................................................157

Figure 7.7: Meteorological inputs and temperature outputs of ELCOM-CAEDYM hydrological model for Lake Alexandrina..........................158

Figure 7.8: Relationship between wind speed and the bed shear stress derived from modelled orbital velocity in the middle of Lake Alexandrina......................159

Figure 7.9: Non-linear regressions of wind speed and modelled bed shear at different water levels..............................................................160

Figure 7.10: Estimates of the critical wind speed required to resuspend sediment at different water levels..............................................................161
Figure 7.11: Frequency of sediment resuspension under different water levels...... 162

Figure 8.1: Area and maximum depth of lakes tested for sediment focussing (squares) and those shown in Table 8.2 (crosses).............................................. 177

Figure A.1.1: A comparison between two methods to calculate the proportion of TPM attributable to sediment resuspension in Lake Alexandrina ($r^2 = 0.91$).. 185

Figure A.2.1: Loss rate of SPM from the top section of the settling column under different salinities................................. 191

Figure A.2.2: Settling velocity for particles under different salinity regimes ....... 192
List of tables

Table 2.1: The relative contribution of water from the River Murray and the Darling River between 2001 and 2010 during the basin-wide drought........................... 19

Table 3.1: Total annual inputs, outputs and balance (± standard deviation) of water for Lake Alexandrina and Lake Albert, 2007-2009. ........................................ 40

Table 3.2: Non-linear regressions between water quality parameters and water level (WL) for Lake Alexandrina and Lake Albert..................................................... 41

Table 3.3: Total annual inputs, outputs and retention (± standard deviation) of nutrients and suspended particulate matter for Lake Alexandrina and Lake Albert........................................................................................................ 42

Table 4.1: Comparison of means (± Standard error) from Tukey's HSD post hoc analysis of sediment type and year for parameters from the two-way ANOVA results in Table 4.2 ................................................................. 69

Table 4.2: Statistical results from two-way analysis of variance examining the influence of sediment type (sand and mud) and sampling year on sediment characteristics...................................................................................................... 70

Table 4.3: One-way analysis of variance of sediment particle size between sandy sediments in 2007 (n = 11) and those in 2009 divided into dry sediment (n = 13), wet sediment (n = 15) and inundated sediments (n = 15) according to the influence of lake water on sediments at the time of sampling ......................... 71
Table 4.4: One-way analysis of variance of biogeochemical parameters between sandy sediments in 2007 (n = 11) and those in 2009 divided into dry sediment (n = 13), wet sediment (n = 15) and inundated sediments (n = 15) according to the influence of lake water on sediments at the time of sampling. .................... 72

Table 4.5: Linear regression results of LOI with volume concentration of sediment particle size categories and nutrients for mud in 2007 and 2009. ...................... 73

Table 5.1: Sedimentation rate for each trap site and each deployment period. ........ 94

Table 5.2: Repeated measures ANOVA with Bonferroni correction comparing parameters at trap sites grouped by sediment type (littoral sand sediments at South/East central and profundal mud sediments at Central/South central). ...... 95

Table 5.3: Sedimentation rates of organic carbon and nitrogen measured in Lake Alexandrina. ............................................................................................................. 96

Table 6.1: Critical shear stress and settling velocities for six particle size categories. ...................................................................................................................... 123

Table 6.2: Lake productivity estimates during well-mixed conditions in Lake Alexandrina. ............................................................................................................. 133

Table 7.1: Multiple regression output for a sinusoidal signal fitted to wind periodicity data. ......................................................................................................................... 151

Table 8.1: Seasonal variation in the proportion of algal species and functional groups for phytoplankton in Lake Alexandrina during 2009. ......................... 175

Table 8.2: Comparison of parameters relating to sediment resuspension in large, shallow lakes compared to Lake Alexandrina at different stages of the water level drawdown. ......................................................................................................................... 176
Table A.1.1: Estimates of the proportion of TPM attributed to sediment resuspension and autochthonous sedimentation using the three calculation methods........ 184

Table A.2.1: Composition of salt in Instant Ocean® and at three sites along the salinity gradient of Lake Alexandrina......................................................... 188

Table A.2.2: Values of coefficients and fit for equation A.2.2 under different salinities........................................................................................................ 189

Table A.2.3: Settling velocity and calculated floc radius under different salinity regimes. ........................................................................................................ 190
Preface

Ecosystem management in the twenty-first century is facing multiple, compounding crises that threaten to undermine the social fabric upon which Western civilisation has been founded (Foley et al. 2005; Annan et al. 2009; Rockström et al. 2009). These crises arise from a fundamental philosophical inconsistency with how we perceive ourselves, the environment and our place within it (see for example Aarons 1972; Capra 1982; Medlin 1992; Wetzel 1992; Norgaard 1995; Skinner 2007; Beddoe et al. 2009). It is not difficult to find examples of the pervasive extent to which human action has impacted ecosystems (the current geological era has even been called the anthropocene Vitousek et al. 1997; Steffen et al. 2007). Some notable figures of the anthropogenic impact on natural resources follow:

- Water impoundments retain over 10 800 cubic kilometers of water, reducing global sea levels by 3 cm (Chao 2008). Shifting this hydrological mass has had an observable influence on Earth’s polar drift and its rotational velocity (Chao 1995).
- Of the easily accessible freshwater runoff, 54% is impacted through consumption or pollution by agriculture, industry or urban use (Postel et al. 1996).
- An additional 4.5 – 14.5 Mt/year of phosphorus is estimated to be accumulating in agricultural lands and aquatic systems compared with pre-industrial times (Bennett et al. 2001).
- Globally, 96-98% of mammalian biomass is directly associated with human activity (i.e. humans, domesticated animals, pets etc. Smil 2002).
- Over 1 000 barrels of crude oil are consumed globally every second (International Energy Agency 2010).
The general problem was succinctly stated in the Millennium Ecosystem Assessment (MEA 2005), where it was written that

“human activity is putting such strain on the natural functions of the Earth that the ability of the planet’s natural ecosystems to sustain future generations can no longer be taken for granted” (p.5).

Moreover, what were previously perceived as sufficient management responses to mitigate these human impacts on ecosystems, have often included unintended consequences. For example, Likens et al. (2009) contend that “knowledge about aquatic ecosystems required to support water-resource management lags behind the increasing problems caused by past, often piecemeal, management approaches” (p.271). To ameliorate these shortcomings, the use of ‘best available science’ to inform policy is widely advocated in aquatic ecosystem management (ARMCANZ/ANZECC 1996; Sutherland et al. 2004; Moss 2008; Ryder et al. 2010). The contention is that applying the most coherent, consistent and detailed scientific knowledge presently available, while accepting that uncertainty is inescapable, will prevent piecemeal, and therefore problematic, management solutions. There remains an imperative to correctly define the scope of management, the scale of intervention and the levers available to achieve certain outcomes. If this is not done, the efforts of aquatic ecosystem managers can be poorly targeted (Pullin et al. 2009; Vörösmarty et al. 2010).

To meet the objective of contributing to the best available science, while preventing any conclusions from being misconstrued as solutions in their own right, this thesis starts (and ends) with a broad and inclusive definition of the problems it is addressing.
The chapters in this thesis have been prepared in a format suitable for later publication in scientific journals. This has inevitably led to some repetition between chapters.
1. Introduction

1.1 Stressors in aquatic ecosystems

Natural aquatic ecosystems are characterised by their efficiency in processing scarce materials, food web structures that ensure this efficient processing, connectivity with surrounding ecosystems and resilience to transient external disturbances (van Voris et al. 1980; Naeem et al. 1994; Naeem & Shibin 1997; Polis 1998; Hulot et al. 2000; Brookes et al. 2005; Raven et al. 2005; Moss 2008). Large perturbations to these characteristics can degrade the structure and function of ecosystems and so their ability to provide ecosystem services (Brock 1986; Costanza et al. 1997; Lake 2005). Anthropogenic perturbations of aquatic ecosystems present in many forms, including river regulation and water extraction, the introduction of invasive species, overharvesting of natural resources, increased nutrient inputs, riparian erosion and the accumulation of contaminants in ecosystems (Allan & Flecker 1993; Dudgeon et al. 2006; Kingsford 2011).

Increasing hydrological variability from climate change (Alcamo et al. 2007; Bates et al. 2008; Milly et al. 2008) will, in many cases, exacerbate these impacts on aquatic ecosystems (Vörösmarty et al. 2000; Carvalho & Kirika 2003; Jeppesen et al. 2005; Tibby & Tiller 2007; Paerl 2009; Williamson et al. 2009; Vörösmarty et al. 2010; Kingsford 2011). However, there is a body of work suggesting that climate
change impacts can be strongly mitigated by reducing the local stressors that result from anthropogenic disturbances and thereby restoring the capacity of ecosystems to adapt (Schindler 2001; Moss et al. 2008; Russell et al. 2009; Vescovi et al. 2009; Kingsford 2011).

Globally, flow regulation has fragmented most of the world’s river systems (Dynesius & Nilsson 1994; Nilsson et al. 2005) with over half of the available freshwater being appropriated for human use (Postel et al. 1996). Ecological stressors in heavily regulated rivers (Baxter 1977; Walker 1985; Kingsford 2000; Nilsson & Berggren 2000; Bunn & Arthington 2002) often derive from the altered timing and magnitude of flow (Rosenberg et al. 2000). This river regulation has increased water retention time (Vörösmarty et al. 1997), leading to the trapping of at least an extra 4-5 Gt yr$^{-1}$ of sediment that would have otherwise flowed into the ocean (Vörösmarty et al. 2003).

Retained sediments can alter the morphology, biogeochemical cycles, habitat availability and primary productivity of water bodies upstream and downstream of regulatory structures (Thoms & Walker 1993; Bourman & Barnett 1995; Lake et al. 2000; Vörösmarty et al. 2003; Matzinger et al. 2007). In shallow aquatic systems, sediments retained from regulation can also add to the available stock of sediment that can be resuspended, especially if regulation affects the size distribution and chemical characteristics of retained sediments (Thoms & Walker 1993; Bourman & Barnett 1995).

1.2 Sediment resuspension in shallow lakes

1.2.1 Sedimentation and resuspension

Sedimentation in lakes is the downward flux of autochthonous and allochthonous matter through the water column. Upon deposition at the sediment surface, this
matter can be resuspended into the water column when the shear stress (the friction force along the sediment surface) overcomes the gravitational and cohesive forces that hold sediment particles in place (Evans 1994; Bloesch 1995).

In shallow lakes, wind is the dominant source of energy to resuspend sediments (Bengtsson et al. 1990; Bailey & Hamilton 1997; Douglas & Rippey 2000; Andersen et al. 2007; Jin & Sun 2007; Chung et al. 2009). Wind shear on the surface of shallow waters generates orbital wave activity that is attenuated with depth. When this wave activity reaches the sediment surface it can act to resuspend sediment particles into the overlying water. The magnitude of this effect is influenced by the wind speed and direction; water depth; the size, density, cohesion and level of compaction of particles; and, the presence, abundance and type of macrophytes or benthic microalgae (Håkanson & Jansson 1983; Evans 1994; Bloesch 1995; Hamilton & Mitchell 1996; Horppila & Nurminen 2001; Madsen et al. 2001; Horppila & Nurminen 2003; James et al. 2004; Huang et al. 2007; Li et al. 2008; Spears et al. 2008).

Other factors can resuspend sediments in shallow lakes, although to a lesser extent than wind-induced wave activity. Water currents, from seiching or density driven flows can also impart a shear stress onto the sediment surface that can resuspend sediments if above the critical shear stress (Gloor et al. 1994; Bloesch 1995). The Common Carp (*Cyprinus carpio*) and other benthivorous fish are thought to resuspend sediments through their feeding action (Cline et al. 1994), the magnitude of which is determined by the fish stocking density and their feeding rate. When compared with other hydrodynamic factors, however, the effect of benthivorous fish is often not considered significant (Fletcher et al. 1985). Localised resuspension of sediments also occurs from the wake of boat motors, which can be a significant source of particle entrainment in lakes with heavy boat traffic (Yousef et al. 1980).
1.2.2 Impacts of sediment resuspension on aquatic ecosystems

Sediment resuspension increases the concentration of suspended sediments, which impacts on the physical aquatic environment. Suspended solids increase turbidity and decrease water clarity, with a corresponding decline in light availability (van de Hulst 1957; Davies-Colley & Smith 2001; van Duin et al. 2001). The reduction in water clarity affects the visibility range of sighted organisms (Vogel & Beauchamp 1999) that can reduce growth rates and alter feeding patterns in fish (Bruton 1985). Additionally, increasing suspended solids add to the pool of detritus within the water column that is a food source for zooplankton and other filter feeders (Hessen et al. 1990). Increasing turbidity also reduces the light availability for photosynthesis by phytoplankton, benthic algae and macrophytes as scattering and absorption of light increases with the concentration of suspended solids (Kirk 1983; Hellström 1991; Lind et al. 1994; Nagid et al. 2001; Lawson et al. 2007). Light availability plays a major role in phytoplankton succession (Bormans et al. 2005).

Denitrification in surface sediments may lead to a relative increase in phosphorus over nitrogen in the water column during resuspension events (Schelske et al. 1995; Hamilton & Mitchell 1997; Niemistö et al. 2008). However, the amount and direction of dissolved phosphorus sorption between particles and the water during resuspension events is controlled by numerous factors, including pH, CaCO$_3$ concentration, iron availability, organic carbon concentration and particle size distribution (Bostrom et al. 1988; Holmoos et al. 2009). Therefore, while many studies report resuspension-mediated desorption of dissolved phosphorus from suspended particles (Kristensen et al. 1992; Sondergaard et al. 1992; Reddy et al. 1996; Nagid et al. 2001; Horppila & Nurminen 2005), others report an increased absorption of soluble phosphorus during resuspension (de Groot 1981; Lennox 1984; Koski-Vähälä & Hartikainen 2000).
Areas of sediment resuspension tend to be inversely related to the area and density of macrophyte stands (Hamilton & Mitchell 1996; Hamilton & Mitchell 1997) as macrophytes need regions of low wave activity and turbulence to develop (Madsen et al. 2001; Schutten et al. 2004). Once present, macrophyte stands reduce the turbulent energy and resuspension in shallow lakes (Horppila & Nurminen 2001; James et al. 2004; Li et al. 2008) with consequent reductions in nutrient loading (Horppila & Nurminen 2003; Horppila & Nurminen 2005).

This control by macrophytes over sediment resuspension and nutrients is considered a feedback mechanism to maintain shallow lakes in one of two stable states: a clear-water state dominated by macrophytes or a turbid state dominated by phytoplankton and resuspended particles (Bachmann et al. 1999; Scheffer et al. 2001; Jackson 2003; Scheffer & Jeppesen 2007; Scheffer & van Nes 2007). There is some evidence that external nutrient reduction alone is sufficient to return macrophyte beds to shallow lakes, thereby increasing the water clarity through a reduction in phytoplankton and sediment resuspension (Jeppesen et al. 2003). However, other studies have shown that despite reductions in nutrient loading resuspension reduces light availability, preventing macrophyte recolonisation (Ibelings et al. 2007). Similarly, during droughts shallow lakes can be influenced by both declining nutrient inputs that should increase water clarity (van Geest et al. 2007) and increased sediment resuspension due to lower water levels that should decrease water clarity (Effler & Matthews 2004).

1.2.3 Water level decline and sediment resuspension

In catchments where climate trends towards warmer, drier conditions and water demand increases, water level decline in lakes is likely to become more common (Coops et al. 2003; Alcamo et al. 2007; Anon 2007; Bates et al. 2008; Tranvik et al. 2009). While the study of the influence of water level decline on shallow lake
ecosystems is increasing (Leira & Cantonati 2008), specific effects remain unresolved, perhaps due to the idiosyncratic responses of lakes.

Water level declines have divergent impacts in the littoral and pelagic zones. In the littoral zone, many macrophytes require seasonal water level drawdown for recruitment (van Geest et al. 2007), but prolonged low water level can lead to the loss of substrate for macrophyte growth or colonisation by benthic microalgae (Coops et al. 2003; Turner et al. 2005). While macrophytes can recolonise the newly formed littoral zone, the species composition is often altered due to changes in substrate type and chemical composition (Barko & Smart 1986; Barko et al. 1991; Blanch et al. 1999; Turner et al. 2005).

As water levels decline, the newly formed inundated littoral sediments are subject to increased scouring of loose, fine material (Håkanson et al. 2000; James et al. 2001; Effler & Matthews 2004; Punning et al. 2006). This generally decreases the organic material present because it has a relatively low density and is readily eroded (Furey et al. 2004). Chemical changes to exposed littoral sediments as water levels decline can also affect nutrient cycling. Partial drying of exposed sediments can increase their affinity for nitrogen and phosphorus, whereas complete desiccation reduces the affinity of sediment iron for phosphorus, exposes bacterial nitrogen and phosphorus to mineralisation and halts any anaerobic processes such as denitrification (Baldwin & Mitchell 2000). Consequently, if these sediments are rewetted, they can rapidly increase lake nutrient concentrations.

The effect of water level decline in regions of shallow lakes that remain pelagic for the duration of the drawdown are less pronounced and have received less attention (Leira & Cantonati 2008). Evaporation can increase the concentration of nutrients as water levels decline (Nõges & Nõges 1999). Sediment resuspension often becomes more prevalent as the area and type of sediment exposed to turbulent energy changes (Håkanson et al. 2000; James et al. 2001; Effler & Matthews 2004; Punning et al. 2006).
This corresponds to an increased load of suspended particles and turbidity (Nõges et al. 1999; Håkanson et al. 2000; Cózar et al. 2005). Declining light availability from increased sediment resuspension may reduce primary productivity (Hellström 1991; Lawson et al. 2007) and phytoplankton biomass (Lind et al. 1994). Conversely, higher sediment resuspension increases the total phosphorus load (Shantz et al. 2004), which can increase the phytoplankton biomass (Nagid et al. 2001), and may return meroplankton into the water column, increasing primary productivity (Schelske et al. 1995; Schallenberg & Burns 2004).

The contrasting impacts of sediment resuspension upon nutrient cycling and primary productivity makes it difficult to predict how increased sediment resuspension resulting from water level decline will affect water quality and primary productivity of individual shallow lakes. However, given the apparent importance of sediment resuspension in shifting shallow lakes between alternative stable states, it is vital to develop an understanding of wind-driven resuspension and how this changes with depth, to manage shallow lakes in a changing climate. This thesis aims to firstly elucidate the effects of water level decline on sediment resuspension and redistribution, accompanying nutrient changes and the underwater light climate. Secondly, it seeks to understand the interactions between water level decline and sediment resuspension in the context of alternative stable states during periods of drought to determine the implications for restoration of large shallow lakes. These objectives were achieved by studying the changes to sediment distribution and water quality during a period of extreme water level drawdown in a shallow, South Australian lake. Drawdown lasted 3 years, between 2007 and 2010. Additional field experimentation and a hydrodynamic model were used to gain an understanding of the mechanisms driving a shift in the physical attributes of the lake towards either of the alternate stable states.
1.3 Thesis outline

The thesis comprises eight chapters. Following the Introduction (Chapter 1), the Site Description (Chapter 2) provides a detailed overview of the Lower Lakes and their catchment, the Murray-Darling Basin. This chapter also outlines the extensive management interventions that occurred during the period of study to try to minimise adverse social and ecological impacts of the unprecedented lake drawdown event.

The main body of the thesis is made up of five experimental chapters. The timing of these experiments relative to the water level drawdown in Lake Alexandrina are depicted in Figure 1.1. Chapter 3 is a water budget and mass balance that demonstrates the importance of internal nutrient loading in the studied lakes. Chapter 4 compares the sediment composition and distribution before (2007) and after (2009) water level decline. The dynamics of sediment distribution are examined with sediment traps in Chapter 5 at low water levels. Chapter 6 explores the links between wind speed, sediment resuspension and changes to light attenuation using an intensive three day field experiment where data were recorded in high-resolution. These data are used to infer changes to productivity during low water levels. The final experimental chapter, Chapter 7, utilises a three-dimensional hydrodynamic model to relate wind speed, bed shear stress and water level with the particle size of sediments undergoing resuspension. This analysis elucidates the frequency of resuspension events under different water levels given the measured wind conditions at the study site. A final discussion (Chapter 8) ties together the central findings of the thesis with the main aims of relating sediment resuspension and water level with changes to stable ecosystem states.
Figure 1.1: Timing of experiments relative to water level drawdown in Lake Alexandrina. Data prior to August 2008 was taken from Aldridge et al. (2009). Numbers refer to the chapter number where the experiment is described.
2. The Murray-Darling Basin and Lower Lakes

2.1 The Murray-Darling Basin

2.1.1 Catchment characteristics

The Murray-Darling Basin (MDB) is Australia’s largest catchment, draining over 14% of the total landmass of the continent (Figure 2.1). The River Murray, the longest in the basin, rises in eastern Australia on the inland side of the Great Dividing Range at the Australian Alps. From here, it flows west across inland plains of low topography along the state border between New South Wales and Victoria before entering South Australia and turning south to drain into the Southern Ocean. The Darling River is the second longest in the basin and rises in the sub-tropical region of southeastern Queensland before flowing southwest to join the River Murray, about 520 km upstream of the Murray Mouth. The catchment has over 20 other major rivers that drain land from diverse climates ranging from sub-tropical to arid, with the vast majority of the catchment classified as semi-arid. This variety of climates in different sub-catchments gives the MDB some of the most variable annual river flows in the world (Finlayson & McMahon 1988).
Since European settlement, river flow in the MDB has become increasingly regulated, with numerous in-channel locks, weirs, dams, barrages and regulators initially built to control water availability for navigation, and later to maintain stable water levels to allow extractions for irrigation. The MDB is considered the ‘food bowl’ of the country, generating about 40% of national agricultural GDP (Quiggin 2001). Irrigation can substantially enhance the value of agricultural output compared with other farming practices so it is used widely throughout the basin. This extensive development and water extraction has reduced the mean annual flow out of the MDB by 61% compared with pre-regulated conditions (CSIRO 2008). Flow extractions and regulation has reduced or eliminated the lateral flood pulses that characterised the River Murray, degrading its floodplain biota and altering the hydrological regime of ecologically diverse wetlands along its reaches (Walker 1985; Kingsford 2000; Lake 2005). Regulated flows and a reduced connectivity to the floodplain have also caused sediments and associated nutrients to accumulate in the main channel of the River Murray (Walker 1985). Inefficient irrigation can exacerbate river salt loads as groundwater pressure differentials force dissolved salts into the river after raising local water tables, irreversibly degrading some irrigated land (Jolly et al. 2001; Knight et al. 2005). Dissolved salts increase in concentration along the river reaches, with chloride concentrations rising from 2 mg L\(^{-1}\) in the upper river reaches to 170 mg L\(^{-1}\) in the river channel in South Australia (Herczeg et al. 1993).

### 2.1.2 Drought in the Murray-Darling Basin

Between 2001 and 2010, the Murray-Darling Basin experienced widespread drought that reduced surface water flows, soil moisture content and groundwater recharge throughout the catchment (Leblanc et al. 2009). While this drought did not correspond to the lowest average rainfall over a 10 year period on record, it did result in inflows lower than any period in recorded history (Murphy & Timbal 2008) with a 1 in 1500 year likelihood of occurrence (Gallant & Gergis 2011). A non-linear
relationship between rainfall and runoff has been modeled for the basin, showing that small decreases in rainfall can magnify decreases in runoff as soil moisture is depleted through increased evaporation, especially if the rainfall deficit is in Autumn (Chiew et al. 1995; Chiew et al. 2009). In this drought there was a significant decline in autumn rainfall, whereas previous droughts were substantially drier during winter months (Murphy & Timbal 2008; Timbal 2009). Lower autumn rainfall reduces catchment wetting prior to winter rain, resulting in significantly lower runoff and river inflows.

A number of additional factors combined to exacerbate the severity of drought on downstream ecosystems including, but not limited to: increasing temperatures thought to result from ongoing climate change (Cai & Cowan 2008); decreasing snow-depths in the Australian Alps (Nicholls 2005); over-allocation of water resources such that environmental flows could not be met during dry years (Quiggin 2001); and, misdirected and delayed government expenditure on remediating this over-allocation (Lee & Ancev 2009).

2.2 The Lower Lakes – Lake Alexandrina and Lake Albert

2.2.1 Site description of the Lower Lakes

The effects of basin-wide regulation and drought became magnified in the downstream reaches of the river system. At the end of the River Murray are two large shallow lakes collectively known as the Lower Lakes (Figure 2.1). The river flows into the northeastern corner of Lake Alexandrina, which is connected on its southeastern side to Lake Albert. A channel on the southwestern side of Lake Alexandrina stretches towards Goolwa (hereafter, the Goolwa Channel), and is dominated by small islands and wetlands of high biodiversity. At five locations (Figure 2.1) barrages were constructed between 1935 and 1940 to stop saline water
intrusions from the downstream coastal lagoon and to maintain average lake water levels of 0.75 m AHD (Australian Height Datum, where zero is the mean sea level recorded between 1966 and 1968). The barrages separate Lake Alexandrina from the northern end of the coastal lagoon, called the Coorong, that stretches from Goolwa past the Murray Mouth and runs along the coast to the southeast. At the Murray Mouth, the River Murray drains into the Southern Ocean.

Along with the Coorong, the Lower Lakes are recognised as internationally significant sites for migratory birds and require protection under the Ramsar agreement (Phillips et al. 2005). Additionally, this area is listed as one of six 'Icon Sites' along the River Murray under the Living Murray Program (Murray Darling Basin Commission 2007). The recognition of the ecological importance of the area results from a high diversity of habitats and ecosystems that emerge from the salinity gradient: from freshwater conditions in the Lower Lakes to the hypersaline conditions in the southern lagoon of the Coorong (Aldridge et al. 2009).

2.2.2 Water Quality

The Lower Lakes are polymictic systems characterised by a shallow morphology, high levels of sediment resuspension and limited light availability (Geddes 1984; Geddes 1988). When full, Lake Alexandrina has a maximum depth ($Z_{\text{max}}$) of 4.1 m, a mean depth ($Z_{\text{mean}}$) of 2.9 m and a surface area of 580.6 km$^2$ (Geddes 1984). Lake Albert is shallower ($Z_{\text{max}} = 1.7$ m) and the cumulative volume of both lakes is 2015 GL (Phillips et al. 2005). Groundwater seepage, rainfall onto the lake surface and overland run-off influence both lakes, but inflows to Lake Alexandrina come predominantly from the River Murray except during dry years when local tributaries are important in maintaining winter water levels (Phillips et al. 2005). Inflows to Lake Albert come almost solely from Lake Alexandrina via the Narrung Narrows. Water level fluctuations in both lakes, as well as the direction of flow between them, are significantly influenced by the strength and direction of prevailing winds.
Turbidity of the Murray River is strongly influenced by the contribution of flows from the highly turbid Darling River (Geddes 1988), and this in turn influences the turbidity of the lakes. During the drought period the contribution of flows from the Darling River declined to very low levels (Table 2.1).

Prior to the drought in the Murray-Darling Basin, seasonal evapo-concentration and reduced River Murray flow occasionally increased electrical conductivity from 0.2 to 0.6 g.L\(^{-1}\) and 0.6 to 1.5 g.L\(^{-1}\) in Lake Alexandrina and Lake Albert, respectively (Lamontagne et al. 2004; Sim & Muller 2004). Modeling of pre-regulated conditions suggests that during extensive droughts, electrical conductivity could have increased to 7 g.L\(^{-1}\), but these events were demonstrably transient (Close 1990). Paleolimnological evidence suggests that both lakes have remained relatively fresh for at least 7000 years (Barnett 1994; Fluin et al. 2007).

Both lakes are classified as eutrophic to hypereutrophic based on total nutrient concentrations, but soluble forms of phosphorus, oxidised nitrogen and ammonium are often below detection levels (Geddes 1984). A substantial increase in nutrient input occurred from agricultural development of the catchment in the mid-twentieth century (Herczeg et al. 2001; Fluin et al. 2007). These nutrients are assimilated into aquatic plants and phytoplankton before being exported from the lake in their organic form (Herczeg et al. 2001; Cook et al. 2009).

Emergent littoral macrophytes, such as *Eleocharis* and *Baumea* spp., were once prevalent on lake-fringes but have been restricted to tributary confluences or been replaced with hardier species such as *Phragmites australis* and *Typha domingensis* as a result of static water levels (Sim & Muller 2004; Phillips et al. 2005). Submerged macrophytes that once spread for kilometres into the lakes are now limited to sheltered areas near the shore with high light availability, such as irrigation channels or at the confluence of tributaries (Phillips et al. 2005).
While the historical evidence is scant, it is thought that these lakes have undergone a regime shift in state from clear to turbid waters (see Scheffer et al. 2001). Herczeg et al. (2001) suggests that a C:N mass ratio of less than 11 in three short sediment cores, which represented 120 years of sedimentation and were collected in Lake Alexandrina in 1995, indicates that aquatic plants have been the dominant form of organic matter undergoing sediment burial. Further anecdotal evidence comes from historical records of residents around the lakes. Numerous reports of reeds, bulrushes and waterweed depict a water body full of macrophytes. For example, in 1933, during preparations for the construction of the barrages, T Charles Good wrote that ‘...fresh waterweed grew for a mile out into the lake...’ in Lake Albert at Meningie and that at Narrung, ‘...reeds eight feet high grew, so that a sailing boat could shelter in them...’ (Sim & Muller 2004, p.64). George Ray Goode wrote in a submission about the barrage construction of Lake Albert that ‘...the shores were protected by freshwater reeds and lignums. In lake freshwater weeds prevented break of waves.’ (Sim & Muller 2004, p.63). It is unclear exactly when this shift from clear to turbid water occurred, however, the introduction of the barrages and static water levels in the 1940s, and the introduction of the Common Carp (Cyprinus carpio) in the 1970s could both have driven the decline (Paton 2010). Surveys between 2004 and 2005 that were repeated between 2008 and 2009 showed further declines in both the abundance and species diversity of aquatic macrophytes in both lakes (Marsland & Nicol 2009).

2.2.3 Impacts of regulation

Sediments in the Lower Lakes have been significantly affected by regulation in the lakes and further upstream in the River Murray. The barrages between Lake Alexandrina and the Coorong led to static water levels that increased shoreline erosion around both lakes, increasing their non-biogenic sediment load (Butler & Woodard 1993; Murdoch 2009). Further upstream, alterations to the hydrological
regime of the River Murray arose from locks and weirs along its length. Peak flow in the River Murray channel now occurs in late summer to meet irrigation demand, where once it occurred in late winter and early spring as snowmelt enhanced run-off from the catchment (Walker 1985). Altering the seasonality and decreasing the variability of flow through regulation of the River Murray has reduced the frequency and magnitude of peak flow events. This has resulted in the retention of larger sediment particles behind locks and weirs in the river channel (Thoms & Walker 1993). Consequently, the type of sediment entering Lake Alexandrina shifted towards finer sediments (Bourman & Barnett 1995).

A decreased propensity to flood has also reduced the volume of water available to flush both sediment and salt out of the lakes and through the Murray Mouth. The long-term sediment accumulation rate has increased as a result (Barnett 1994; Herczeg et al. 2001), which has reduced the maximum water depth in the lakes, especially in the Goolwa Channel (Bourman & Barnett 1995). As a result of extensive hydrological regulation, the type of sediment being deposited has changed from bioclastic sand to fine mud (Barnett 1994; Bourman & Barnett 1995; Herczeg et al. 2001).

Stable water levels following the construction of the barrages, have allowed iron sulfites to accumulate in sediments (Hall et al. 2006; Lamontagne et al. 2006; McCarthy et al. 2006). When exposed to the atmosphere during water level declines, iron sulfites can oxidise, forming sulfuric acid and releasing heavy metals into porewater (Fitzpatrick 2003; Simpson et al. 2010).

### 2.2.4 Recent changes

Changes to the Lower Lakes during drought were dominated by a rapid and unprecedented drawdown of water level between 2007 and 2010 (Figure 2.2). The rate of water level decline was greater than 0.5 m per year in 2008 and 2009. Water
levels in Lake Alexandrina reached a minimum of -1.1 m AHD in April 2009 as 70% of the lake’s volume evaporated. At this water depth, $Z_{\text{mean}}$ had been reduced from 2.9 m to approximately 1 m (Figure 2.3). Declining water levels resulted in increases in electrical conductivity as evaporative concentration of salt and seawater ingress through the barrages increased concentrations in the middle of Lake Alexandrina from ~0.5 g.L$^{-1}$ to >3.9 g.L$^{-1}$.

A number of interventions occurred in the Lower Lakes during the period of drawdown as managers attempted to minimise the risk and threat of acid sulfate soils oxidising (Figure 2.2). To maintain water levels in Lake Albert and prevent the exposure of sediments rich in iron-sulfides, an earthen embankment was constructed in the Narrung Narrows to disconnect Lake Alexandrina from Lake Albert and allow water to be pumped into Lake Albert. This embankment was completed in April 2008 and pumping was maintained to keep water levels in Lake Albert at a nominal depth of -0.5 m AHD.

During the Austral summer of 2008 and 2009, Currency Creek dried out completely and acid sulfate soils reduced the pH of porewater below 2 at these sites (Dominic Skinner, Unpublished Data). During June and July of 2009, aerial dosing of finely ground particulate limestone to increase local alkalinity was applied at the confluence of Lake Alexandrina and Currency Creek (Figure 2.1) to counteract acidity. Following the application of limestone, two earthen embankments were built to create pooled water over the acid sulfate soils, thus preventing further exposure to air. The first embankment was completed in August 2009 Clayton, forming a pool of water between Clayton and the barrage at Goolwa (Figure 2.1). There was 27.5 GL of water pumped from Lake Alexandrina into this pool to raise water levels to 0.7 m AHD. The second embankment was built at the end of Currency Creek to prevent the first flush of water over the acidic soils from immediately entering the lake. This embankment was completed in September 2009.
In spring 2009, large scale aerial seeding was conducted on many exposed banks of both Lake Alexandrina and Lake Albert to stabilise sediments of the exposed lakebed. Crops of cereal and grasses were planted in this manner and became widely established on the exposed shoreline until water levels rose from basin-wide flooding in winter 2010.
Table 2.1: The relative contribution of water from the River Murray and the Darling River between 2001 and 2010 during the basin-wide drought.

<table>
<thead>
<tr>
<th>Year</th>
<th>River Murray (GL)</th>
<th>Darling River (GL)</th>
<th>Contribution from the Darling (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2001</td>
<td>206.3(^2)</td>
<td>1902.7</td>
<td>90.2</td>
</tr>
<tr>
<td>2002</td>
<td>615.2</td>
<td>253.9</td>
<td>29.2</td>
</tr>
<tr>
<td>2003</td>
<td>96.2</td>
<td>17.5</td>
<td>15.4</td>
</tr>
<tr>
<td>2004</td>
<td>1019.7</td>
<td>86.9</td>
<td>7.9</td>
</tr>
<tr>
<td>2005</td>
<td>1699.2</td>
<td>35.2</td>
<td>2.0</td>
</tr>
<tr>
<td>2006</td>
<td>817.9</td>
<td>15.6</td>
<td>1.9</td>
</tr>
<tr>
<td>2007</td>
<td>412.7</td>
<td>9.1</td>
<td>2.2</td>
</tr>
<tr>
<td>2008</td>
<td>673.6</td>
<td>359.2</td>
<td>34.8</td>
</tr>
<tr>
<td>2009</td>
<td>1342.3</td>
<td>51.8</td>
<td>3.7</td>
</tr>
<tr>
<td>2010(^3)</td>
<td>463.6</td>
<td>474.1(^4)</td>
<td>50.6</td>
</tr>
</tbody>
</table>

\(^1\) Flow in the Darling River was measured at Burtundy, upstream of the confluence with the River Murray. Between 2001-2007, the contribution from the River Murray was estimated as the difference between flow measured at Lock 9, downstream from the confluence with the Darling River, and that measured at Burtundy. From 2007 onwards, River Murray flow was measured downstream of Mildura, just upstream of the confluence with the Darling River.

\(^2\) Data provided by the Murray-Darling Basin Authority.

\(^3\) Flow data in 2010 is limited to the period from 1\(^{st}\) January to 24\(^{th}\) March.

\(^4\) Floods in the Darling River occurred between February and March 2010, but the water from these floods did not start to enter Lake Alexandrina until early April 2010.
Figure 2.1: The location of Lake Alexandrina (35.4278° S 139.16142° E) and Lake Albert within the Murray-Darling Basin. Lake Alexandrina is the only drainage point for the River Murray, which enters to the northeast of the lake.
Figure 2.2: Timeline of significant management events during the drought-induced decline in water level of Lake Alexandrina (inset).

**2002 – 2010**: Dredging of the Murray Mouth to maintain connectivity between Coorong Lagoon and Southern Ocean

**April**: Construction of the temporary regulator at Narrung, with pumping from Lake Alexandrina into Lake Albert beginning shortly thereafter.

**January**: Drinking water pipeline from Tailem Bend to major settlements on the Lower Lakes completed.

**June**: Pumping between Lake Alexandrina and Lake Albert stopped over winter.

**June-July**: Aerial dosing of limestone onto Currency Creek to neutralize acid sulfate soils.

**August**: Construction of the temporary regulator at Clayton completed.

**September**: Construction of the temporary regulator at Currency Creek completed.

**September**: Bioremediation of acid sulfate soils, which involved aerial seeding of cereal and grass crops onto exposed lakebeds was undertaken.

**October**: Irrigation pipeline between Jervois (River Murray) and the wine regions of Currency Creek and Langhorne Creek completed.

**January**: Drinking water pipeline extended to Point Sturt & Hindmarsh Island.

**February**: Carp removal program implemented in Lake Albert to minimise the risk of adverse public health outcomes resulting from the predicted salinity induced mass fish kills.

**April**: Water from floods in the Darling River reaches Lake Alexandrina and begins to refill the lake.

**August**: Water from basin wide floods reaches Lower Lakes, raising water levels to long-term average and flowing across the barrages into the Coorong.
Figure 2.3: Bathymetry of Lake Alexandrina, Lake Albert and the northern end of the Coorong. Depths shown are in metres Australian Height Datum. Data provided by the South Australian Department of Environment and Heritage.
3. Water quality changes during water level drawdown in two regulated shallow lakes

3.1 Introduction

There is now considerable evidence that shallow lake ecosystems can shift between two alternative stable states: one a clear water state, dominated by macrophytes and the other a turbid water state with high phytoplankton density (Scheffer et al. 2001; Jackson 2003; Scheffer & Jeppesen 2007; Scheffer & van Nes 2007). The reasons that these regime shifts occur, and how these shifts can be reversed if they do occur, are complex. Most studies have emphasised the role of eutrophication in promoting phytoplankton growth as the initial cause of the switch from a clear to turbid state (Lowe et al. 2001; Jackson 2003; Håkanson et al. 2007). However, changing salinity (Davis et al. 2003; Jeppesen et al. 2007) and water level fluctuations (Wallsten & Forsgren 1989; Håkanson et al. 2000; Coops et al. 2003) have also been implicated.

Drought-induced water level declines have been shown to shift some shallow European lakes towards a transient clear water state as external nutrient inputs decline, with the effect most prolonged in smaller lakes (van Geest et al. 2007). Conversely, other studies have shown that drawdown can enhance the influence of nutrient release from sediments, which can increase phytoplankton abundance and
turbidity (Baldwin et al. 2008; Özen et al. 2010). Lake size may play a role in determining the direction of change resulting from water level decline as the external nutrient load required to shift shallow lakes with a large surface area from clear to turbid waters is lower than that for smaller shallow lakes (Scheffer & van Nes 2007). Large, shallow lakes are more prone to increases in turbidity for a number of reasons. Larger lakes are more likely to be populated with fish, increasing zooplankton predation and potentially removing grazing pressure on phytoplankton (Scheffer & van Nes 2007). This would increase biogenic turbidity. In addition, greater lake size increases the wind fetch, and consequently the wind energy available to resuspend sediments (Nagid et al. 2001). Higher wind energy imparts more turbulent energy at the sediment surface, which can disturb or prevent the establishment of macrophytes (Madsen et al. 2001; Schutten et al. 2004). This, in turn, can increase shoreline erosion, reduce the stability of sediment, and increase the amount of sediment resuspension (Horppila & Nurminen 2001; Madsen et al. 2001; James et al. 2004; Huang et al. 2007; Li et al. 2008). Consequently, lake area can influence the interactions between wind energy and macrophyte colonization, with macrophytes in smaller lakes more likely to withstand interannual storm events. Water level declines can exacerbate the resuspension of sediments by reducing the depth through which wind energy is attenuated before it reaches the sediment surface (Effler & Matthews 2004). This sediment resuspension can increase a lake’s nutrient load (Kristensen et al. 1992; Sondergaard et al. 2003), even if previous external nutrient loads have been reduced (Lijklema 1994).

In lake catchments where climate trends are towards a warmer and drier state (Alcamo et al. 2007), periods of water level decline are likely to become more prevalent. Consequently, understanding the interactions between drought-induced declines in both water level and external nutrient inputs is essential to managing shallow lakes in an uncertain climatic future. In addition, periods of lake drawdown provide insights into the mechanisms that determine the directional response of a
shallow lake towards either a clear or turbid state. To achieve this, the current study utilises a monthly mass balance during a period of extreme water level drawdown in the two interconnected large, shallow lakes (Lake Alexandrina and Lake Albert) in semi-arid South Australia. Both lakes have suffered degradation resulting from extensive hydrological regulation and changing land-use, with evidence of a shift to a more turbid state corresponding with this regulation (see section 2.2). The mass balance is calculated for phosphorus (both total and soluble) to show changes to nutrient loading, as well as suspended particles to distinguish between external and internal nutrient loading.

3.2 Methods

3.2.1 Site description

Lake Alexandrina (580 km²; mean depth 2.7 m) and Lake Albert (239.5 km²; mean depth 1.4 m) are polymictic, eutrophic shallow water bodies described in detail in section 2.2 (Figure 3.1). As a result of a basin-wide drought, water levels in Lake Alexandrina declined between 2007 and 2010 from 0.4 m AHD to a minimum of -1.1 m AHD (Figure 3.2).

Water samples were collected at 24 sites in Lake Alexandrina, Lake Albert, the channel between Lake Alexandrina and Goolwa (hereafter, the Goolwa Channel) and two tributaries, Currency Creek and Finniss River (Figure 3.1). Samples were collected every four weeks between August 2008 and February 2010. Data collected every eight weeks between January 2007 and April 2008 from a previous study (Aldridge et al. 2009) were appended to provide a three year dataset spanning the full range of water levels during the drawdown. Fifteen sites were shared between the two monitoring programs (Figure 3.1). As water levels dropped, some sites became inaccessible due to shallow water. Currency Creek dried out completely in February 2009 and was not sampled again until rainfall in May 2009 formed isolated
3.2.2 Monitoring water quality

Depth integrated water samples of the entire water column were taken with a PVC pipe (internal diameter 4 cm) to fill a bucket (~10 L) from which all further water samples were taken for analyses. Subsamples of water were filtered on-site immediately after collection through a pre-washed 0.45 µm membrane for analysis of filterable reactive phosphorus (FRP) and dissolved organic carbon (DOC). Unfiltered water samples were stored in the dark at 4°C for later determination of suspended particulate matter (SPM), chlorophyll a (chl a), total phosphorus (TP) and total nitrogen (TN). These analyses normally occurred within 3 days of collection.

A known volume of water from each site was filtered through a pre-washed Whatman GF/C filter, dried at 55°C for 2 hours and reweighed to give SPM gravimetrically. A second subsample of water was filtered in the same manner, then contents collected on the filter were extracted over 24 hours with methanol (99.8%) to determine chl a spectrophotometrically at 665 nm and 750 nm (Golterman et al. 1978). Water samples for TP, TN, DOC and FRP analysis were sent to external analytical laboratories. Until April 2008 samples went to the CSIRO Analytical Laboratory (Waite, South Australia), thereafter, samples were sent to the Australian Water Quality Centre (Bolivar, South Australia), a National Association of Testing Authorities (NATA) endorsed laboratory. Methods for water analysis were consistent between laboratories. TP was determined by acid digestion followed by inductively coupled plasma optical emission spectrophotometry (ICP-OES) according to Lamontagne et al. (2004). TN was calculated as the sum of oxidised nitrogen, total organic nitrogen and ammonia all of which were measured with automated flow colorimetry (Eaton et al. 2005). FRP was determined colorimetrically using segmented flow analysis with ammonium molybdate, potassium antimony tartrate
and ascorbic acid according to standard methods (Eaton et al. 2005). DOC was measured by infrared detection after the removal of inorganic carbon (Eaton et al. 2005).

Triplicate scalar irradiance ($E_0$) readings were recorded through the water column at 0.25 m intervals using a LI-COR underwater spherical quantum sensor with a LI-1400 data logger (LI-COR, Nebraska). Extinction coefficients ($k_d$) were calculated by regressing log-transformed irradiance measurements against depth, according to equation 3.1

$$\ln E_0(z) = \ln E_0(0) - k_d z$$  \hspace{1cm} (3.1)

where $E_0(z)$ and $E_0(0)$ are the scalar irradiance readings at water depth $z$ and immediately below the water surface, respectively (Kirk 1983). Depth profiles of electrical conductivity and temperature were also measured using a pre-calibrated multi-probe sonde (TPS-90FLT, TPS, Queensland). Electrical conductivity was converted to salinity at 25°C using standard methods (Eaton et al. 2005).

All statistical analyses were conducted with GraphPad Prism (Graphpad Software Inc.). Nutrient concentrations for both lakes were taken as the monthly average of all sites within each lake. For these calculations, the area of Lake Alexandrina was restricted to the main basin only and did not include the water in the Goolwa Channel (Figure 3.1). Non-linear regressions were used to test for relationships between changing lake nutrient concentration and water level, which are expected to be exponential due to the evaporative concentration of nutrients. Where exponential fits were found for a parameter in both lakes, data were log-transformed to give a linear relationship that allowed comparison of the rate of change in nutrient concentration between the two lakes using an analysis of covariance (ANCOVA). For all tests, a $p$-value less than 0.05 was deemed significant.
3.2.3 Water balance

The monthly change in water volume ($B_{\text{calculated}}$) was calculated independently for both Lake Alexandrina and Lake Albert according to the equations 3.2 – 3.3 from Özen et al. (2010):

$$B_{\text{calculated}} = RM + T + R \pm P - E$$  \hspace{1cm} (3.2)

Where RM is the inflow from the River Murray, T is the inflow from tributaries (Currency Creek and Finniss River), R is rainfall onto the lake surface, P is pumping between lakes and E is the evaporation. Daily inflow into Lake Alexandrina from the River Murray (RM) was estimated using the BIGMOD hydrological model (Murray-Darling Basin Authority) and summed to give monthly inflow volumes. BIGMOD output was only available until the end of June 2009, so the average inflow of each month during the preceding two years was used to estimate monthly inflows between July 2009 and February 2010. This was justified on the basis that flow was relatively stable until April 2010 (Murray-Darling Basin Authority, A.Ahmad, personal communication). Prior to construction of an embankment to separate Lake Alexandrina and Lake Albert, in April 2008, River Murray inflows were distributed between the two lakes according to their surface area. After construction, all flow from the River Murray was attributed to Lake Alexandrina. Daily pumping volumes (P) were summed to give monthly input to Lake Albert or output from Lake Alexandrina for use in the water balance. Daily flow from local tributaries (T, Currency Creek and Finniss River) was metered by the Department for Water, Land and Biodiversity Conservation South Australia (DWLBC) and these values were summed to give monthly input.

Daily rainfall data (R) was provided by the Australian Bureau of Meteorology (BoM) for three locations, Murray Bridge (used for Lake Alexandrina), Goolwa (used for the Goolwa Channel) and Meningie (used for Lake Albert). The volume of rainfall
input was calculated as the product of the surface area of the lake and the depth of rainfall. Daily volumes were summed to give monthly inputs. Similarly, evaporated volume (E) was calculated as the product of lake evaporation depth and surface area.

For this calculation, reference evapotranspiration (ET$_o$) rates were calculated using the Penman-Monteith equations given by Allen et al. (1998) from mean daily meteorological data collected at Narrung by the South Australian Natural Resources Management Board. Values of E were taken as 1.23 ET$_o$. This was derived after converting ET$_o$ to pan evaporation (PE) using the pan coefficient of 0.65 (Allen et al. 1998) and correcting pan evaporation to local conditions (PE = E/0.8) according to Webster (2005).

Daily water level (provided by DWLBC) and lake bathymetry data with a 10 metre spatial resolution (provided by the South Australian Department of Environment and Heritage) were used to calculate lake volume ($V_{measured}$). $B_{calculated}$ was subtracted from the measured lake volume ($V_{measured}$) to give an estimate of groundwater movement (G) according to equation 3.3:

$$V_{measured} = B_{calculated} \pm G$$

(3.3)

Outflow was not included in the water balance because water levels were never high enough to flow over the barrages that separate Lake Alexandrina from the Coorong and Southern Ocean. Irrigation extraction was also considered to be insignificant because during the water level declines presented here, drought and increasing salinities meant extractions ceased (Kingsford et al. 2011).

3.2.4 Monthly nutrient and suspended particulate matter balance

For the purposes of calculating nutrient (TP and FRP) and SPM mass balances, the lakes in this study were broken into three discrete regions according to water quality. Lake Alexandrina was divided to separate the main body of the lake from the higher
salinities in the Goolwa Channel (see Figure 3.1). Lake Albert was considered to be the third region in the mass balance. The monthly mass of TP, FRP and SPM in each region was determined as the average monthly concentration multiplied by the volume. For Lake Alexandrina, this value was summed with that of the Goolwa Channel to give a total monthly mass of nutrients for the lake. Changes to mass between months were calculated by difference. Annual retention was estimated as the difference between annual inputs and annual outputs. Standard deviations of annual water and nutrient retention were obtained by averaging the 12 months of data for each year.

The movement of nutrients was estimated by multiplying the nutrient concentration with the volume of water transferred. Nutrient inputs from the River Murray were based on nutrient concentrations measured in the river. Nutrient inputs from tributaries were derived from nutrient concentrations in samples collected in each tributary (or in the confluence of the tributary and Lake Alexandrina for the period between January 2007 and April 2008). The nutrient concentrations of water pumped into Lake Albert were derived from measurements at Narrung, while the nutrient concentrations of water pumped into the weir pool between Clayton and Goolwa were determined from measurements at Clayton. Groundwater input or output were assumed to have the same soluble nutrient concentration as the average of all surface water samples within that region so that SPM and particulate bound phosphorus were not lost via these processes. The concentration of particles in rainfall was assumed to be negligible and was not included in calculations. Phosphorus concentrations in rainfall were estimated from FRP measurements taken at Adelaide airport by Wilkinson et al. (2006).

When concentrations of nutrients were below the detection limit (5 µg L\(^{-1}\) for FRP), the concentration used for the nutrient balance was assumed to be half of the detection limit. For the few months when nutrient data at one site was not collected,
the average of the immediately preceding and subsequent month’s concentration was taken.

3.3 Results

3.3.1 Water balance

The water balance revealed that inflows from the River Murray, followed by rainfall were the dominant water inputs to Lake Alexandrina, with a total of 1589 GL and 586 GL being added over the three year study, respectively (Figure 3.3). For Lake Albert, River Murray flows were the main water source until the construction of the earthen embankment at Narrung in May 2008. After this, water pumped from Lake Alexandrina became the major water source (Figure 3.4). Over the 14 months of pumping, 171 GL was transferred from Lake Alexandrina to Lake Albert. Water volume decline was driven by evaporative losses, which in both lakes was greater than the total inputs. Evaporation from Lake Alexandrina totaled 2665 GL between 2007 and 2009, while for Lake Albert, evaporative losses were 653 GL.

The monthly change in water volume ($B_{\text{calculated}}$) showed a strong correlation with the measured volume ($V_{\text{measured}}$) of both Lake Alexandrina ($r^2 = 0.92$; slope = 1.14 ± 0.06) and Lake Albert ($r^2 = 0.92$; slope = 1.05 ± 0.05) before groundwater calculations were included. Monthly $B_{\text{calculated}}$ for both lakes were consistently higher than $V_{\text{measured}}$ (Figure 3.5). This suggested that either groundwater was being lost from both lakes, or else that $B_{\text{calculated}}$ was an overestimate. Over the three year study, the cumulative difference between $B_{\text{calculated}}$ and $V_{\text{measured}}$ (defined as G) in Lake Alexandrina was 225 GL with 139 GL of this loss occurring in 2009. For Lake Albert, the net loss of surface water to G was 22.6 GL, which consisted of losses of 27.5 GL in 2007, 0.5 GL in 2009 and a gain of 5.4 GL in 2008. According to these estimates, the direction of groundwater flow was generally away from both lakes.
Over the three years, the total annual input of water was lowest in 2007, with inputs increasing in subsequent years. Over the same period, water losses remained relatively constant (Table 3.1). Consequently, the net rate of water loss was highest for both lakes in 2007, with 655 mm yr\(^{-1}\) and 656 mm yr\(^{-1}\) lost from Lake Alexandrina and Lake Albert, respectively.

### 3.3.2 Changes to water quality during water level decline

The water quality characteristics measured in this study tended to increase in concentration as water levels declined (Figure 3.6). Salinity increased as water level declined in Lake Alexandrina \((p < 0.0001)\) and Lake Albert \((p < 0.0001; \text{Table 3.2})\). The rate of salinity increase in Lake Albert was higher than Lake Alexandrina because water inputs had a greater concentration of salt than those in Lake Alexandrina, there was no loss of salt from the lake, and the loss of water to evaporation was higher as a proportion of volume in Lake Albert compared with Lake Alexandrina. Differences between Lake Alexandrina and Lake Albert for other parameters were not as obvious, although Lake Albert generally had higher concentrations of nutrients, SPM and DOC and higher \(k_d\) values.

Total nutrient concentrations increased as water level declined in Lake Alexandrina (TP: \(p < 0.0001\); TN: \(p < 0.0001\)). For Lake Albert, TN increased significantly with decreasing water levels \((p < 0.0001)\), but TP \((p = 0.06)\) did not (Table 3.2; Figure 3.6). Conversely, FRP concentrations decreased with declining water levels in a weak, but significant manner for Lake Albert \((p = 0.04)\), whereas there was no relationship between FRP and water level in Lake Alexandrina \((p = 0.80)\).

SPM concentrations increased with declining water levels in Lake Alexandrina \((p = 0.0009)\) but not in Lake Albert \((p = 0.14; \text{Table 3.2})\). However, SPM concentrations were higher in Lake Albert than in Lake Alexandrina. DOC increased with declining water levels in both lakes, with concentrations increasing faster (ANCOVA: \(F = 17.0, p = 0.0002\)) for Lake Albert \((p < 0.0001)\) than Lake Alexandrina \((p < 0.0001)\).
Similarly, $k_d$ in both Lake Alexandrina ($p < 0.0001$) and Lake Albert ($p = 0.005$) increased as water levels declined, but the rate of change was not significantly greater in Lake Albert (ANCOVA: $F = 4.12$, $p = 0.054$). Concentration of chla increased with water level decline in Lake Alexandrina ($r^2 = 0.35$, $p < 0.05$), but not in Lake Albert ($p = 0.16$).

Concentrations of chla increased significantly with time in Lake Alexandrina ($< 0.05$), but not in Lake Albert ($p = 0.07$; Figure 3.7). There was no apparent seasonality in chla concentrations in the time series data.

### 3.3.3 Mass balance of TP, FRP and SPM

Despite 2007 having the lowest total water input during the analysis period, the input of TP ($0.110 \pm 0.005$ g m$^{-2}$ yr$^{-1}$) and SPM ($33.6 \pm 1.6$ g m$^{-2}$ yr$^{-1}$) was higher than in subsequent years for Lake Alexandrina (Table 3.3). This led to a decline in net retention of TP and SPM between 2007 and 2009, a fact that was exacerbated by the active removal of TP and SPM from Lake Alexandrina as water was pumped into Lake Albert. The declining TP input, increasing lake concentration and active pumping of this water out of Lake Alexandrina resulted in a net loss of TP in 2009. These changes were driven by decreasing concentration of TP and SPM in the inflowing River Murray water, with mean TP concentration declining from 0.11 mg L$^{-1}$ in 2007 to 0.05 mg L$^{-1}$ in 2009 and mean SPM concentration declining from 35.4 mg L$^{-1}$ in 2007 to 14.0 mg L$^{-1}$ in 2009. A similar decline was observed for mean annual River Murray FRP concentration, which was 0.015 mg L$^{-1}$ in 2007 and 0.008 mg L$^{-1}$ in 2009.

The cumulative retention of SPM in Lake Albert between 2007 and 2009 of 186.4 g m$^{-2}$, was four times larger than that in Lake Alexandrina, of 45.1 g m$^{-2}$. Water quality in Lake Albert deteriorated rapidly throughout the study period as an earthen embankment prevented the exchange of water with Lake Alexandrina, and turbidity and nutrient concentration of the water pumped into Lake Albert increased. TP
retention in Lake Albert increased from $0.105 \pm 0.005 \text{ g m}^{-2} \text{ yr}^{-1}$ in 2007 to $0.139 \pm 0.009 \text{ g m}^{-2} \text{ yr}^{-1}$ in 2009. The net retention of SPM also increased from $30.3 \pm 1.8 \text{ g m}^{-2} \text{ yr}^{-1}$ in 2007 to $76.8 \pm 6.4 \text{ g m}^{-2} \text{ yr}^{-1}$ in 2009. Despite an increasing retention of TP, the retention rates of FRP declined from $0.020 \pm 0.0008 \text{ g m}^{-2} \text{ yr}^{-1}$ in 2007 to $0.016 \pm 0.0008 \text{ g m}^{-2} \text{ yr}^{-1}$ in 2009.

The variation in total mass of nutrients in the water column between months was often an order of magnitude greater than the monthly sum of inputs and outputs calculated using the water transfers and average nutrient concentrations (Figure 3.8 & 3.9). These differences were dominated by large variation between adjacent sampling periods that were independent of external inputs and outputs. For example in Lake Alexandrina, the increase in mass of TP calculated from lake volume and nutrient concentrations in September 2008, compared with that of the preceding sampling period, was 73 tonnes. Variation between the mass of SPM calculated using lake volume and average nutrient concentration (hereafter ‘measured mass’) in Lake Alexandrina was similar to that of TP as concentrations of the two parameters were correlated ($r^2 = 0.56$, Figure 3.10). The corresponding relationship between TP and SPM concentration was not as strong in Lake Albert ($r^2 = 0.14$). Similarly, the measured mass of FRP varied over a larger range than the sum of monthly inputs and outputs, however, this variance was greater in Lake Alexandrina than in Lake Albert. Lake-wide FRP concentrations were not correlated to TP or SPM in either lake.

### 3.4 Discussion

#### 3.4.1 Validity of water balance

While groundwater is often estimated as the difference between water volume and the balance of all other inputs and outputs in a water balance (e.g. Özen et al. 2010), this method is prone to error because of the relative differences between water volumes used in the calculation. Over the three years presented in this study, outputs
were 225 GL higher than inputs for Lake Alexandrina, while outputs were 22 GL higher than inputs in Lake Albert. This was initially attributed to a loss of 10% of the volume of both lakes to groundwater over the study period, which is a very high estimate compared with other studies. For instance, a report that monitored 5 wells around Lake Alexandrina suggested that even during low lake levels, groundwater was a net source of water into the lakes, the magnitude of which was estimated to be 188 ML yr\(^{-1}\) (Walsh & Martin 2009). Another study suggested that during wetter years the eastern shore of Lake Alexandrina receives up to 3 GL yr\(^{-1}\) of groundwater through aquifers fed from the Mount Lofty Ranges to the East, a relatively high rainfall catchment compared with other regions surrounding both lakes (Telfer et al. 2004). This suggests that the groundwater loss reported in this study may reflect an artifact of its method of calculation rather than any substantial loss to groundwater.

The large error in groundwater estimates can be explained by much smaller errors in other inputs and outputs. For example, it was assumed that evaporation was limited to the surface area of water in each lake and that this could be estimated as 1.23 times the reference evaporative transpiration ET\(_0\). Using this method, total evaporation from both lakes averaged 1101.3 GL yr\(^{-1}\), or 1593 mm yr\(^{-1}\). This is higher than other estimates of annual evaporative loss from Lake Alexandrina, including that of 1330 mm modeled by McJannet et al. (2008) for 1978 and 1286 mm measured by Cheng (1978) using an energy budget and aerodynamic method in 1974. Both of these estimates were performed during normal lake conditions. Water temperatures, and hence evaporation, are likely to be higher under the low water level conditions of this study. Furthermore, annual mean maximum temperature data for Meningie shows that 1974 was approximately 1°C colder than all years included in this study (Bureau of Meteorology, unpublished data). Perhaps a more significant introduced error than the rate of evaporation is the surface area from which evaporative losses occur. These surfaces are most likely greater than the surface area of water in both lakes. The periphery of Lake Alexandrina is dominated by sandy,
permeable sediments (Chapter 4) that were exposed for most of this study. Wind-induced seiching is common to both lakes, causing frequent, albeit temporary, incursions of surface water over exposed sediments. Water retained within the sediments could be lost to evaporation once the seiching or lake-tilt subsided. This process would essentially increase the surface area from which lake water is evaporated. Indeed, piezometer data suggests that during low water levels, evaporation was occurring from shallow, lake-fed groundwater in sediments up to 100 metres away from the shoreline of Lake Alexandrina (Luke Mosley, personal communication). However, insufficient data availability prevents these estimates from being reliably quantified.

The total water balance for both lakes over the three year study period showed that 1809 GL of water entered the lakes from the River Murray, 89 GL of water flowed from the tributaries of Lake Alexandrina, rainfall provided 787 GL of water to both lakes to give a total input of 2685 GL, while evaporative losses totaled 3543 GL. Because these components are linearly related to the calculated water balance, errors in the water balance introduced by errors in the rate of evaporation would be proportionally higher than errors of the various inputs. A simple sensitivity analysis shows that by increasing the rate of evaporation by 5% in both lakes, the total calculated groundwater losses is reduced from 247 GL to 90 GL. An evaporation rate 10% higher than that used in this study, leads to a net gain of 66 GL of groundwater, which is likely a substantial overestimate for reasons discussed previously. Conversely, reducing inflows from the River Murray by 10% reduces groundwater loss from 247 GL to 78 GL. Given that evaporation is likely to be an underestimate of actual losses and that increases to the evaporation rate of between 5-10% have a pronounced effect on estimates of groundwater flow (bringing it within the realm of other studies), the water balance presented here can be considered as broadly representative of the water level drawdown observed during this study. Since nutrient concentrations were unaffected by evaporation and groundwater flow had only a
small effect on FRP concentrations, associated errors in the nutrient mass balance are likely to be minor. However, *in situ* adsorption and desorption of phosphate with suspended clay particles may influence the measured concentration of FRP and hence the mass balance.

### 3.4.2 Influence of water level decline on water quality

Water level decline affected nutrient retention and water quality in both lakes. TP inputs from the River Murray decreased during the drought period as a result of lower nutrient input from the catchment and enhanced sedimentation within the river channel (Mosley *et al.* In Prep). In addition, as water levels declined, increasing concentration of nutrients resulted in a greater mass of TP being pumped into Lake Albert, thereby increasing the rate of TP removal from Lake Alexandrina. TP retention decreased from 0.11 g m\(^{-2}\) yr\(^{-1}\) in 2007 to negligible retention rates in Lake Alexandrina as a result of decreasing inputs and increasing outputs. These TP retention rates contrasted to the significantly higher retention for Lake Alexandrina during normal water levels, estimated as 0.84 g P m\(^{-2}\) yr\(^{-1}\) by Cook *et al.* (2009) between 1979 and 1996. These observations were reversed in Lake Albert, where higher concentrations of TP in pumped water inputs increased retention rates of TP from 0.11 g P m\(^{-2}\) yr\(^{-1}\) in 2007 to 0.14 g P m\(^{-2}\) yr\(^{-1}\) in 2009.

While TP retention in Lake Alexandrina during the period of water level decline was low, the monthly variation in measured mass of TP in the water column was much larger than the total inputs or outputs shown in the nutrient balance (Figure 3.8 & Figure 3.9). In shallow lakes, sediment is often a dominant source of phosphorus, through the flux of soluble nutrients (Burger *et al.* 2007) or by resuspending sediment particles (Kristensen *et al.* 1992). The significant relationship between TP and SPM, in conjunction with the large variation in monthly mass of TP and SPM, suggests that sediment resuspension is driving changes in these water quality parameters. These observations are consistent with those of Geddes (1984), who
observed that turbidity was related to TP in Lake Alexandrina. Geddes (1984) hypothesised that the resuspension of sediments was the cause of the high turbidity, although turbid inflows from the Darling River, a significant tributary of the River Murray, may also have contributed to the elevated turbidity (Geddes 1988).

In shallow lakes, sediments are readily resuspended into the water column when turbulent energy at the sediment surface is sufficient to overcome countervailing influences that keep sediments deposited (Bengtsson et al. 1990; Bloesch 1995). This resuspension of sediments has been previously linked to changes in both TP (Kristensen et al. 1992) and FRP (Sondergaard et al. 1992). As water levels decline, the depth through which wind-induced wave energy is attenuated before it reaches the sediment surface is reduced, leading to more frequent resuspension events of higher magnitude (Effler & Matthews 2004). Regression between measured water quality parameters and water level support the hypothesis of increasing resuspension of sediments as water levels decline. SPM and total nutrients (TP & TN) increased in concentration at lower water levels in Lake Alexandrina, despite decreasing annual retention of TP. The increasing resuspension of sediments showed obvious impacts on the light attenuation, which increased in line with higher SPM and DOC concentrations as water levels declined. This resuspension provides a source of nutrients into the water column (Nagid et al. 2001) that, along with increasing light attenuation, can advantage pelagic productivity over the benthic productivity of microalgae and macrophytes (Scheffer et al. 2001).

In semi-arid and arid regions, water level declines are likely to increase in frequency from climate change and anthropogenic demand for freshwater (Wetzel 1992; Alcamo et al. 2007). The results presented in this study suggest that water level declines in these regions could lead to increased eutrophication and light attenuation, both factors that would promote or maintain a turbid state in shallow lakes. This is in contrast to studies on drought and water level decline in temperate lakes in the northern hemisphere, which suggest that a decreased external nutrient supply would
push shallow lake ecosystems into a transient clear water state that reverts to a turbid state when nutrient inputs increase again (Scheffer & van Nes 2007; van Geest et al. 2007). The results presented here are consistent with those of Özen et al. (2010), who showed that internal nutrient concentrations increased in two shallow semi-arid lakes during drought, despite lower fluvial input of nutrients. This suggests that internal processes, such as sediment resuspension, play a more dominant role in the ecosystem state of shallow lakes in semi-arid regions that may be exacerbated by declining water levels.
Table 3.1: Total annual inputs, outputs and balance (± standard deviation) of water for Lake Alexandrina and Lake Albert, 2007-2009. Data shown are in millimetres of water depth per year (mm yr\(^{-1}\)).

<table>
<thead>
<tr>
<th>Year</th>
<th>Total inputs</th>
<th>Total outputs</th>
<th>Balance</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>mm yr(^{-1})</td>
<td>mm yr(^{-1})</td>
<td>mm yr(^{-1})</td>
</tr>
<tr>
<td>Lake Alexandrina</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2007</td>
<td>1431 ± 41</td>
<td>2086 ± 75</td>
<td>-655 ± 87</td>
</tr>
<tr>
<td>2008</td>
<td>1542 ± 40</td>
<td>1916 ± 70</td>
<td>-374 ± 76</td>
</tr>
<tr>
<td>2009</td>
<td>1663 ± 44</td>
<td>2072 ± 65</td>
<td>-409 ± 81</td>
</tr>
<tr>
<td>Lake Albert</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2007</td>
<td>1473 ± 42</td>
<td>2129 ± 74</td>
<td>-656 ± 100</td>
</tr>
<tr>
<td>2008</td>
<td>1503 ± 47</td>
<td>1846 ± 72</td>
<td>-343 ± 93</td>
</tr>
<tr>
<td>2009</td>
<td>1751 ± 56</td>
<td>1913 ± 63</td>
<td>-162 ± 108</td>
</tr>
</tbody>
</table>
Table 3.2: Non-linear regressions between water quality parameters and water level (WL) for Lake Alexandrina and Lake Albert. ‘ns’ indicates a non-significant regression. TP: total phosphorus; TN: total nitrogen; FRP: filterable reactive phosphorus; SPM: suspended particulate matter; DOC: dissolved organic carbon; $k_d$: light attenuation coefficient; chl$\alpha$: chlorophyll $\alpha$.

| Parameter | Lake Alexandrina | | Lake Albert | |
|-----------|-----------------|-----------------|-----------------|
| Salinity (S) | $S = 1.4e^{-0.7WL}$ | $r^2 = 0.86, p < 0.0001$ | $S = 1.8e^{-0.5WL}$ | $r^2 = 0.90, p < 0.0001$ |
| TP | $TP = 0.12e^{-0.7WL}$ | $r^2 = 0.57, p < 0.0001$ | ns | $p = 0.06$ |
| TN | $TN = 1.8e^{-0.7WL}$ | $r^2 = 0.61, p < 0.0001$ | $TN = 2.6e^{-0.7WL}$ | $r^2 = 0.51, p < 0.0001$ |
| FRP | ns | $p = 0.80$ | FRP | $0.01e^{0.6WL}$ | $r^2 = 0.17, p = 0.04$ |
| SPM | $SPM = 86e^{-0.5WL}$ | $r^2 = 0.37, p = 0.0009$ | ns | $p = 0.14$ |
| DOC | $DOC = 8.5e^{-0.3WL}$ | $r^2 = 0.66, p < 0.0001$ | $DOC = 13e^{-0.6WL}$ | $r^2 = 0.65, p < 0.0001$ |
| $k_d$ | $k_d = 3.6e^{-0.6WL}$ | $r^2 = 0.77, p < 0.0001$ | $k_d = 7.2e^{-0.6WL}$ | $r^2 = 0.58, p = 0.005$ |
| chl$\alpha$ | $chl\alpha = 48e^{-0.6WL}$ | $r^2 = 0.35, p < 0.05$ | ns | $p = 0.16$ |
Table 3.3: Total annual inputs, outputs and retention (± standard deviation) of nutrients and suspended particulate matter for Lake Alexandrina and Lake Albert. The ‘—’ indicates that no outputs occurred during the year because SPM was not lost to evaporation or groundwater and pumping had not begun.

<table>
<thead>
<tr>
<th>Year</th>
<th>Total inputs</th>
<th>Total outputs</th>
<th>Retention</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>g m⁻² yr⁻¹</td>
<td>g m⁻² yr⁻¹</td>
<td>g m⁻² yr⁻¹</td>
</tr>
<tr>
<td>Lake Alexandrina</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TP</td>
<td>2007</td>
<td>0.110 ± 0.0045</td>
<td>0.002 ± 0.0002</td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>0.074 ± 0.0024</td>
<td>0.019 ± 0.0012</td>
</tr>
<tr>
<td></td>
<td>2009</td>
<td>0.041 ± 0.0015</td>
<td>0.046 ± 0.0023</td>
</tr>
<tr>
<td>FRP</td>
<td>2007</td>
<td>0.022 ± 0.0006</td>
<td>0.002 ± 0.0002</td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>0.018 ± 0.0005</td>
<td>0.001 ± 0.0001</td>
</tr>
<tr>
<td></td>
<td>2009</td>
<td>0.020 ± 0.0006</td>
<td>0.001 ± 0.0002</td>
</tr>
<tr>
<td>SPM</td>
<td>2007</td>
<td>33.59 ± 1.6</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>21.55 ± 0.85</td>
<td>17.60 ± 1.20</td>
</tr>
<tr>
<td></td>
<td>2009</td>
<td>16.57 ± 0.66</td>
<td>9.01 ± 1.45</td>
</tr>
<tr>
<td>Lake Albert</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TP</td>
<td>2007</td>
<td>0.110 ± 0.0050</td>
<td>0.004 ± 0.0006</td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>0.109 ± 0.0052</td>
<td>0.002 ± 0.0002</td>
</tr>
<tr>
<td></td>
<td>2009</td>
<td>0.141 ± 0.0065</td>
<td>0.002 ± 0.0004</td>
</tr>
<tr>
<td>FRP</td>
<td>2007</td>
<td>0.025 ± 0.0006</td>
<td>0.004 ± 0.0005</td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>0.014 ± 0.0006</td>
<td>0.002 ± 0.0001</td>
</tr>
<tr>
<td></td>
<td>2009</td>
<td>0.018 ± 0.0007</td>
<td>0.002 ± 0.0003</td>
</tr>
<tr>
<td>SPM</td>
<td>2007</td>
<td>30.33 ± 1.78</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>79.24 ± 5.14</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>2009</td>
<td>76.80 ± 6.44</td>
<td>—</td>
</tr>
</tbody>
</table>
Figure 3.1: Water sampling sites and significant geographical features in the Lower Lakes. Sites sampled between January 2007 and December 2009 are shown with closed circles, while additional sites sampled between August 2008 and December 2009 are shown with open circles. Dotted lines show the location of the embankments at Narrung and Clayton that were built during the course of this study and had water pumped from Lake Alexandrina into Lake Albert or towards Goolwa. The wavy line indicates the separation (for purposes of nutrient mass calculations only) between Lake Alexandrina and the Goolwa Channel, which extended to the Goolwa Barrage until the construction of the embankment at Clayton was complete. At this time, the embankment became the new boundary of the Goolwa Channel. Bold lines indicate 5 barrages built between 1935 and 1940 that separate Lake Alexandrina from the Southern Ocean.
Figure 3.2: Changes in water level (---) and salinity (——) of Lake Alexandrina, January 2007-April 2010. Dates when water samples were collected are also indicated (△).
Figure 3.3: Monthly water balance for Lake Alexandrina. Components of the water balance and the net change in volume calculated using bathymetry and water level are shown.
Figure 3.4: Monthly water balance for Lake Albert. Components of the water balance and the net change in volume calculated using bathymetry and water level are shown.
Figure 3.5: Comparison between calculated water balance and measured water volume of both lakes. Monthly measured volume of Lake Alexandrina (Top) and Lake Albert (Bottom) is compared with estimated volume using the water balance, but excluding calculated groundwater. Correlations between presented values are $r^2 = 0.92$ (S.E. of slope = 0.06) for both lakes.
Figure 3.6: The relationship between water quality parameters and water level in Lake Alexandrina (closed circles) and Lake Albert (open circles). SPM: Suspended Particulate Matter; TP: Total Phosphorus; TN: Total Nitrogen; $K_d$: Light attenuation coefficient; DOC: Dissolved Organic Carbon. Water level is measured in metres above or below Australian Height Datum (AHD) where zero is mean sea level.
Figure 3.7: Time series of chlorophyll $a$ data for the duration of the study. Data shown are from the middle of Lake Alexandrina and Lake Albert.
Figure 3.8: Mass balance of TP (top panel), FRP (middle panel) and SPM (bottom panel) in Lake Alexandrina. Columns show the monthly change from sources of input and output (left axis) and the moving line shows the measured change in nutrient mass between months (right axis).
Figure 3.9: Mass balance of TP (top panel), FRP (middle panel) and SPM (bottom panel) in Lake Albert. Columns show the monthly change from sources of input and output (left axis) and the moving line shows the measured change in nutrient mass between months (right axis).
Figure 3.10: Relationship between TP and SPM measured from integrated water samples in Lake Alexandrina.
4. Effects of water level decline on the composition and distribution of surface sediments in the large, shallow Lake Alexandrina, South Australia

4.1 Introduction

In semi-arid and arid regions, lakes are exposed to large fluctuations in water level as a result of significant variability in seasonal and annual rainfall and surface run-off. These regions are forecast to undergo disproportionate changes to water availability as a result of human-induced global climatic changes and water extraction (Wetzel 1992; Vörösmarty et al. 2000; Vörösmarty & Sahagian 2000), which is expected to increase the frequency and magnitude of water level variation (Coops et al. 2003; Cai & Cowan 2008). A better understanding of the effect of water level decline on the movement and composition of sediments is therefore essential in forecasting ecosystem responses and identifying management options.

Water level decline in lakes can cause major changes in the biogeochemistry of aquatic ecosystems. For example, reduced water depth increases the transfer of oxygen into sediments, shifting patterns of organic matter mineralisation from anoxic methanogenesis to oxic carbon dioxide production (Juutinen et al. 2001; Yamamoto
Reduced water depth also influences the distribution of sediments, with sediment character an important factor controlling biogeochemical processes.

As water levels decline, the resuspension and redistribution of sediments can increase if the downward movement of the wave-mixed layer towards the sediment surface influences a larger area (Nagid et al. 2001), or disturbs sediments that are more easily resuspended (Håkanson & Jansson 1983; Håkanson 2005). Increased areas of shallow water mean that for any given wind speed, there will be an increased likelihood that the shear stress at the sediment surface exceeds that necessary to entrain sediment particles (Evans 1994; Håkanson et al. 2000). This leads to greater erosion of the fine sediments and organic matter that had accumulated under deeper waters as they become exposed to increased shear stresses and wave action (Gottgens 1994; Furey et al. 2004; Punning et al. 2006). These eroded sediments can then be either transported to deeper, sheltered water or become distributed throughout the lake, depending on the lake morphology. Increasing sediment erosion reduces the abundance of rooted macrophytes by decreasing the availability of suitable substrate (Turner et al. 2005). As the abundance of macrophytes decreases sediments become more exposed and prone to resuspension, escalating the loss of substrate (James et al. 2004; Horppila & Nurminen 2005; Huang et al. 2007).

Water level drawdown has been shown to consolidate sediment by enhancing the mineralisation of organic matter, thereby increasing sediment density and decreasing its propensity to be resuspended (James et al. 2001). Reductions in sediment water content and organic carbon following complete drawdown were implicated in increased sediment consolidation and reduced turbidity upon refill of Lake Kraenepoel (van Wichelen et al. 2007). However, exposing sediments to the atmosphere poses other risks to lake ecosystems, such as enhanced nutrient release upon refill (Baldwin & Mitchell 2000) and sulfide oxidation and acidification in soils containing sulfidic material (van Wichelen et al. 2007; Simpson et al. 2010).
Gottgens (1994) has shown a decrease of organic matter in inundated, peripheral sediments in Newnans Lake, Florida, as water levels were partially drawn down. Consequently, partial drawdown may provide the advantages of water level drawdown, namely, enhanced sediment erosion of newly formed littoral zone to reduce sediment resuspension upon refill, while limiting the adverse affects of a complete drawdown. Partial drawdown may have other benefits too, for example, increasingly oxic surface sediments generally retain more phosphorus, thereby decreasing internal loading (Gunnars & Blomqvist 1997; Sondergaard et al. 2003). However, there appear to have been very few studies linking the changing distribution of sediments during partial water level decline to the consequent changes in the abundance and distribution of nutrients.

This study aims to investigate the effects that declining water levels in Lake Alexandrina have on (i) the redistribution of surface sediments and (ii) the abundance and distribution of carbon, nitrogen and phosphorus in the surface sediments.

### 4.2 Methods

#### 4.2.1 Site description

Lake Alexandrina (35.42784°S 139.16142°E; Figure 4.1) is a shallow, eutrophic lake at the end of the River Murray, in the semi-arid region of South Australia described in detail in section 2.2. As a result of barrage construction, lake levels were reasonably constant for the period 1935-2006, fluctuating less than 0.4 m in previous droughts. Since 2001, low inflows from the River Murray prevented any flow out of the lakes across the barrages. Record low flows into Lake Alexandrina in the summer of 2006-2007 due to a multi-year regional drought (Leblanc et al. 2009) resulted in rapid water level decline. For the duration of this study, which commenced with sediment sampling in 2007, the rate of drawdown was greater than 0.5 m per year.
Water levels reached -0.9 m AHD at the completion of the study (February 2009) when lake water volume was 36% of the long-term mean (Figure 4.2). This study presents a comparison of the spatial distribution and composition of surface sediments before (February 2007) and during (February 2009) this extreme water level decline with minimum water levels of -1.1 m AHD reached in April 2009.

4.2.2 Sediment analysis

Sediment samples from February 2007 were collected as part of an earlier study (Aldridge et al. 2009) which involved collecting single samples from 22 sites arranged in five transects that spanned from the lake shoreline to opposite shoreline (Figure 4.1). These sites were revisited in February 2009, when triplicate sediment cores were collected. Sediments were analysed individually and the results averaged for each site. Ten peripheral sites were above the waterline, but were sampled nonetheless. These sites were classified by their level of connectivity with the lake as either completely dry (5 sites), or still wet as a result of wind-induced seiching (5 sites). This reduced the proportion of sites that had sediments inundated by shallow water and so three additional sites were sampled in 2009. A total of 73 sediment samples were collected in 2009, as one site had only a single sediment core collected.

Sediment cores approximately 20 cm in length were collected using a cylindrical PVC corer (internal diameter 5.8 cm). The corer was pushed vertically into the sediment, sealed on top and raised vertically out of the sediment. The overlying water, when present, was siphoned off and the topmost 1 cm of sediment was extruded using a piston, homogenised and stored on ice in the dark until return to the laboratory. Under normal water levels, a long-term sedimentation rate of 3 mm yr\(^{-1}\) (Herczeg et al. 2001) suggests that the topmost 1 cm of sediment represents approximately 3 years of deposition.

An aliquot of fresh sediment (~6 g) was diluted for analysis of sediment particle size distribution (PSD) using a Laser In-Situ Scattering and Transmissometry instrument
fitted with a cuvette (LISST-100, Sequoia Scientific, Washington). Each sample was suspended in deionised water (20 mL), shaken to disaggregate any weakly flocculated particles and then further diluted until laser transmission was greater than 30% to avoid multiple scattering (Agrawal et al. 2008). For each sample, the beam attenuation of the 32 photodiodes of the LISST-100 were inverted using a kernel matrix derived for randomly shaped particles (Agrawal et al. 2008) to give PSD. These transformed data were combined into six particle size categories, reporting each category as a proportion of the total volume concentration: <4.43 µm (clay), 4.43 – 6.24 µm (very fine silt), 7.36 – 19.9 µm (fine silt), 23.5 – 63.3 µm (silt), 74.7 – 186 µm (fine sand), and >219 µm (sand). A second aliquot of fresh material was sieved through a 600 µm stainless steel mesh to determine if particles larger than 600 µm were present in sediment samples.

The remaining sediment was weighed (fresh weight) and dried to constant weight at 55°C to determine dry weight. The sediment water content (WC) was estimated by difference between these two and reported as a percentage of fresh weight. Loss-on-ignition (LOI) was determined gravimetrically following combustion at 550°C for one hour and reported as a percentage of dry weight (Boyle 2004; Eaton et al. 2005) or wet weight for use in equation 4.1. Total carbon (TC) and total nitrogen (TN) were determined by combustion of a known mass of dry weight to 1300°C with a LECO CNS 2000 Analyzer (Environmental Analysis Laboratory, Southern Cross University, New South Wales, National Association of Testing Authorities endorsed). Organic carbon (OC) was estimated to be half of the measured LOI (Nelson & Sommers 1982). When OC was greater than TC using this method, it was assumed that all carbon was in an organic form and equated with the TC. Inorganic carbon (IC) was calculated as the difference between TC and OC. Total phosphorus (TP) was determined following digestion of a known mass of sediment dry weight at 150°C in aqua regia and analysis of the supernatant with inductively coupled plasma-
atomic emission spectrometry (ICP-AES; Waite Analytical Service, University of Adelaide, South Australia).

4.2.3 Water sampling

Water samples were collected between October 2007 and March 2009 from three sites according to methods outlined in Chapter 3 (Figure 4.1) for later use in calculating the saturation index of calcium carbonate. Depth profiles of temperature, pH and electrical conductivity were measured at intervals of 0.25 m using sensors on a pre-calibrated multi-probe sonde (TPS-90FLT, TPS, Queensland). Total alkalinity and bicarbonate concentrations were determined using automated acidimetric titration (Australian Water Quality Centre, NATA endorsed). Samples for calcium analysis were collected in acid-washed polyethylene bottles and concentrations were determined using ICP spectrometry. Field measurements of pH were replicated using the electrometric method from standard methods upon return to the laboratory (Eaton et al. 2005, Australian Water Quality Centre, NATA endorsed).

4.2.4 Data analysis

Bulk density was calculated from WC and LOI according to the formula of Håkanson and Janson (1983)

\[
\rho = \frac{100 \times \rho_m}{100 + (WC + IG_o) \times (\rho_m - 1)}
\]  

(4.1)

where \( \rho_m \) is the density of inorganic particles taken to be 2.6 g cm\(^{-3}\) and IG\(_o\) is the loss-on-ignition reported as a percentage of fresh weight. This calculated bulk density and the depth of the core sample were used for areal conversions of TC, OC, IC, TN and TP from mg g\(^{-1}\) to g m\(^{-2}\) using equation 4.2, where \( d \) is the depth of the sediment sample.

\[
\text{ArealMass}(g.m^{-2}) = \rho(g.cm^{-3}) \times d(cm) \times \text{Concentration}(mg.g^{-1})
\]  

(4.2)
Data from sediment samples collected in 2007 and in 2009 were analysed separately and measured parameters from each site or replicate were pooled for statistical comparison. Analysis revealed two distinct sediment types, so differences in characteristics between sediment types and between sampling years were tested using a two-way ANOVA. When significant interactions were found, a comparison of means using Tukey’s HSD post hoc test was performed. In addition, differences between sediment types in any one year were tested with a one-way ANOVA and Tukey’s test. All statistical analyses were conducted with JMP-IN 8.0 (SAS Institute Inc.) with an α of less than 0.05 deemed significant. Normality was tested with the Shapiro-Wilk test for goodness of fit and data were log-transformed where necessary.

The saturation index of calcium carbonate in the water column (SI) was estimated using pH and the concentration of components of the bicarbonate equilibrium according to standard methods (Eaton et al. 2005). This method provides an indication of the likely precipitation or dissolution reactions of calcium carbonate. SI is positive when calcium and bicarbonate concentrations suggest over-saturation and likely precipitation of calcium carbonate, while negative values signify an under-saturated state that would favour calcium carbonate dissolution.

4.3. Results

4.3.1 Sediment composition and bimodality

Water content (WC) of Lake Alexandrina sediments in 2009 was related to water depth ($r^2 = 0.81, p < 0.0001$, Figure 4.3A). A significant relationship between water depth and LOI was also found, but this explained a lower proportion of the variation ($r^2 = 0.57, p < 0.0001$, Figure 4.3B). There was a transition between sediment types
at a water depth of approximately 1.4 meters. Regressions between WC and water depth were significant in sediments shallower than 1.4 m ($r^2 = 0.36, p < 0.0001$; Figure 4.3A) but not in sediments deeper than 1.4 m. Regressions between water depth and LOI were also not significant when separated by a depth of 1.4 m (Figure 4.3B). Sediment samples in 2009 appeared to be bimodal, grouping into two distinct categories according to WC ($F = 648.9, p < 0.05$, Figure 4.4). The first category, which had WC less than 40%, was classified as sand (mean WC of $17.4 \pm 1.6\%$). The second category, with WC greater than 55%, was classified as mud ($76.7 \pm 1.7\%$). The variance of the distribution ($\sigma \pm$ standard error) for sand ($9.4 \pm 1.4$) was similar to mud ($9.3 \pm 1.7$).

All sandy sediments had LOI values below 2.8% (Table 4.1) and were dominated by larger particles, with a peak nominal diameter of 238 µm (Figure 4.5). Sediments categorised as mud had LOI values above 4.1% (Table 4.1) and a peak in particle size distribution at 19.8 µm (Figure 4.5). Particles larger than 600 µm were only present in very small amounts at three sites and consisted of woody debris (2 sites closest to the River Murray in mud sediments) or calcareous shells (1 site in dry sediment close to Goolwa).

Sediment WC in 2007 also suggested a bimodal distribution of sediments into two distinct categories ($F = 509.9, p < 0.05$, Figure 4.4). Sand had WC below 24% (mean WC $20.5 \pm 1.7\%$), while mud had WC above 63% ($75.7 \pm 1.7\%$). The variance of the distribution for sand ($2.3 \pm 0.7$) was lower than for mud ($7.8 \pm 2.3$). LOI for all sandy sediments in 2007 was below 0.7% (Table 4.1). For mud, LOI values were all above 3.0% (Table 4.1). Particle size distribution peaked at 170 µm for sand and 45.3 µm for mud (Figure 4.5). Sand and mud were also distributed by depth in 2007 (WC: $r^2 = 0.63, p < 0.05$, LOI: $r^2 = 0.60, p < 0.05$). Sandy sediments were found in shallow waters and mud was generally found in deep water, however, the water depth of transition was deeper than in 2009, at about 2.5 m, reflecting a decline in water levels
of 1.3 m. One site near Goolwa was an exception to this sediment distribution (depth 0.9 m, WC 68%, LOI 17.6%), where macrophytes were present in 2007.

4.3.2 Changes to sediment particle size distribution

Two-way ANOVA and subsequent post-hoc analysis showed significant changes in the size distribution of sediment particles between sediment type and year (Table 4.1 and Table 4.2). Changes to particle composition had occurred in both sand and mud over the two year period. For sand, there was a significant increase in fine silt (7.36 – 19.9 µm) between 2007 and 2009, while in mud, there was a significant increase in all three categories below 19.9 µm (Clay <4.43 µm, Very Fine Silt 4.43 – 6.24 µm, Fine Silt 7.36 – 19.9 µm) and a significant decrease in fine sand particles (74.7 – 186 µm; Table 4.1). Further division of sandy sediments according to the level of water inundation in 2009, showed that all particle size categories with a nominal diameter below 63.3 µm increased in inundated sand, however, the increase was only significant for very fine silts and fine silts (Table 4.3).

4.3.3 Chemical characteristics of sediments

LOI increased between 2007 and 2009 in mud, but not in sand (Table 4.1). However, the increase in LOI in sand between 2007 and 2009 was only significant at $p = 0.09$. Areal concentration of TC decreased significantly in sand between 2007 and 2009 as water levels declined, which was predominantly due to the loss of IC (Table 4.1). For mud, TC increased slightly, but not significantly. Because LOI increased in mud, there was a non-significant increase in OC and a corresponding decrease in IC. TN and TP showed a similar pattern to TC of significant decreases in sand, but not in mud. However, the level of inundation of sediments with water strongly influenced the mass ratio of TC, TN and TP (Table 4.4), with the decline in TC relative to TN and TP largest in sites that remained inundated in 2009. The mass ratio of TC to TN
was highest in 2007 for sand and the significant decrease in 2009 was dominated by lower C:N ratios in inundated and wet sands.

Linear regressions revealed that LOI was positively correlated to TC in 2009 for both sand ($r^2 = 0.75, F = 122.9, p < 0.05$) and mud ($r^2 = 0.78, F = 97.5, p < 0.05$), but in 2007 no relationship was found for either sand ($p = 0.5$) or mud ($p = 0.2$; Table 4.5 and Figure 4.6). Similarly, for 2009 data, when LOI was regressed against TN, a significant positive relationship was found for sand ($r^2 = 0.38, F = 25.2, p < 0.05$) and mud ($r^2 = 0.76, F = 90.8, p < 0.05$) in 2009, but not for 2007. The regression between LOI and TP showed a positive relationship for mud in 2009 ($r^2 = 0.45, F = 22.8, p < 0.05$), but not for sand ($r^2 = 0.06, F = 2.5, p = 0.1$). Neither sand nor mud showed any significant relationship between LOI and TP in 2007.

A positive relationship between LOI and volume concentration of particles ranging from very fine silts to fine silt was observed in mud for 2009 (Table 4.5). A significant negative relationship between LOI and silt was found for mud in 2009. However, there was no relationship between LOI and volume concentration of clay, fine sand or sand sized particles for mud in 2009. LOI in 2007 was unrelated to the volume concentration of all particle size categories.

### 4.3.4 Calcium carbonate saturation index

Between October 2007 and March 2009, all sites in Lake Alexandrina had SI values greater than zero, indicating an over-saturation of calcium carbonate in the water column. The site closest to the River Murray had the lowest SI values (0.47 ± 0.18), followed by the site in the middle of Lake Alexandrina (0.71 ± 0.15) and the site furthest downstream (0.75 ± 0.17). pH levels remained above 8 for all water samples used in these calculations.
4.4. Discussion

4.4.1 Climate and lake water level declines

The drought in the Murray-Darling Basin that led to water shortages and the lake water level decline observed in this study was severe, spanning a decade from 2000 to 2010 (Leblanc et al. 2009). The changing patterns of rainfall and temperature that led to the drought were consistent with modelled forecasts of conditions under an enhanced greenhouse effect (Cai & Cowan 2008; Murphy & Timbal 2008; Timbal 2009). Future water availability in the basin is forecast to continue to decline over the coming decades (CSIRO 2008) increasing the likelihood of reduced lake water levels as observed in this study and in other regions of the Murray-Darling Basin (Baldwin et al. 2008). Furthermore, water level declines are forecast to become more prevalent in semi-arid regions around the world as extraction for human use, coupled with climatic changes, decrease the availability of surface water (Vörösmarty et al. 2000; Alcamo et al. 2007).

4.4.2 Changes to sediment composition and distribution during water level declines

Sediments of Lake Alexandrina grouped into two distinct categories of sand and mud based on water content. Low WC, low LOI and a correspondingly high bulk density characterised sandy sediments. Sand was also made up of a large volume of coarse particles and these sediments dominated the shallow, littoral region of Lake Alexandrina. Mud was characterised by significantly higher WC and LOI, with lower bulk density and finer particles. The mud dominated the large, flat profundal zone of the lake in deeper water. These characterisations are consistent with findings elsewhere with sediments in areas undergoing continuous erosion having low WC and LOI and those in areas where transient or permanent sediment deposition occurs
having high WC and LOI (Håkanson & Jansson 1983; Malmaeus & Håkanson 2003).

As water levels declined between 2007 and 2009, the range of WC in each sediment type increased. This change corresponded to an increase in LOI in sandy sediments, although the increase was only significant at $p = 0.09$, and was dominated by higher LOI in dry sand. For sand, the increased variance in 2009 extended either side of the 2007 range, indicating that the increased range did not simply reflect the drying of peripheral sites. Increases in the volume of fine particles in sandy sediments, coupled with the increased range of WC, suggests that sediment distribution has become more homogenous with the distinction between sand and mud in Lake Alexandrina becoming less distinct as water levels have fallen. These observations oppose expectations that fine sediments and LOI would decrease in concentration from peripheral waters as water levels decline and the wave-induced turbulent energy experienced at the sediment surface increases. The fluvial input of particles was 45.1 g m$^{-2}$ yr$^{-1}$ between 2007 and 2009 (Chapter 3), which equates to only a small fraction of the topmost centimetre of sediment sampled in this study (≤ 3% for mud, < 1% for sand). Therefore, the changes are indicative of an increased redistribution of finer, organic-rich sediments from profundal areas, coupled with an unchanged concentration of LOI.

In contrast to these findings, Gottgens (1994) found that organic material was eroded from littoral sediments during a period of low water level in Newmans Lake, Florida. Results presented here suggest that the fine sediments have increased in sandy sediments and that LOI has stayed the same or has increased slightly (Table 4.4). One possible explanation for this is that water level declines increase the wind-driven wave energy at the sediment surface, thereby enhancing the frequency and magnitude of sediment resuspension events (Effler & Matthews 2004). While this is expected to increase the rate of erosion of the finer sediment particles from shallow water, it would also increase the surface area of mud under deeper water that is
Changes to sediment composition and distribution

exposed to shear forces sufficient to resuspend sediments (Evans 1994). If water levels were reduced below some depth threshold, the redistribution of fine sediments, classified in this study as mud, would become more prevalent. The increasing variance in 2009 for both sediment types compared with 2007 and the increase in LOI throughout the lake support this hypothesis.

Regressions of TC, TN and TP with LOI suggested that nutrients in surface sediments were predominantly of an organic form following water level declines in 2009. This is in general agreement with Cook et al. (2009), who showed that Lake Alexandrina has historically transformed soluble, inorganic forms of phosphorus and nitrogen from fluvial input into organic nutrients. The conversion of inorganic phosphorus and nitrogen into their organic constituents was thought to have occurred by the assimilation and growth of phytoplankton, which subsequently settled onto the bottom substrate. When water flowed through the lakes, these organic nutrients would be exported from the lake in peak flow and replaced by fluvial inputs of inorganic nutrients (Cook et al. 2009). But, as fluvial inflows declined due to low flows during the recent drought period, Lake Alexandrina effectively became a closed basin, with no outflow across the barrages. Consequently, the organic form of phosphorus and nitrogen accumulated as demonstrated by the strong regressions between LOI and nutrients in 2009 but not in 2007. This assumption is further supported by the carbon to nitrogen (C:N) mass ratio of sediment in 2009. Mud had C:N ratios of 7-8, while wet and inundated sand had ratios of 4 – 6.5. By comparison, in 2007 ratios for sand were 15 – 16 and above 9 for mud. According to Meyers and Ishiwatari (1993) sediments with C:N ratios above 20 suggest that particulate organic matter therein is generally derived from vascular terrestrial plants, while ratios between 7 – 11 are indicative of aquatic plant remains and ratios of 4 – 10 indicate phytoplankton material. Given this, it appears that recently deposited or transported algal matter has become the predominant contributor to TC and TN in sediments sampled in 2009.
4.4.3 Loss of inorganic carbon in peripheral sediments

As water level declined in Lake Alexandrina, the nutrient concentrations in sand declined too. According to mass ratios of TC, TN and TP, declines were largest for carbon. These changes were more evident when sediments that were not inundated with lake water in 2009 were excluded from the analysis (Table 4.4). It appears that the TC loss was predominantly due to the loss of inorganic carbon because LOI increased while TC declined. Consequently, the decline in inorganic carbon from littoral sediments is most likely a result of the loss of carbonates.

A number of mechanisms may explain this reduction. During the water level decline in Lake Alexandrina, sulfidic sediments became exposed to the air and were oxidised (Simpson et al. 2008), which could have resulted in a decrease in pH and corresponding dissolution of calcium carbonate. However, this scenario is unlikely for several reasons. For a start, sediments that did acidify in Lake Alexandrina during the course of this study were limited to isolated fringing areas in Currency Creek, and the Finniss River, upstream of the lake (EPA 2009). In addition, the loss of inorganic carbon occurred in sediments that remained inundated and were thus not exposed to the atmosphere. Finally, calculations from concentrations of calcium and bicarbonate in water samples taken from three sites in the lake showed that the saturation index for calcium carbonate remained positive, indicating that the water column was over-saturated with respect to calcium carbonate and that precipitation of calcium carbonate was likely (Cole et al. 1994). pH also remained above 8 in the water column throughout the study period.

Other possibilities remain. Firstly, declining water levels have resulted in increased evaporative concentration of salts entering the lakes from the River Murray, smaller tributaries and through marine seepage from the barrages (Figure 4.2). Evaporative concentration of salts is likely to increase the concentration of magnesium in the water column. This has been shown experimentally to inhibit calcite precipitation
through surface disruption of the crystallisation process (Morse et al. 1997; Zhang & Dawe 2000) and a concomitant increase in the solubility of the resultant mineral (Möller & Parekh 1975). Similar disruptions to the crystal lattice formation of calcium carbonate from dissolved and particulate organic substances have been well established (Hoch et al. 2000). Secondly, oxic conditions of pore water in surficial sediments are enhanced by the increased wave energy of shallower waters (e.g. Webster 2003). Oxic pore waters enhance the decomposition of organic matter. Morse et al. (1985) have shown neritic carbonate dissolution from sediments below marine waters over-saturated in calcium carbonate as a result of pore water mineralisation of organic matter and the resultant increase in local pCO₂. Finally, Müller et al. (2003) used microsensors and ion-selective electrodes to demonstrate that calcite in surface sediments of a freshwater lake underwent dissolution that was driven by aerobic decomposition of organic carbon as the pH immediately below the sediment surface dropped. This observation occurred despite the SI index for calcite suggesting precipitation from the overlying water column (Müller et al. 2003). Assuming a 1:1 O₂:C respiration stoichiometry, oxygen demand of sandy sediments in Lake Alexandrina produces 23.7 mmol C m⁻² d⁻¹ as CO₂ from aerobic mineralisation of organic carbon (Dominic Skinner; unpublished data). The consequent decrease in localised pH could dissociate calcium carbonate compounds in the surface sediments in the same manner as described by Müller et al. (2003). Sediment traps showed that organic carbon was being deposited onto sandy sediments in excess of the rate of mineralisation by an order of magnitude (Chapter 5), suggesting that incoming organic carbon was not limiting mineralisation rates in the surface sediments, despite falling water levels.

This study has shown that declining water levels have altered the sediment distribution within Lake Alexandrina with consequences for the chemical composition of surface sediments. This study, like others (Furey et al. 2004), found the influence of water level decline to be most evident in littoral sediments.
However, unlike other studies (Gottgens 1994; James et al. 2001), an increase in LOI at deeper sites did not correspond with the erosion of LOI from shallow sites. Instead, LOI and particles with a nominal diameter below 63.3 μm increased or underwent no significant change in sandy sediment below shallow water. These observations appear to be caused by an increase in the resuspension and redistribution of fine, organic-rich particles from deeper water that became exposed to higher wave-induced turbulent energy as water levels declined or from autochthonous phytoplankton productivity. The transient settling of these particles below shallow water increased fine particles (4.43 – 19.9 μm diameter) and offset any increased erosion of less dense organic particles in sandy sediments. The substantial decrease in inorganic carbon that was observed appears to be due to loss of carbonates from the sediment. This was related to the increase of fine, organic-rich particles in sandy sediment, through a hypothesised pH decline and calcium carbonate dissolution resulting from localized oxidation and mineralisation of organic carbon.
Table 4.1: Comparison of means (± Standard error) from Tukey's HSD post hoc analysis of sediment type and year for parameters from the two-way ANOVA results in Table 4.2.¹

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>2007</th>
<th>2009</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Sand</td>
<td>Mud</td>
</tr>
<tr>
<td>Dry Weight</td>
<td>g.m⁻²</td>
<td>26547 ± 1275</td>
<td>4405 ± 1337</td>
</tr>
<tr>
<td>Loss-on-Ignition</td>
<td>%</td>
<td>0.4 ± 1.2</td>
<td>7.87 ± 1.3</td>
</tr>
<tr>
<td>Bulk Density</td>
<td>g.cm⁻³</td>
<td>2.0 ± 0.05</td>
<td>1.19 ± 0.06</td>
</tr>
<tr>
<td>Clay</td>
<td>µL.L⁻¹</td>
<td>0.1 ± 0.6</td>
<td>2.76 ± 0.7</td>
</tr>
<tr>
<td>Very Fine Silt</td>
<td>µL.L⁻¹</td>
<td>0.1 ± 0.5</td>
<td>2.12 ± 0.5</td>
</tr>
<tr>
<td>Fine Silt</td>
<td>µL.L⁻¹</td>
<td>0.8 ± 1.5</td>
<td>12.79 ± 1.6</td>
</tr>
<tr>
<td>Silt</td>
<td>µL.L⁻¹</td>
<td>2.6 ± 2.2</td>
<td>35.04 ± 2.3</td>
</tr>
<tr>
<td>Fine Sand</td>
<td>µL.L⁻¹</td>
<td>43.9 ± 4.2</td>
<td>33.63 ± 4.4</td>
</tr>
<tr>
<td>Sand</td>
<td>µL.L⁻¹</td>
<td>52.5 ± 5.6</td>
<td>13.67 ± 5.9</td>
</tr>
<tr>
<td>Total Phosphorus</td>
<td>g.m⁻²</td>
<td>2.4 ± 0.2</td>
<td>1.57 ± 0.2</td>
</tr>
<tr>
<td>Total Nitrogen</td>
<td>g.m⁻²</td>
<td>23.5 ± 1.3</td>
<td>15.16 ± 1.3</td>
</tr>
<tr>
<td>Total Carbon</td>
<td>g.m⁻²</td>
<td>303.2 ± 16.2</td>
<td>141.92 ± 17.0</td>
</tr>
<tr>
<td>Organic Carbon</td>
<td>g.m⁻²</td>
<td>55.8 ± 14.6</td>
<td>125.78 ± 15.3</td>
</tr>
<tr>
<td>Inorganic Carbon</td>
<td>g.m⁻²</td>
<td>247.4 ± 6.7</td>
<td>16.14 ± 7.0</td>
</tr>
</tbody>
</table>

¹ The same superscript letter denotes values for each parameter that are not significantly different (p < 0.05).
Table 4.2: Statistical results from two-way analysis of variance examining the influence of sediment type (sand and mud) and sampling year on sediment characteristics.

<table>
<thead>
<tr>
<th>Sediment characteristic</th>
<th>df $^1$</th>
<th>$r^2$</th>
<th>Sediment Type</th>
<th>$F$</th>
<th>$p$</th>
<th>Year</th>
<th>$F$</th>
<th>$p$</th>
<th>Interaction</th>
<th>$F$</th>
<th>$p$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dry weight</td>
<td>1,94</td>
<td>0.89</td>
<td>463.1</td>
<td>&lt;0.05</td>
<td>2.4</td>
<td>0.12</td>
<td>0.2</td>
<td>0.64</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Loss-on-ignition</td>
<td>1,94</td>
<td>0.66</td>
<td>93.5</td>
<td>&lt;0.05</td>
<td>6.9</td>
<td>&lt;0.05</td>
<td>4.7</td>
<td>&lt;0.05</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bulk density</td>
<td>1,94</td>
<td>0.87</td>
<td>422.5</td>
<td>&lt;0.05</td>
<td>0.02</td>
<td>0.90</td>
<td>1.1</td>
<td>0.30</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Clay</td>
<td>1,94</td>
<td>0.63</td>
<td>64.1</td>
<td>&lt;0.05</td>
<td>20.3</td>
<td>&lt;0.05</td>
<td>9.1</td>
<td>0.0032</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Very fine silt</td>
<td>1,94</td>
<td>0.89</td>
<td>202.5</td>
<td>&lt;0.05</td>
<td>147.6</td>
<td>&lt;0.05</td>
<td>85.3</td>
<td>&lt;0.05</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fine silt</td>
<td>1,94</td>
<td>0.90</td>
<td>289.7</td>
<td>&lt;0.05</td>
<td>142.8</td>
<td>&lt;0.05</td>
<td>57.5</td>
<td>&lt;0.05</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Silt</td>
<td>1,94</td>
<td>0.74</td>
<td>220.9</td>
<td>&lt;0.05</td>
<td>0.4</td>
<td>0.54</td>
<td>9.7</td>
<td>0.0025</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fine sand</td>
<td>1,94</td>
<td>0.42</td>
<td>22.9</td>
<td>&lt;0.05</td>
<td>17.2</td>
<td>&lt;0.05</td>
<td>3.4</td>
<td>0.069</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sand</td>
<td>1,94</td>
<td>0.58</td>
<td>81.3</td>
<td>&lt;0.05</td>
<td>2.3</td>
<td>0.14</td>
<td>0.4</td>
<td>0.52</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total phosphorus</td>
<td>1,94</td>
<td>0.53</td>
<td>0.8</td>
<td>0.38</td>
<td>28.5</td>
<td>&lt;0.05</td>
<td>38.2</td>
<td>&lt;0.05</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total nitrogen</td>
<td>1,94</td>
<td>0.69</td>
<td>2.3</td>
<td>0.13</td>
<td>34.3</td>
<td>&lt;0.05</td>
<td>89.1</td>
<td>&lt;0.05</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total carbon</td>
<td>1,94</td>
<td>0.67</td>
<td>8.5</td>
<td>&lt;0.05</td>
<td>80.3</td>
<td>&lt;0.05</td>
<td>83.6</td>
<td>&lt;0.05</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Organic carbon</td>
<td>1,94</td>
<td>0.42</td>
<td>40.7</td>
<td>&lt;0.05</td>
<td>1.0</td>
<td>0.32</td>
<td>0.3</td>
<td>0.58</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Inorganic carbon</td>
<td>1,94</td>
<td>0.93</td>
<td>442.4</td>
<td>&lt;0.05</td>
<td>574.2</td>
<td>&lt;0.05</td>
<td>442.1</td>
<td>&lt;0.05</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

$^1$ Two degrees of freedom (df) are reported. First, the degrees of freedom between groups ($n - 1$; where $n = 2$). Second, the degrees of freedom within a group, which is the number of sediment samples ($n - 1$; where $n = 95$).
Table 4.3: One-way analysis of variance of sediment particle size between sandy sediments in 2007 (n = 11) and those in 2009 divided into dry sediment (n = 13), wet sediment (n = 15) and inundated sediments (n = 15) according to the influence of lake water on sediments at the time of sampling\(^1\)

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Clay</td>
<td>(\mu L.L^{-1})</td>
<td>0.11</td>
<td>2.12</td>
<td>0.11</td>
<td>0.1 ± 0.5(^A)</td>
<td>1.1 ± 0.5(^A)</td>
<td>0.2 ± 0.4(^A)</td>
<td>1.4 ± 0.4(^A)</td>
</tr>
<tr>
<td>Very Fine Silt</td>
<td>(\mu L.L^{-1})</td>
<td>0.25</td>
<td>5.62</td>
<td>&lt;0.05</td>
<td>0.1 ± 0.4(^A)</td>
<td>1.3 ± 0.4(^AB)</td>
<td>0.5 ± 0.4(^A)</td>
<td>2.0 ± 0.4(^B)</td>
</tr>
<tr>
<td>Fine Silt</td>
<td>(\mu L.L^{-1})</td>
<td>0.27</td>
<td>6.05</td>
<td>&lt;0.05</td>
<td>0.8 ± 1.6(^A)</td>
<td>6.8 ± 1.5(^BC)</td>
<td>3.2 ± 1.4(^AB)</td>
<td>9.0 ± 1.4(^C)</td>
</tr>
<tr>
<td>Silt</td>
<td>(\mu L.L^{-1})</td>
<td>0.17</td>
<td>3.51</td>
<td>&lt;0.05</td>
<td>2.6 ± 2.3(^AB)</td>
<td>7.7 ± 2.1(^AB)</td>
<td>3.1 ± 2.0(^B)</td>
<td>10.6 ± 2.0(^A)</td>
</tr>
<tr>
<td>Fine Sand</td>
<td>(\mu L.L^{-1})</td>
<td>0.24</td>
<td>5.33</td>
<td>&lt;0.05</td>
<td>43.9 ± 4.8(^A)</td>
<td>47.4 ± 4.4(^A)</td>
<td>36.5 ± 4.1(^AB)</td>
<td>25.2 ± 4.1(^B)</td>
</tr>
<tr>
<td>Sand</td>
<td>(\mu L.L^{-1})</td>
<td>0.11</td>
<td>2.06</td>
<td>0.12</td>
<td>52.5 ± 7.1(^A)</td>
<td>35.6 ± 6.5(^A)</td>
<td>56.5 ± 6.1(^A)</td>
<td>51.7 ± 6.1(^A)</td>
</tr>
</tbody>
</table>

\(^1\) The same superscript letter denotes values for each parameter that are not significantly different (\(p < 0.05\)).
Table 4.4: One-way analysis of variance of biogeochemical parameters between sandy sediments in 2007 (n = 11) and those in 2009 divided into dry sediment (n = 13), wet sediment (n = 15) and inundated sediments (n = 15) according to the influence of lake water on sediments at the time of sampling.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>DW</td>
<td>g.m$^{-2}$</td>
<td>0.62</td>
<td>26.7</td>
<td>&lt;0.05</td>
<td>26547 ± 997$^A$</td>
<td>35255 ± 917$^B$</td>
<td>26782 ± 853$^A$</td>
<td>24841 ± 853$^A$</td>
</tr>
<tr>
<td>LOI</td>
<td>%</td>
<td>0.32</td>
<td>7.8</td>
<td>&lt;0.05</td>
<td>0.4 ± 0.2$^A$</td>
<td>1.5 ± 0.2$^B$</td>
<td>0.5 ± 0.2$^A$</td>
<td>0.7 ± 0.2$^A$</td>
</tr>
<tr>
<td>Density</td>
<td>g.cm$^{-3}$</td>
<td>0.58</td>
<td>23.1</td>
<td>&lt;0.05</td>
<td>2.0 ± 0.04$^A$</td>
<td>2.3 ± 0.04$^B$</td>
<td>1.9 ± 0.04$^A$</td>
<td>1.8 ± 0.04$^A$</td>
</tr>
<tr>
<td>TC</td>
<td>g.m$^{-2}$</td>
<td>0.80</td>
<td>64.6</td>
<td>&lt;0.05</td>
<td>303.2 ± 16.9$^A$</td>
<td>114.1 ± 15.0$^B$</td>
<td>39.3 ± 14.0$^C$</td>
<td>36.8 ± 14.0$^C$</td>
</tr>
<tr>
<td>OC</td>
<td>%</td>
<td>0.31</td>
<td>7.6</td>
<td>&lt;0.05</td>
<td>55.8 ± 29.0$^A$</td>
<td>114.1 ± 13.3$^B$</td>
<td>39.3 ± 12.4$^A$</td>
<td>36.7 ± 12.4$^A$</td>
</tr>
<tr>
<td>IC</td>
<td>g.m$^{-2}$</td>
<td>0.94</td>
<td>273.9</td>
<td>&lt;0.05</td>
<td>247.4 ± 7.7$^A$</td>
<td>0 ± 7.1$^B$</td>
<td>0 ± 6.6$^B$</td>
<td>0.1 ± 6.6$^B$</td>
</tr>
<tr>
<td>TN</td>
<td>g.m$^{-2}$</td>
<td>0.74</td>
<td>46.9</td>
<td>&lt;0.05</td>
<td>23.5 ± 1.4$^A$</td>
<td>9.3 ± 1.3$^B$</td>
<td>6.0 ± 1.1$^B$</td>
<td>7.2 ± 1.1$^B$</td>
</tr>
<tr>
<td>TP</td>
<td>g.m$^{-2}$</td>
<td>0.52</td>
<td>17.7</td>
<td>&lt;0.05</td>
<td>2.4 ± 0.2$^A$</td>
<td>0.6 ± 0.2$^B$</td>
<td>0.6 ± 0.2$^B$</td>
<td>0.7 ± 0.2$^B$</td>
</tr>
<tr>
<td>C:N:P</td>
<td></td>
<td>12.6 : 1</td>
<td>11.9 : 1</td>
<td>6.7 : 1</td>
<td>5.3 : 1</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

1 Values reported are mean ± standard error.
2 The same superscript letter denotes values for each parameter that are not significantly different (p < 0.05).
3 DW, Dry Weight; LOI, Loss-on-Ignition; Density, Bulk Density; TC, Total Carbon; OC, Organic Carbon; IC, Inorganic Carbon; TN, Total Nitrogen; TP, Total Phosphorus

Values reported are mean ± standard error.

The same superscript letter denotes values for each parameter that are not significantly different (p < 0.05).

DW, Dry Weight; LOI, Loss-on-Ignition; Density, Bulk Density; TC, Total Carbon; OC, Organic Carbon; IC, Inorganic Carbon; TN, Total Nitrogen; TP, Total Phosphorus
<table>
<thead>
<tr>
<th>LOI regressed with:</th>
<th>2007 (df 1,8)</th>
<th>2009 (df 1,28)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Slope</td>
<td>$r^2$</td>
</tr>
<tr>
<td>Clay</td>
<td>0.03</td>
<td>0.0029</td>
</tr>
<tr>
<td>Very Fine Silt</td>
<td>0.06</td>
<td>0.038</td>
</tr>
<tr>
<td>Fine Silt</td>
<td>0.3</td>
<td>0.080</td>
</tr>
<tr>
<td>Silt</td>
<td>-0.05</td>
<td>0.00059</td>
</tr>
<tr>
<td>Fine Sand</td>
<td>-0.8</td>
<td>0.12</td>
</tr>
<tr>
<td>Sand</td>
<td>0.4</td>
<td>0.065</td>
</tr>
<tr>
<td>TC</td>
<td>-6.3</td>
<td>0.20</td>
</tr>
<tr>
<td>TN</td>
<td>-0.4</td>
<td>0.32</td>
</tr>
<tr>
<td>TP</td>
<td>-0.05</td>
<td>0.14</td>
</tr>
</tbody>
</table>
Figure 4.1: Sediment sampling sites in Lake Alexandrina. Those visited in both 2007 and 2009 (●), and those visited in 2009 only (○) are indicated. Water sampling sites used to calculate the saturation index of calcium carbonate are also indicated (x). Locations of the five barrages separating Lake Alexandrina from the Coorong are shown by thick black bars.
Figure 4.2: Sediment sampling dates relative to water level drawdown and increasing salinity in Lake Alexandrina. Changes in water level (—) and salinity (- - -) in Lake Alexandrina from early 2007 to April 2010 are shown, as are the dates when water samples were collected for use in calculating the calcium carbonate saturation index (Δ). Vertical dashed lines indicate dates of sediment sampling in February of 2007 and 2009.
Figure 4.3: Relationship of water depth with water content (A) and organic matter as loss-on-ignition (B) of surface sediments in Lake Alexandria. Linear regression results of the overall relationship, and that for sediments below shallow (<1.4 m depth) and deep (>1.4 m depth) water are shown.

**Linear relationships**

Overall: Slope $29.9 \pm 1.7$; $r^2 = 0.81$; $p < 0.0001$

<1.4 m depth: Slope $12.9 \pm 2.7$; $r^2 = 0.36$; $p < 0.0001$

>1.4 m depth: Slope $4.8 \pm 3.9$; $r^2 = 0.05$; $p = 0.23$

Overall: Slope $5.9 \pm 0.6$; $r^2 = 0.57$; $p < 0.0001$

<1.4 m depth: Slope $-0.1 \pm 0.3$; $r^2 = 0.003$; $p = 0.72$

>1.4 m depth: Slope $4.0 \pm 2.7$; $r^2 = 0.07$; $p = 0.15$
Figure 4.4: Frequency distribution of sediment water content in 2007 (—) and in 2009 (- - -). Two distinct groups of sediment type, sand and mud, are indicated with bars above plot.
Figure 4.5: Comparison between the mean particle size distribution in 2007 (circles) and 2009 (squares) of mud (closed symbols) and sand (open symbols) in Lake Alexandrina. Error bars represent one standard deviation.
Changes to sediment composition and distribution

Figure 4.6: Relationship between total carbon (TC) and loss-on-ignition (LOI) for sand and mud in 2007 and in 2009. Neither regression in 2007 was significant, whereas both in 2009 were.
5. Fine sediment dynamics in a large, shallow lake following water level decline

5.1 Introduction

Resuspension of sediment into the water column is a widely studied phenomena with an important role in lake ecosystem dynamics (Evans 1994; Bloesch 1995). In shallow lakes, sediment resuspension is a dominant driver of sediment distribution (Hilton 1985; Douglas & Rippey 2000) and light availability (Schallenberg & Burns 2004; Lawson et al. 2007); it influences dissolved organic carbon release from particulate organic matter (Koelmans & Prevo 2003); and, has been shown to increase both total phosphorus (Kristensen et al. 1992) and soluble phosphorus concentrations (Sondergaard et al. 1992). Resuspension-induced light limitation has been shown to reduce productivity in shallow lakes (Hellström 1991), especially after water level declines (Lind et al. 1994). However, nutrient desorption from resuspended sediment can also enhance productivity (Reddy et al. 1996), particularly when pH is high during a phytoplankton bloom (Holmroos et al. 2009). Additionally, meroplankton (algae resuspended from the sediment) can contribute towards primary productivity during resuspension events (Schallenberg & Burns 2004).
Sediment resuspension occurs through wave action, bioturbation, water currents at the sediment surface and internal seiching (Bengtsson et al. 1990; Gloor et al. 1994). However, in shallow lakes, many studies suggest that wind-induced wave action is the dominant process affecting particle resuspension. Waves induce turbulence at the sediment surface, which generates the shear stress that can overcome the gravitational and cohesive forces holding bottom sediments in place (Gons et al. 1986; Bengtsson et al. 1990; Luettich Jr. et al. 1990; Chung et al. 2009).

The type of sediment being resuspended will influence its residence time in the water column (Håkanson & Jansson 1983; Evans 1994) and the effect on light availability through different optical properties (Kozerski 1994). The interaction between rates of sedimentation and the frequency of resuspension events is likely to be a strong determinant of sediment distribution and the long-term light climate.

Hilton (1985) describes a qualitative framework to estimate the type of sediment distribution in lakes with stable water levels. The framework was based upon water depth and frequency of mixing and has good empirical support (Davies-Colley & Smith 2001). For example, in large polymictic lakes, Hilton (1985) suggests that sediment focusing occurs through peripheral wave attack. This involves the preferential erosion of fine sediments below shallow water that are transported and accumulate in the deeper sections of the lake where wave energy is relatively low. Aldridge et al. (2009) suggested this mechanism of sediment focusing was occurring in Lake Alexandrina prior to water level drawdown. When water levels fluctuate, however, sediment distribution can be altered as the area and type of sediment exposed to resuspension events is altered (Douglas & Rippey 2000). Under lowered water levels, fine particles that had accumulated in deeper sites can become exposed to more frequent shear stress of sufficient magnitude to resuspend and redistribute particles. Under these conditions, the type of sediment distribution on the framework proposed by Hilton (1985) can shift to random redistribution. This occurs when no...
sediment focusing is present, wave energy is imparted across the entire lake bed and sediment resuspension occurs in both deep and shallow regions of a lake.

A range of methods have been used to approximate the amount of suspended particulate matter in the water column that is sourced from resuspension of bottom sediments. These include mass balances (Lind et al. 1994; Effler & Matthews 2004; Håkanson 2005; Punning et al. 2006); various modeling approaches of wave energy and in situ measurement of suspended particulate matter and turbidity (Dillon et al. 1990); and, high frequency measurements using optical (Hamilton & Mitchell 1996; Bailey & Hamilton 1997; Cózar et al. 2005) or acoustic (Pierson & Weyhenmeyer 1994) instruments. However, the sediment trap, which measures the downward flux of particles through the water column integrated over the duration of a deployment, is the most commonly used tool for estimating sedimentation and resuspension (Chung et al. 2009). Sediment traps can link the physical conditions that resuspend sediments, with the characteristics of sediments being resuspended.

Chapter 3 provided evidence that fine sediments in Lake Alexandrina were being distributed onto shallow, littoral sands, based on a survey of sediment characteristics. The present chapter describes an experiment using sediment traps to determine the extent to which sediments are resuspended in Lake Alexandrina, the type of sediment that is resuspended and the redistribution of these sediments throughout the lake. It aims to test the hypothesis that fine sediments are being increasingly redistributed into shallower water as a result of water level decline. This information is used to discuss the likely effects of hydrological regulation on the sediment accumulation and resuspension in Lake Alexandrina.
5.2 Methods

5.2.1 Site description

The general characteristics of Lake Alexandrina have been described previously (section 2.2) but salient points will be highlighted briefly. The River Murray is heavily regulated with a series of locks and weirs altering the timing and reducing the magnitude of flow. The altered hydrology has increased the deposition of coarse particles upstream of regulatory structures (Gasith 1975; Kristensen et al. 1992; Håkanson 1995; Douglas & Rippey 2000; Niemistö et al. 2008; Ostrovsky & Yacobi 2010) and increased the proportion of fine sediments entering the lake (Thoms & Walker 1993). Additionally, changing riparian land-use has increased allochthonous, nutrient-rich sediment input from non-point sources (Bourman & Barnett 1995). Five barrages separate Lake Alexandrina from the estuarine Coorong and these reduce the flushing of fine sediments out of Lake Alexandrina. Static water levels since barrage construction have increased shoreline erosion (Herczeg et al. 2001), which along with lower peak flows has increased the rate of sediment accumulation in Lake Alexandrina (Butler & Woodard 1993). Severe reductions in river flow, resulting from over-extraction, drought and climatic changes (Barnett 1994; Bourman & Barnett 1995; Herczeg et al. 2001), prevented any large outflow events from Lake Alexandrina between 2001 and mid-2010, and led to the extreme water level declines described previously (Chapter 2). During the period of sediment trap deployment the mean depth of the lake was 1.3 m.

5.2.2 Sediment traps

Sediment traps were designed according to the specifications of Bloesch and Burns (Murphy & Timbal 2008; Leblanc et al. 2009; Timbal 2009) and Bloesch (1980) for turbulent waters. Cylindrical PVC tubes with an aspect ratio (height/diameter) of 10 and an internal diameter of 5 cm were suspended on an anchored rope, 70 cm above
the sediment surface with a subsurface buoy to maintain vertical alignment and a surface buoy for retrieval (1994). Single traps were deployed at 8 sites in Lake Alexandrina from 27 October 2009 (Figure 5.1). Traps were retrieved at approximately three week intervals through to 10 February 2010 and weekly thereafter until 24 March 2010. Loss of traps reduced the number of samples retrieved. In February 2010, an additional two traps were deployed at four sites to give triplicate samples for the remainder of the deployments (Figure 5.1). These sites were chosen to represent the two distinct sediment types present in Lake Alexandrina, coarse sand (<50% water content) and fine mud (>50% water content, Chapter 4). Samples from South and East central were representative of coarse, sandy sediments, while samples from South central and Central represented fine, muddy sediments. Upon retrieval, trap contents were transferred to polyethylene bottles, sealed and kept in the dark at 4°C. Traps were washed thoroughly and redeployed. Given the shallow depth of water (≤ 2 m), trap anchors could be placed gently on the sediment surface, preventing sediment from being disturbed during redeployment.

At the beginning and end of every deployment period, depth-integrated water samples were collected with a tube sampler according to methods outlined in Chapter 3. Surface sediment samples were retrieved using a corer pushed directly into the sediment surface and the topmost centimetre of sediment was extruded and homogenised according to methods outlined in Chapter 4.

Depth integrated water samples were analysed for suspended particulate matter (SPM) gravimetrically following filtration through pre-combusted Whatman GF/C filter paper and drying at 60°C (Eaton et al. 2005). The filter papers were then combusted in a muffle-furnace at 550°C for 1 hour to determine suspended particulate inorganic matter (SPIM) and suspended particulate organic matter (SPOM, Eaton et al. 2005). The same procedure was conducted for samples collected from sediment traps for determination of trapped particulate matter (TPM),
trapped inorganic particulate matter (TIPM) and trapped organic particulate matter (TOPM). The dry weight and loss-on-ignition (LOI) of surface sediments was determined in the same manner as trap contents but without any filtration. The proportion of TPM attributed to sediment resuspension was calculated by subtracting the long-term net accumulation rate of 1.7 – 9.2 g m$^{-2}$ d$^{-1}$ (Håkanson et al. 1989) from the sedimentation rate, which was TPM converted to an areal deposition rate (see Appendix 1). The extremely low water levels, coupled with decreased SPM input from the River Murray (Chapter 3), suggests that this net accumulation rate is likely to be an overestimate of actual rates during this study.

Particle size distributions (PSD) of water samples, sediment trap contents and surface sediments were determined with laser transmissometry using a LISST-100 (Sequoia Scientific, Washington USA) according to methods outlined in Agrawal et al. (Barnett 1994; Herczeg et al. 2001). PSD was reported as the diameter, in µm, below which 10% ($D_{10}$), 50% ($D_{50}$), 60% ($D_{60}$) and 90% ($D_{90}$) of particles by volume were present in a given sample. In addition, mean particle size (MPS) and the Hazen-Uniformity coefficient, equal to $D_{60}/D_{10}$, are reported. The Hazen-Uniformity coefficient is a measure of the homogeneity of the particle size distribution.

Chlorophyll $a$ (chl$a$) concentrations of water, sediment trap contents and surface sediments were extracted over 24 hours with methanol (99.8%) following filtration of water and trap samples (as for SPM) and for a known mass of surface sediment. Chl$a$ was determined spectrophotometrically at 665 nm and 700 nm according to Golterman et al. (2008).

The total carbon (TC) and total nitrogen (TN) content of surface sediment and sediment trap contents were determined for the two final deployment periods (Table 5.1) at the four sites with triplicate sediment traps (Central, South central, East central & South). TC and TN were determined by combustion of dried sample to 1300°C with a LECO CNS analyzer (Environmental Analysis Laboratory, Southern
Cross University, New South Wales, National Association of Testing Authorities endorsed). Organic carbon (OC) was estimated as half of the measured LOI for sediments or TPM for trap contents (1978), unless this was greater than TC, in which case all carbon was assumed to be in the organic form.

Where replicate samples were taken, a repeated measures ANOVA, with Bonferroni correction (Horppila & Nurminen 2005), was used to determine whether the amount and type of sedimentation occurring over the two sediment types present in Lake Alexandrina (Chapter 4) varied significantly. This repeated measures ANOVA was applied to all data from the 4 sites where triplicate sediment traps were deployed. To assess the presence or extent of sediment focusing in Lake Alexandrina during low water levels, gross sedimentation rates at sites with replicate traps were standardized to the water depth above the trap opening to enable comparisons between sites through time. This involved the division of the sedimentation rate by the water depth above the trap opening. For all other analysis, except the repeated measures ANOVA, data from sediment traps at all sites were used without depth correction. An $\alpha$ value of 0.05 was deemed significant for all statistical tests.

### 5.2.3 Calculating the depth of the wave-mixed layer

Wave-induced sediment resuspension is considered to occur when wave energy reaches the bottom sediments and overcomes a critical shear stress (Nelson & Sommers 1982). This is estimated to occur in deep water when the wavelength is greater than twice the water depth according to linear wave theory (Evans 1994). For shallow water, this method will be an overestimate, but it is useful here to allow comparison with other shallow systems that have used this method (Douglas & Rippey 2000) for shallow lakes. Thus, the depth of the wave-mixed layer ($D_{WML}$) can be calculated from the wavelength ($L$), as

$$D_{WML} = 0.5L$$ (5.1)
For specific wind conditions at each site, $L$ can be calculated according to equation 5.2 from Carper and Bachman (Smith & Sinclair 1972; Wetzel 2001):

$$
L = \frac{gT^2}{2\pi}
$$

(5.2)

where $g$ is gravitational acceleration and $T$ is the wave period calculated using equation 5.3:

$$
\frac{gT}{2\pi W} = 1.2 \tanh \left[ 0.077 \left( \frac{gF}{W^2} \right)^{0.25} \right]
$$

(5.3)

Here, $W$ is the wind speed and $F$ is the effective fetch. $F$, which was calculated using equation 5.4, is the distance from a site to the shore ($x_i$) in the direction ($\lambda_i$) of the prevailing wind and $\pm 45^\circ$ on each side:

$$
F = \frac{\sum x_i \cos \lambda_i}{\sum \cos \lambda_i}
$$

(5.4)

The distance from each site to shore was integrated every $5^\circ$ for use in equation 5.4. Wind data was taken from an automated weather monitoring station operated by the South Australian Department of Water, Land and Biodiversity Conservation (DWLBC), which recorded average wind speed and direction every 15 minutes at East central (Figure 5.1). Mean $D_{WM}$ was calculated for each site and each deployment period from this wind data. Nephelometric turbidity data was provided by DWLBC for a site 2.5 km southeast of South central with a 15-minute resolution from 4 to 24 March 2010 (Figure 5.1).
5.3 Results

5.3.1 Rate of sedimentation and resuspension

Over the duration of this study, sedimentation rates varied between 266 g m$^{-2}$ d$^{-1}$ and 2351 g m$^{-2}$ d$^{-1}$. The minimum sedimentation rate was recorded at East central between 29 December and 15 January 2010, while the maximum sedimentation rate occurred at North between 24 November and 18 December 2009 (Table 5.1). This maximum sedimentation rate corresponded with the highest mean wind speed of 6.9 m s$^{-1}$, which was dominated by winds from the south. There were no significant differences in mean sedimentation rates between sites ($p = 0.15$), suggesting that sedimentation was on average spatially homogenous. However, sedimentation averaged across all sites varied significantly between deployment periods ($p < 0.05$) as a result of different wind conditions. Sedimentation rates measured at each site except East showed significant exponential relationships with mean D$_{WML}$ (Figure 5.2). The strongest correlations were found at East central ($r^2 = 0.84, p < 0.05$) and Central ($r^2 = 0.81, p < 0.05$), with the weakest correlations at East ($r^2 = 0.17, p = 0.22$) and North central ($r^2 = 0.47, p < 0.05$). Sedimentation calculated at the point that D$_{WML}$ equaled zero using these exponential relationships ranged from 56.0 g m$^{-2}$ d$^{-1}$ at Confluence to 459.8 g m$^{-2}$ d$^{-1}$ at East. However, because sedimentation rates measured in this study were significantly higher than the long-term accumulation rate of 1.7 – 9.2 g m$^{-2}$ d$^{-1}$, sediment resuspension was considered to contribute over 98% of the particles caught in traps.

Repeated measures ANOVA revealed that sedimentation rates were significantly lower at sites in the littoral region of the lake where sediments were coarse and sandy (South and East central) than at sites in the profundal region of the lake where sediments were fine and muddy (South central and Central; Table 5.2). However, when sedimentation rates were corrected for the depth at the trap site, there were no differences ($p = 0.69$).
5.3.2 Type of sediments undergoing resuspension

Particles caught in traps at all sites showed a similar size distribution (Figure 5.3), with MPS for trapped particles only significantly higher at North (30 ± 4 µm) than at East (24 ± 1 µm), South central (23 ± 2 µm) and South (24 ± 1 µm). This size distribution for trapped particles was similar to profundal muddy sediments (Central, South central, North central), whereas sediment particles at littoral sandy sediments (South, Confluence, East central) was skewed towards larger particles compared to material caught in traps (Figure 5.3).

Repeated measures ANOVA also showed that differences in sediment composition did not necessarily result in differences to trapped material. For example, chlα, LOI, and all reported particle size parameters (D_{10}, D_{50}, D_{90}, MPS and the Hazen-uniformity coefficient) were significantly different between sediment types. In contrast, the same parameters were not significantly different in trapped material over both sediment types (Table 5.2).

5.3.3 Relationship between $D_{WML}$ and turbidity

Turbidity was strongly correlated with $D_{WML}$ even during significantly different wind conditions (Figure 5.4). Between 4 and 10 March 2010, mean wind speed, based on data already averaged over 15 minute periods, (± 95% C.I.) was 6.62 ± 0.23 m s$^{-1}$. The maximum $D_{WML}$ at South central during this time was 17.4 m, which corresponded with a maximum turbidity of 615.1 NTU during the afternoon of 8 March 2010. Between 20 and 24 March 2010, mean wind speed was 4.19 ± 0.28 m s$^{-1}$ and $D_{WML}$ and turbidity were significantly lower than between 4 and 10 March 2010 (Figure 5.4). For both trap deployment periods, turbidity correlated strongly with $D_{WML}$ with $r^2 = 0.47$, $p < 0.05$ between 4 and 10 March 2010, and $r^2 = 0.50$, $p < 0.05$ between 20 and 24 March 2010.
5.3.4 Rate of nutrient deposition

Surface sediments of littoral sandy sites (South and East central) had significantly lower concentrations of TC ($F = 34.5, p < 0.05$) and TN ($F = 36.0, p < 0.05$) than profundal muddy sites (Central and South central) over the last two deployment periods where nutrients were analysed. For these samples, TC values were below half the recorded LOI and TOPM, suggesting that OC was the dominant form of carbon in sediment samples and traps. The rate of OC and TN sedimentation varied between sites (Table 5.3) but no clear pattern was evident. The mean sedimentation rate of OC was 31.1 g m$^{-2}$ d$^{-1}$ when averaged over all sites, while for TN it was 3.1 g m$^{-2}$ d$^{-1}$.

5.4 Discussion

5.4.1 Quantifying sediment resuspension in Lake Alexandrina

Sediment resuspension in Lake Alexandrina was very high at all sites and was the dominant source of sedimenting particles. Sedimentation recorded in this study ranged between 266 g m$^{-2}$ d$^{-1}$ and 2351 g m$^{-2}$ d$^{-1}$ with temporal variation strongly correlated to $D_{WML}$ at most sites (Figure 5.2). As there was no significant difference in sedimentation rates between sites or sediment types, the sediment redistribution in Lake Alexandrina can be categorised as random redistribution according to the framework outlined by Hilton (1984). According to Hilton (1985), random redistribution of sediments can be demonstrated by plotting water depth with sedimentation rate. If the resultant graph is a scattergram, with no obvious trends in sedimentation with depth, then sediment redistribution is expected to be random throughout the lake during resuspension events. Indeed, the data presented in this study conforms to such a scattergram (Figure 5.5), confirming the random redistribution of sediments.
The sedimentation rate of caught particles in Lake Alexandrina was significantly higher than other studies of shallow lakes. The maximum sedimentation rate recorded in Lough Neagh, Northern Ireland was 233.9 g m\(^{-2}\) d\(^{-1}\), at 2 m above the sediment surface (1985). However, Lough Neagh is deeper (mean depth 8.9 m) and smaller (area 383 km\(^2\)) than Lake Alexandrina and sedimentation was recorded further off the sediment surface. Lake Võrtsjärv, in Southern Estonia, has a depth similar to that of Lake Alexandrina before water levels were drawn down (mean depth 2.8 m), but is about half its size (area 270 km\(^2\)). Nõges et al. (Douglas & Rippey 2000) recorded maximum sedimentation rates of 700 g m\(^{-2}\) d\(^{-1}\) in Lake Võrtsjärv with traps exposed 1 m above the sediment surface. The shallow (mean depth 1.5 m), small (area 35 km\(^2\)) Lake Tämnaren had a maximum sedimentation rate of 157 g m\(^{-2}\) d\(^{-1}\) recorded in sediment traps by Bengtsson et al. (1999). Likely reasons for the significantly higher maximum rate of sedimentation are fourfold. Lake Alexandrina is larger and shallower than all lakes presented, with a larger effective fetch and corresponding larger transfer of wind energy onto the sediment surface. The D\(_{WML}\), therefore, will reach the sediment surface with higher frequency and higher force, resuspending larger quantities of surface sediment. Additionally, the height of trap exposure was closer to the sediment surface (0.7 m) than in both Lough Neagh and Lake Võrtsjärv (the height of sediment traps above the sediment surface was not recorded for Lake Tämnaren). Lake Alexandrina is filled via a regulated river, which delivers particle loads dominated by fine sediments that are easily resuspended. Finally, low water levels during this study suggest that sediments prone to resuspension are those that were previously protected by deeper waters. It has been shown that the grain size of sediments below deeper water in Lake Alexandrina is significantly smaller than in shallow water (Chapter 4), which suggests an increased propensity for resuspension.
5.4.2 Dynamics of sediment resuspension in Lake Alexandrina

In Lake Alexandrina, sediments are present in two major categories according to water depth: littoral sediments, which are dominated by coarse particles of low water and organic content, and profundal sediments, fine sediments that have high water retention and organic content (Chapter 3). The particle size of trapped particles was uniform across the lake, suggesting spatial homogeneity in resuspended sediments. This confirms the hypothesis in Chapter 3 that sediments are undergoing a dynamic redistribution as they are transported throughout the lake during resuspension events. Furthermore, given a deposition rate of OC onto sandy sediments of 27 ± 4 g m\(^{-2}\) d\(^{-1}\) (Table 5.3), a relatively high volume of OC is supplied frequently to littoral sediments. This adds some support to the hypothesis in Chapter 4 that inorganic carbon is being lost from sandy sediments as a result of localised mineralisation leading to pH declines and the dissolution of calcium carbonate.

Trap contents had a mean particle size of 26 ± 3 µm (Figure 5.3). According to Stokes law (1990), with a particle density of 1.25 g cm\(^{-3}\) for profundal sediments (chapter 4), this gives a settling velocity of 0.3 m hr\(^{-1}\). For the mean depth of Lake Alexandrina (1.3 m), the average resuspended particle will remain in suspension for up to 4.3 hours after the onset of quiescent conditions.

To get a rough approximation of the frequency of resuspension events, we can estimate the concentration of SPM required to give the recorded sedimentation rates in a single resuspension event (Massey 1979). These calculations rely on a number of assumptions. First, that sedimentation is described with first order kinetics and second, that there is no primary sedimentation during the period of trap deployment. The sedimentation rate is taken to be 0.3 m hr\(^{-1}\) for particles 26 µm in diameter, and the water depth is 1.3 m. Under these conditions, SPM would need to reach 2816 mg L\(^{-1}\) to achieve the minimum recorded resuspension of 257.7 – 265.2 g m\(^{-2}\) d\(^{-1}\). Similarly, a SPM concentration of 25259 mg L\(^{-1}\) would be required to achieve the
maximum recorded resuspension rate of 2341.7 – 2349.2 g m$^{-2}$ d$^{-1}$. Of course, these sedimenting solids would not all be resuspended in a single event. Supposing that a single resuspension event occurs every day, then the concentration of SPM required to achieve the observed resuspension falls to a range of 166 mg L$^{-1}$ to 1052 mg L$^{-1}$. The maximum recorded concentration of SPM in Lake Alexandrina between August 2008 and December 2009 was 587 mg L$^{-1}$ (Chapter 3; although these samples were limited to conditions that boat access onto the lake was possible, so maximum values may be considerable higher). This suggests that, as a rough approximation, resuspension events in Lake Alexandrina occur on a daily cycle. Indeed, even during low wind speeds, turbidity correlated with the D$_{WML}$, both of which showed some diel variation during the course of this study (Figure 5.4).

The results presented in this study suggest that: wind-induced sediment resuspension is a dominant process in Lake Alexandrina; resuspended sediments are dominated by fine particles; resuspension events occur with high frequency; and resuspended sediments are distributed homogenously throughout the lake. There is considerable evidence that flow regulation has influenced sediment loads and sediment types brought into the lakes from the River Murray (Close 1990; Thoms & Walker 1993; Barnett 1994; Herczeg et al. 2001). These changes to the accumulation of fine sediments may well have exacerbated the frequency and extent of resuspension events in Lake Alexandrina.
Table 5.1: Sedimentation rate for each trap site and each deployment period. A ‘—’ indicates trap was not retrieved.

<table>
<thead>
<tr>
<th>Deployment period</th>
<th>North</th>
<th>North central</th>
<th>Central</th>
<th>South central</th>
<th>South</th>
<th>East central</th>
<th>East</th>
<th>Confluence</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>g m$^{-2}$ d$^{-1}$</td>
<td>g m$^{-2}$ d$^{-1}$</td>
<td>g m$^{-2}$ d$^{-1}$</td>
<td>g m$^{-2}$ d$^{-1}$</td>
<td>g m$^{-2}$ d$^{-1}$</td>
<td>g m$^{-2}$ d$^{-1}$</td>
<td>g m$^{-2}$ d$^{-1}$</td>
<td>Mean Sedimentation</td>
</tr>
<tr>
<td>27 Oct – 13 Nov</td>
<td>—</td>
<td>1119</td>
<td>—</td>
<td>813</td>
<td>—</td>
<td>604</td>
<td>1137</td>
<td>918</td>
</tr>
<tr>
<td>13 Nov – 24 Nov</td>
<td>—</td>
<td>1198</td>
<td>—</td>
<td>1291</td>
<td>—</td>
<td>688</td>
<td>505</td>
<td>920</td>
</tr>
<tr>
<td>24 Nov – 18 Dec</td>
<td>2351</td>
<td>1248</td>
<td>1737</td>
<td>1969</td>
<td>—</td>
<td>1123</td>
<td>1590</td>
<td>1670</td>
</tr>
<tr>
<td>18 Dec – 29 Dec</td>
<td>—</td>
<td>1197</td>
<td>963</td>
<td>734</td>
<td>551</td>
<td>1108</td>
<td>1108</td>
<td>933</td>
</tr>
<tr>
<td>29 Dec – 15 Jan</td>
<td>1165</td>
<td>954</td>
<td>1103</td>
<td>865</td>
<td>266</td>
<td>1886</td>
<td>717</td>
<td>1000</td>
</tr>
<tr>
<td>15 Jan – 10 Feb</td>
<td>1171</td>
<td>995</td>
<td>975</td>
<td>1172</td>
<td>546</td>
<td>1390</td>
<td>881</td>
<td>1001</td>
</tr>
<tr>
<td>10 Feb – 4 Mar</td>
<td>1061</td>
<td>808</td>
<td>1123</td>
<td>1439</td>
<td>1169</td>
<td>885</td>
<td>855</td>
<td>647</td>
</tr>
<tr>
<td>4 Mar – 10 Mar</td>
<td>1425</td>
<td>1175</td>
<td>1882</td>
<td>1937</td>
<td>1509</td>
<td>1405</td>
<td>1181</td>
<td>2117</td>
</tr>
<tr>
<td>10 Mar – 20 Mar</td>
<td>453</td>
<td>451</td>
<td>552</td>
<td>629</td>
<td>639</td>
<td>352</td>
<td>455</td>
<td>354</td>
</tr>
<tr>
<td>20 Mar – 24 Mar</td>
<td>775</td>
<td>708</td>
<td>872</td>
<td>951</td>
<td>892</td>
<td>525</td>
<td>891</td>
<td>397</td>
</tr>
<tr>
<td>Mean ± SD</td>
<td>1200 ± 597</td>
<td>985 ± 260</td>
<td>1150 ± 445</td>
<td>1106 ± 440</td>
<td>1105 ± 408</td>
<td>647 ± 387</td>
<td>1018 ±419</td>
<td>945 ± 562</td>
</tr>
</tbody>
</table>
Table 5.2: Repeated measures ANOVA with Bonferroni correction comparing parameters at trap sites grouped by sediment type (littoral sand sediments at South/East central and profundal mud sediments at Central/South central).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Sediment Classification</th>
<th>df</th>
<th>F</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sedimentation*</td>
<td>1</td>
<td>1.31</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>Dep-Cor-Sed</td>
<td>1</td>
<td>0.02</td>
<td>0.69</td>
<td></td>
</tr>
<tr>
<td>DWML</td>
<td>1</td>
<td>0.41</td>
<td>0.07</td>
<td></td>
</tr>
<tr>
<td>Chlα Sediment</td>
<td>1</td>
<td>3.32</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>Chlα Trap</td>
<td>1</td>
<td>0.41</td>
<td>0.07</td>
<td></td>
</tr>
<tr>
<td>Chlα Water</td>
<td>1</td>
<td>0.00068</td>
<td>0.9</td>
<td></td>
</tr>
<tr>
<td>SPM</td>
<td>1</td>
<td>0.45</td>
<td>0.06</td>
<td></td>
</tr>
<tr>
<td>LOI Sediment</td>
<td>1</td>
<td>13.97</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>TOPM Trap</td>
<td>1</td>
<td>0.12</td>
<td>0.3</td>
<td></td>
</tr>
<tr>
<td>SOPM Water</td>
<td>1</td>
<td>0.39</td>
<td>0.08</td>
<td></td>
</tr>
<tr>
<td>(D_{10}) Sediment</td>
<td>1</td>
<td>0.88</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>(D_{10}) Trap</td>
<td>1</td>
<td>0.27</td>
<td>0.1</td>
<td></td>
</tr>
<tr>
<td>(D_{10}) Water</td>
<td>1</td>
<td>12.90</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>(D_{50}) Sediment</td>
<td>1</td>
<td>10.68</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>(D_{50}) Trap</td>
<td>1</td>
<td>0.0017</td>
<td>0.9</td>
<td></td>
</tr>
<tr>
<td>(D_{50}) Water</td>
<td>1</td>
<td>1.80</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>(D_{90}) Sediment</td>
<td>1</td>
<td>16.86</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>(D_{90}) Trap</td>
<td>1</td>
<td>0.20</td>
<td>0.2</td>
<td></td>
</tr>
<tr>
<td>(D_{90}) Water</td>
<td>1</td>
<td>0.31</td>
<td>0.1</td>
<td></td>
</tr>
<tr>
<td>MPS Sediment</td>
<td>1</td>
<td>16.73</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>MPS Trap</td>
<td>1</td>
<td>0.0028</td>
<td>0.87</td>
<td></td>
</tr>
<tr>
<td>MPS Water</td>
<td>1</td>
<td>1.32</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>HUC Sediment</td>
<td>1</td>
<td>4.02</td>
<td>&lt;0.025*</td>
<td></td>
</tr>
<tr>
<td>HUC Trap</td>
<td>1</td>
<td>0.64</td>
<td>0.03</td>
<td></td>
</tr>
<tr>
<td>HUC Water</td>
<td>1</td>
<td>0.64</td>
<td>0.03</td>
<td></td>
</tr>
</tbody>
</table>

1 Sedimentation: sedimentation rate (g m\(^{-2}\) d\(^{-1}\)); Dep-Cor-Sed: depth corrected sedimentation rate (g m\(^{-3}\) d\(^{-1}\)); DWML: depth of wave-mixed layer; Chlα: chlorophyll a; LOI: organic matter as loss-on-ignition; TOPM: trapped organic particulate matter; SOPM: suspended organic particulate matter; SPM: suspended particulate matter; \(D_{10, 50, 90}\): particle diameter which 10%, 50% and 90% of particulate matter by mass is below; MPS: mean particle size; HUC: Hazen-uniformity coefficient

* indicates a significant difference between sites where sediment is classified as either sand and mud following Bonferroni correction.
Table 5.3: Sedimentation rates of organic carbon and nitrogen measured in Lake Alexandrina. Errors shown are ± standard error. All carbon is assumed to be in the organic form because TOPM/2 > TC for all samples measured.

<table>
<thead>
<tr>
<th></th>
<th>Organic carbon g m$^{-2}$ d$^{-1}$</th>
<th>Total nitrogen g m$^{-2}$ d$^{-1}$</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>10 Mar – 20 Mar</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Central</td>
<td>26.6 ± 2.9</td>
<td>2.7 ± 0.3</td>
</tr>
<tr>
<td>South central</td>
<td>39.1 ± 2.9</td>
<td>4.0 ± 0.3</td>
</tr>
<tr>
<td>East central</td>
<td>23.6 ± 2.9</td>
<td>2.6 ± 0.3</td>
</tr>
<tr>
<td>South</td>
<td>34.7 ± 2.9</td>
<td>3.4 ± 0.3</td>
</tr>
<tr>
<td><strong>20 Mar – 24 Mar</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Central</td>
<td>34.2 ± 1.5</td>
<td>3.2 ± 0.1</td>
</tr>
<tr>
<td>South central</td>
<td>40.5 ± 1.5</td>
<td>3.8 ± 0.1</td>
</tr>
<tr>
<td>East central</td>
<td>21.8 ± 1.5</td>
<td>2.1 ± 0.1</td>
</tr>
<tr>
<td>South</td>
<td>28.6 ± 1.5</td>
<td>2.5 ± 0.1</td>
</tr>
</tbody>
</table>
Figure 5.1: Location of deployed sediment traps within Lake Alexandrina. Black circles show sites with a single trap only. Open circles show sites where triplicate traps were deployed in February and March 2010. The location of the turbidity meter is indicated by a +. Wind speed and direction was recorded at East central every 15 minutes for the duration of the experiment.
Figure 5.2: Variation of gross sedimentation rate and depth of wave-mixed layer (D_{WML}). Lines show an exponential fit at each site with r^2 and p values indicated.
Figure 5.3: Particle size distribution of sediments, trap contents (TPM) and suspended particles (SPM) averaged across all deployment periods. MPS\textsubscript{trap} is the mean particle size of trap contents at each site with 95% confidence interval shown.
Figure 5.4: Time series of turbidity and the depth of the wave mixed layer calculated using linear wave theory for deep water. Turbidity was measured 2.5 km southeast of South central and the depth of the wave mixed layer was measured at Central and East central during high mean wind speed (Top panel; $W_{\text{mean}}$ ± 95% C.I. = 6.62 ± 0.23 m s$^{-1}$; correlation between $D_{\text{WML}}$ and turbidity: $r^2 = 0.47$, $p < 0.0001$) and low mean wind speed (Bottom panel; $W_{\text{mean}}$ ± 95% C.I. = 4.19 ± 0.28 m s$^{-1}$; correlation between $D_{\text{WML}}$ and turbidity: $r^2 = 0.50$, $p < 0.0001$) deployments (Note that the ordinate axis for turbidity is scaled differently between the two graphs and the depth of the wave mixed layer is inverted for clarity).
Figure 5.5: Distribution of sedimentation with water depth in Lake Alexandrina as measured using all sites across changing water depths for the duration of the experiment.
6. The influence of sediment resuspension on light availability in a shallow lake following extreme water level decline: implications for lake productivity

6.1 Introduction

Light availability can determine the amount of and relative contributions to primary production by macrophytes, benthic algae and phytoplankton in aquatic ecosystems. By altering the form of primary productivity, changes in light availability has implications for the food web structure. Numerous factors influence the light available for photosynthesis, including lake depth and mixing regime, the concentration, type and size of inorganic and organic particles suspended in the water column and the concentration of coloured dissolved substances (Douglas 1997; Douglas & Rippey 2000). In shallow lakes, where wind energy is readily imparted onto the sediment surface, particle resuspension plays a dominant role in determining light availability by influencing these factors (Kirk 1983).

Turbulence imparted onto the sediment surface from wave action generates the shear forces that can overcome the gravitational and cohesive forces holding bottom sediments in place. Turbulent mixing can then distribute the resuspended particles
vertically and horizontally through the water column (Hellström 1991). The increased concentration of suspended particles affects light penetration through higher absorbance and scattering of incident light. Thus, the rate of light attenuation is higher when particle concentrations have increased (Fischer et al. 1979; Evans 1994).

Sediment resuspension can also increase the concentrations of dissolved substances that absorb incoming light through numerous mechanisms (van de Hulst 1957). The surface area of suspended particulate matter is more exposed to the water and the rate of desorption of mineral-bound dissolved organic carbon (DOC) increases (Kirk 1976; Karlsson et al. 2009; Schelske et al. 2010). DOC is further produced by photolysis of particulate organic material as high energy ultraviolet light is incident on a water body (Komada & Reimers 2001; Koelmans & Prevo 2003). Resuspended particles also attenuate UV light and so reduce photo-oxidation of DOC compared with less turbid conditions (Kieber et al. 2006).

Light availability for photosynthesis can be limited by resuspended particles (Kieber et al. 2006). This light limitation may act to advantage phytoplankton species over benthic microalgae and submerged macrophytes through physiological adaptations to low light intensities, physical mixing into different light conditions and morphological adaptations such as buoyancy regulation or flagella that can influence the distribution of phytoplankton with depth (Geddes 1984; Lind et al. 1994; Lawson et al. 2007). These differences allow individual species to maximise the amount of light captured under different mixing regimes (Sherman et al. 1998; Bormans et al. 2005).

Water level decline in lakes can interact with sediment resuspension to influence light availability and primary productivity. Decreasing water level reduces the depth through which light must pass to enable photosynthesis by primary producers in both the pelagic and benthic zones. However, the depth of water through which wind
energy is attenuated decreases and this can increase the area of sediment exposed to wind-induced turbulent energy (Phlips et al. 2000; Strand & Weisner 2001; Torremorell et al. 2007). Furthermore, if sediments are distributed according to water depth (Effler & Matthews 2004), the composition of sediments exposed to turbulence may change as the shoreline recedes (Håkanson & Jansson 1983; Hilton 1985). The type of sediment deposited on a lake-bed is an important factor in determining resuspension and its effects on light attenuation. This is because the type of sediment will not only influence the amount and size of sediment particles resuspended, but also their chemical composition, optical properties and residence time in the water column (Håkanson et al. 2000, Chapter 4).

This study examines the dynamics of sediment resuspension and its influence on the light climate in the deepest region of Lake Alexandrina during a period of extremely low water levels. The aims were threefold: (i) to understand if and when sediment is being resuspended from beneath the deepest waters of the lake; (ii) to determine the influence of this resuspended sediment on light attenuation; and (iii), to explore the influence of a changing light climate on lake productivity under conditions of extreme water level drawdown.

6.2 Methods

6.2.1 Study site

Lake Alexandrina is a shallow (March 2010: $Z_{\text{max}} = 2.2$ m, $Z_{\text{mean}} = 1.3$ m) coastal freshwater lake at the end of the River Murray, Australia. It is 34 km wide along the east-west axis and 17.8 km wide along the north-south axis. During the course of this study, water levels were 1.65 m below the long-term mean (1940 – 2006) and the water was brackish (3.9 g.L$^{-1}$) due to the evaporative concentration of salt and seawater ingress. Lake area fluctuates depending on water levels; it is 580 km$^2$ when
full and was 430 km² during this study. The lake has been characterised as a eutrophic to hypereutrophic environment and is considered light-limited for phytoplankton growth (Hanlon et al. 1998, Chapter 5). During the study period, the lake was dominated by the blue-green alga, *Planktolyngbya subtilis*, and it has been affected by numerous toxic algal blooms since European settlement (Geddes 1984). Sediments have accumulated in the lake since the construction of five barrages that separate Lake Alexandrina from downstream estuaries, and are frequently resuspended by wind driven wave mixing (Chapter 5).

### 6.2.2 Instrument deployment

Intensive field measurements were made over 70 hours between 1 pm on 21 March and 11 am on 24 March 2010. Instruments were deployed in the middle of Lake Alexandrina (S35.60313 E139.33638; Figure 6.1) over the flat profundal zone at the maximum water depth of 2.2 m (Figure 6.2). A Laser *In Situ* Scattering and Transmissometry instrument (LISST-100, Sequoia Scientific, Washington USA) was deployed 0.3 m above the sediment surface to measure particle size concentration. Measurements were recorded at 2 Hz for 10 seconds every minute. The optical transmission of the laser decreased gradually to zero from 4 am until 11:30 am on the morning of 23 March, most likely as a result of the instrument sinking into the soft mud sediment present at the study site. The instrument was cleaned, repositioned and reset so that optical transmission showed unobstructed laser transmission for the remainder of the experiment. LISST data from one hour either side of this event were not considered in data analysis. Raw particle size data was reported as cubic micrometres of suspended sediment volume per cubic metre of water (µm³/m³) in six categories of nominal particle size: <4.43 µm, 4.43 – 6.24 µm, 7.36 – 19.9 µm, 23.5 – 63.3 µm, 74.7 – 186 µm, and >219 µm, as described in Chapter 4.

A 16MHz Acoustic Doppler Velocimeter (ADV; SonTek, San Diego USA) sampled velocity in three planes at 20 Hz for 15 minutes 13 times between 22 and 23 March
over a range of wind speeds (Figure 6.2). This instrument recorded three-dimensional water velocity in a small sample volume 0.3 m above the sediment surface. Acquiring and processing of ADV data for spectral analysis was undertaken using HorizonADV. For the duration of this experiment, wind speed and direction (Vaisala WMS 301, Vaisala Pty. Ltd, Victoria Australia) was logged every minute at a site 9 km east of the deployment site (Figure 6.1). Wind was scaled to 10 m above the lake surface from a measured height of 6.7 m using MATLAB scripts from the Lake Analyzer software package (Codd et al. 1994). A multi-probe sonde (Hydrolab, Colorado USA) was used to measure nephelometric turbidity, electrical conductivity and temperature 1.5 m above the sediment surface every 5 minutes from 1 pm on 22 March until 11 am on 24 March.

Between 9 am and 6 pm on 22 and 23 March, scalar irradiance \( E_0 \) was recorded at 0.2 m depth intervals from the water surface every half an hour using a LI-COR underwater spherical quantum sensor with a LI-1400 data logger (LI-COR, Nebraska). At each depth, three instantaneous readings of scalar irradiance were averaged and used to calculate the vertical attenuation coefficient \( k_d \) and the euphotic depth \( z_{eu} \). \( k_d \) was calculated using equation 6.1, by linear regression between log-transformed irradiance and depth to give \( k_d \) as the slope. Fits were tested for linearity to confirm that \( k_d \) did not change with depth. \( E_z \) and \( E_0 \) are scalar irradiance readings at water depth \( z \) and immediately below the water surface, respectively (Read et al. 2011).

\[
E_z = E_0 \times e^{-k_d z} \tag{6.1}
\]

The euphotic depth, defined as the depth at which light intensity falls to 1% of surface irradiance (Kirk 1983), was determined using equation 6.2:

---

Water temperature was measured with 7 thermistors (RBR Ltd. Ottawa, Canada) spaced at 0.25 m intervals between the surface and 1.5 m depth. These were programmed to log temperature every 10 minutes from 4 March.

During the same period, dissolved oxygen concentration (DO) was logged with D-Opto luminescent dissolved oxygen sensors (Zebra-Tech Ltd. New Zealand) at 4 depths (0.25, 0.75, 1.25 and 1.75 m) every 15 minutes. Photosynthetic production was estimated from the diel rate of change in oxygen concentration on days where the water column was well mixed and oxygen saturation fluctuated around 100%. Under these conditions, over a diel period, the net gas flux between the water column and atmosphere can be considered negligible (Kirk 1983). Net production and respiration were estimated by rates of change in oxygen concentration during light and dark periods, according to Staehr et al. (2010). Gross production was calculated as the sum of net production and respiration over a diel period.

Instrument loss prevented data from being collected from the thermistors and DO sensors after 9:30 am 22 March. The LISST (1.9 m from the water surface) and multi-probe sonde (0.7 m) recorded temperature and these data were used to extend the temperature profile for the remainder of the field campaign. The LISST temperature data was post-calibrated according to the following assumption: when the thermistors showed that the uppermost 1.5 m of water was mixed, it was assumed that the mixed layer comprised the entire water column. The average temperature difference between the thermistors and the LISST over this period was calculated and used to offset the remaining temperature record of the LISST. This assumption was justified because during high wind speeds on 22 and 23 March, temperature recorded by the LISST and multi-probe sonde were closely aligned (Figure 6.3). Thus, while the temperature data no longer had the vertical resolution of a thermistor.
chain, it provided relative changes in temperature that were sufficient to elucidate stratification and destratification events.

### 6.2.3 Water samples

Water samples were collected from three depths (0.3, 0.8 and 1.3 m) every hour between 9am and 5pm on 22 and 23 March, 2010 using a modified microsampler (Staehr et al. 2010) with bottle volume of 200 mL. Water samples were analysed for suspended particulate matter (SPM), suspended inorganic particulate matter (SPIM), suspended particulate organic matter (SPOM), chlorophyll a (chl$\alpha$) and particle size distribution according to methods outlined in Chapter 4. An aliquot of water was 0.45 μm filtered (Sartorius Biotech GmbH PES membrane filter) immediately after collection from 0.8 m depth. The filtrate was analysed for absorbance at 440 nm to estimate the concentration of gilvin ($g_{440}$) according to Kirk (Baker 1970). Chl$\alpha$ concentrations were converted to algal-SPOM using a multiplier of 40 (1976). This assumes that chl$\alpha$ comprises 2.5% of the cells ash free dry-weight. This assumption was justified because cell counts showed that phytoplankton was dominated by cyanobacteria (> 98% of cell counts) and the concentration of diatoms with their silica frustules was insignificant (Dominic Skinner, unpublished data). Non-algal SPOM was calculated as the difference between SPOM and algal-SPOM.

Stepwise multiple regressions were run using JMP-IN 8.0 (SAS Institute Inc.). SPIM, algal-SPOM, non-algal-SPOM and $g_{440}$ were regressed against $k_d$ to elucidate the components of sediment resuspension that affected light availability. An α level of 0.05 was considered to be significant for all factors of the regression.

### 6.2.4 Determining critical benthic shear stress

The critical shear stress ($\tau_c$) is the force at which wave- or current-induced shear at the sediment surface overcomes the cohesive, gravitational and friction forces that keep sediments deposited (Reynolds 1984). $\tau_c$ can be estimated from sediment
properties and those of the water into which the sediment is being resuspended using the equation from Chung et al. (2009):

\[ \tau_{cr} = \tau_{cr}^* \rho_s R g D \quad (6.3) \]

where \( \rho_s \) is the sediment density, \( D \) the particle diameter, \( g \) the acceleration due to gravity (9.81 m s\(^{-2}\)) and \( R \) is the submerged specific gravity, which was calculated using the density of water (\( \rho \)) at 20°C and salinity of 3.9 mg L\(^{-1}\) measured during the course of this experiment:

\[ R = \frac{\rho_s - \rho}{\rho} \quad (6.4) \]

The parameter, \( \tau_{cr}^* \), is the critical Shields’ parameter that can be estimated using formula provided by Chung (2009):

\[ \tau_{cr}^* = 0.5 \left[ 0.22 \text{Re}_{p}^{-0.6} + 0.06 \cdot 10^{(7.7 \text{Re}_{p}^{-0.4})} \right] \quad (6.5) \]

where \( \text{Re}_{p} \) is the dimensionless Reynolds particle number estimated according to equation 6.6 and using the kinematic viscosity of water (\( \nu \)) as 1.004 \times 10^{-6} m\(^2\) s\(^{-1}\):

\[ \text{Re}_{p} = \frac{D \sqrt{g R D}}{\nu} \quad (6.6) \]

For these calculations, the density of sediment was taken to be 1210 kg m\(^{-3}\) as measured at the study site in previous experiments (Chapters 3 and 4). In addition, the settling velocity (\( V_s \)) of each particle size category was estimated using Stokes law Parker et al. (2003):

\[ V_s = \frac{g D^2 (\rho_s - \rho)}{18 \nu} \quad (6.7) \]

As Stokes law only holds for quiescent conditions, calculated sedimentation times were modified for turbulent conditions according to Reynolds (Massey 1979),
whereby sedimentation is explained by an exponential decay curve such that 99% of particles are cleared from the water column after 4.6 times the calculated sedimentation time.

The calculated values of $\tau_{cr}$ were tested against empirical data to assess the applicability of these calculations to the sediments in Lake Alexandrina. This was achieved by determining the bed shear stress derived from orbital wave motion ($\tau_b$), which is given by equation 6.8 according to Wiberg (1995):

$$\tau_b = \rho C_D u_b^2$$  \hspace{1cm} (6.8)

where $C_D$ is a dimensionless friction coefficient, taken as 0.0022 for mud and $u_b$ is the orbital velocity at the sediment surface. This orbital velocity is parameterised according to measures of wave-induced turbulence given by Wiberg and Sherwood (2008) as:

$$u_b = \sqrt{2(u' + v')}$$  \hspace{1cm} (6.9)

where $u'$ and $v'$ are second moment statistics describing the fluctuating component of velocity readings. These can be enumerated by subtracting the mean over 5 seconds from the measured velocity (Soulsby 1983). $\tau_{cr}$ was then estimated empirically by regressing $\tau_b$ against the rate of change in turbidity (dTurb/dt in NTU hr$^{-1}$) for the hour following onset of each ADV measurement. The transition between sedimentation (where dTurb/dt was negative) and sediment resuspension (where dTurb/dt was positive) was taken to be the range in which $\tau_{cr}$ occurred.
6.3 Results

6.3.1 Wind and wave induced sediment resuspension

Minimum wind speeds occurred either side of noon on 22 and 23 March 2010 as winds turned from an easterly land breeze towards a westerly sea breeze (Figure 6.4). Westerly wind speeds increased through the afternoon on both days, before turning again in the evenings to blow from the east or southeast and decreasing in intensity until about noon on the following day. As wind speed increased, the formation of waves became clearly evident in spectral plots as shown by the peaks at a frequency between 0.4 and 0.5 Hz (Figure 6.5). There was a delay between the increasing wind speed and the onset of wave formation. For example, wind speeds increased after noon on the 22 March, but between 2:48 pm and 3:45 pm on 22 March, while there was no further increase in wind speed, waves were present at 1.9 m depth in the spectral plots for the latter time only (Figure 6.5). A corresponding delay was observed between increasing wind speed and the resuspension of sediments, as measured by turbidity (Figure 6.3). The increase in turbidity coincided with the breakdown of transient temperature stratification that had set up during periods of low wind speed (and high solar radiation) around noon on both days.

6.3.2 Estimates of critical shear stress

Calculations of $\tau_c$ for sediments at the study site suggest that resuspension occurs at bed shear stresses of 0.0035 N m$^{-2}$ for particles 2.7 µm in diameter and of 0.47 N m$^{-2}$ for particles larger than 219 µm (Table 6.1). To resuspend the mean particle size of sediments measured in Chapter 5 (26µm), the shear stress at the sediment surface would need to be greater than 0.031 N m$^{-2}$. The relationship between the rate of change in turbidity and log $\tau_b$ measured by the ADV was positive and linear ($d\text{Turb}/dt = 6.07 \log \tau_b + 8.5; r^2 = 0.76, F = 25.1, p = 0.001$). This relationship allowed $\tau_c$ to be estimated empirically, as the point where turbidity increased due to
sediment resuspension. Mathematically, this occurs at a shear stress greater than 0.04 N m$^{-2}$, which corresponds to the point that the line of best fit crosses the abscissa. However, considering the spread of wind conditions in which $\tau_b$ was calculated from measured water velocity, the range of $\tau_{cr}$ (95% Confidence Interval) would be between 0.012 N m$^{-2}$ and 0.12 N m$^{-2}$ (Figure 6.6). These empirical estimates align well with the values of $\tau_{cr}$ suggested by theoretical calculations.

6.3.3 Sediment resuspension and particle size characteristics

Particle size distribution was measured at 1.9 m below the water surface, which was within the hypolimnion during periods of stratification (Figure 6.3). Changes to particle size distribution coinciding with the point of mixing were greatest for larger particles. At the point of water column mixing on 22 March (3:00 pm; Figure 6.7), the concentration of particles in the largest category (> 219 µm) decreased, as did particles sized between 74.7 and 186 µm (Figure 6.7). Smaller particles showed the same pattern of concentration change, but the effect was smaller. Thereafter, there was a steady increase in concentration of particles as winds increased, with those larger than 219 µm increasing further and for longer than other particle size categories. On 23 March, at the point of destratification (2:50 pm), particles in the largest particle size category (> 219 µm) again decreased in concentration, but all other particles appeared to be less affected. As with the previous day, particles larger than 219 µm showed the largest and most sustained increase in concentration following the mixing event.

The value of $\tau_b$ measured at 3:45 pm on 22 March was 0.15 N m$^{-2}$, which was smaller than $\tau_{cr}$ calculated for particles larger than 219 µm. In contrast, at 4:32 pm 23 March, $\tau_b$ was 0.55 N m$^{-2}$, which was significantly larger than that required to resuspend the largest particle class (Figure 6.7). The LISST recorded a decline in the total volume of particles during the first mixing event, from 1700 µm$^3$/m$^3$ at 2:30 pm to a low of 595 µm$^3$/m$^3$ at 4:55 pm on the 22 March. There was no matching
reduction in total volume concentration for the second mixing event on the 23 March, but two data spikes coincide with the point of mixing.

Settling velocity of particles with a nominal diameter of 23.5 µm was calculated to be 0.27 m hr\(^{-1}\) (Table 6.1). For particles 74.7 µm in diameter, the settling velocity was 2.7 m hr\(^{-1}\), while particles with a diameter of 219 µm would fall at 23.3 m hr\(^{-1}\). For 99% of particles to have settled out of the water column in turbulent conditions, these particles would take between 0.4 hours (219 µm) and 37.5 hours (23.5 µm), provided no further resuspension took place. Particles sized 74.7 µm would take 3.8 hours to settle under the same conditions.

### 6.3.4 Influence of sediment resuspension on light attenuation

Stepwise multiple regression revealed that resuspended particulate matter dominated light attenuation \(k_d = 0.025 \text{ SPIM} + 0.033 \text{ non-algal SPOM} + 0.89; r^2 = 0.78, F = 41.5, p < 0.0001\). SPIM \((p < 0.0001)\) and non-algal SPOM \((p < 0.0001)\) were significant contributing factors to light attenuation, whereas algal-SPOM \((p = 0.33)\) and \(g_{440}\) \((p = 0.17)\) were not.

SPM (mg L\(^{-1}\)) was positively correlated with chl\(_a\) (µg L\(^{-1}\), \(r^2 = 0.73, F = 145.9, p < 0.0001\)) but had concentrations over three orders of magnitude higher. There was no obvious seasonality in chl\(_a\) concentrations (Chapter 3), suggesting that hydrodynamics play a dominant role in determining the concentration of chl\(_a\) in suspension. Additionally, SPM was strongly correlated with turbidity measured in nephelometric turbidity units \((r^2 = 0.91, F = 128.7, p < 0.0001)\), which allowed measured turbidity to be used to calculate SPM when samples were not taken (see Figure 6.8) according to equation 6.10:

\[
SPM = 1.78 \text{Turbidity} + 16.43
\]  
(6.10)
On 22 March 2010, SPM reached a minimum of $151.5 \pm 6.7 \text{ mg L}^{-1}$ at 3 pm, rising to $166.1 \pm 2.4$ and $194.3 \pm 5.3 \text{ mg L}^{-1}$ at 4 pm and 5 pm, respectively, as wind speed increased. Similarly, on 23 March 2010, after a minimum of $157.5 \pm 10.3 \text{ mg L}^{-1}$ at 2 pm, SPM increased to $161.6 \pm 0.3$ then $196.7 \pm 2.0 \text{ mg L}^{-1}$ in the following two hours, which again corresponded to increasing wind speeds (Figure 6.8). There was a positive linear relationship between turbidity and light attenuation ($r^2 = 0.78$, $F = 138.6$, $p < 0.0001$), shown by equation 6.11:

$$K_d = 0.058 \text{Turbidity} + 0.69 \quad (6.11)$$

### 6.3.5 Dissolved oxygen profiles

The interactions between wind and temperature stratification influenced the distribution of DO in the water column (Figure 6.9). When wind speeds were persistently high between 8 March and 9 March 2010, the diurnal fluctuation in DO was minimal. During well-mixed conditions, gross primary production ranged between $29.4$ and $123.2 \text{ g O}_2 \text{ m}^2 \text{ d}^{-1}$ (Table 6.2). The lowest production corresponded with the lowest solar radiation and highest wind speed.

### 6.4 Discussion

#### 6.4.1 Physical conditions

Wind energy imparted onto the water surface of Lake Alexandrina generated shear stress at the sediment surface dominated by orbital wave motion, which showed as distinct peaks in the spectral plots of water velocity (Figure 6.5). Additionally, calculated $\tau_{cr}$ estimates corresponded closely to empirical estimates using measured $\tau_b$ from wave-induced turbulence (Figure 6.6). Other studies of shallow lakes have also shown that wave activity is the dominant driver of sediment resuspension.
(Andersen et al. 2007; Wiberg & Sherwood 2008), because wave-induced orbital motion is generally responsible for water velocities at the sediment surface 5-10 times larger than currents (Bengtsson et al. 1990; Mian & Yanful 2004; Jin & Sun 2007).

This study demonstrated that transient stratification, in the order of a few hours, delayed the wind-induced resuspension of bottom sediments, preventing a clear relationship between wind speed and SPM concentrations or turbidity. Studies linking wind-driven sediment resuspension with changing light climate often ignore this time offset between wind-induced surface mixed layer deepening and sediment resuspension (Andersen et al. 2007). However, the observation of a time lag between increasing wind speed and a rise in the concentration of suspended solids has been previously reported (e.g. Hanlon et al. 1998). These observed delays were caused by either a lag between increasing wind speed and current velocity reaching a sufficient threshold to overcome the critical shear stress of sediments (Podsetchine & Huttula 1994; Jin & Sun 2007), or else due to the erosion of temperature stratification (Podsetchine & Huttula 1994).

### 6.4.2 Sediment resuspension

*In situ* particle size data suggested that particles with a diameter greater than 219 µm dominated the particles entrained in the water column. This observation was supported by the calculated $\tau_{cr}$ for particles of 219 µm in diameter, which was 0.28 N m$^{-2}$. This was lower than values of $\tau_b$ measured in the afternoon of both days during periods of sediment resuspension and the shear force would be sufficient to resuspend all sediment particles up to 219µm in diameter. Grab samples and turbidity measurements both indicated an increase in particle load following the onset of wave activity at the sediment surface, and this corresponded with a steady increase in the concentration of particles with diameters greater than 219 µm as measured by the LISST (Figure 6.7).
Nevertheless, a number of inconsistencies remain with the conclusion that particles with diameters over 219 µm were being resuspended. For instance, the resuspension-induced increase in turbidity (Figure 6.3) and SPM (Figure 6.8) took in the order of 12-16 hrs to return to pre-resuspension levels. The settling velocity of particles with diameters of 219 µm is 23.3 m hr\(^{-1}\) and so 99% of these particles would have been removed from the water column after 0.4 hours when no more resuspension was occurring. For particles sized 74 µm, the sedimentation time would be 4.6 hours, still well below observations. To clear the water column in 12-16 hours requires a settling velocity of ~0.15 m hr\(^{-1}\), which would be indicative of particles sized ~33 µm in diameter. Additionally, a previous study of resuspension in the middle of Lake Alexandrina (Chapter 5) showed that the mean size of settling particles caught in sediment traps was 23.6 µm and that surface sediments at the site were dominated by fine particles (Figure 5.3). This apparent paradox can be reconciled if it is assumed that particle flocculation occurred during the study, which has been observed in other areas of the Murray-Darling Basin during periods of increasing salinity (Jin & Sun 2007). Flocculation would explain a difference in particle size when measured \textit{in situ} compared with that of grab samples that are stored and resuspended \textit{ex situ} for analysis (e.g. Grace \textit{et al.} 1997) as was done in Chapter 5. In addition, grab samples taken during the course of this study showed particle sizes dominated by fine particles (Dominic Skinner, unpublished data). Finally, the density of flocs would be lower than that of solid particles with the same diameter, thereby reducing the expected settling velocity.

As the LISST was deployed within the hypolimnion, the effects of concentrating particles below the thermocline would result in a sharp reduction of particle concentration upon mixing as these particles become distributed throughout the entire water column. Alternatively, as the thermocline deepened, prior to resuspension, the LISST would temporarily be recording data in the epilimnion, which would result in a similar observation. Indeed, such an event occurred during
initial mixing on 22 March. There was, however, no corresponding drop in particle concentration on 23 March, perhaps due to a lower amount of resuspension during the previous day.

6.4.3 Light availability for primary productivity

Resuspended sediments control aspects of light availability in the middle of Lake Alexandrina. Only SPIM and non-algal SPOM were significant predictors of light attenuation based on a stepwise multiple regression. This is consistent with Geddes (Phillips & Walling 1995) who suggested that inorganic turbidity in Lake Alexandrina was limiting light-availability. The importance of non-algal SPOM is consistent with the increasing concentration of organic matter in sediments between 2007 and 2009 as water levels declined and a lack of outflow led to the accumulation of organic-rich detritus in surface sediments (Chapter 4).

The factors affecting light attenuation were identified from water samples taken over two days in autumn, when temperatures were high and wind speeds low (Chapter 7). Under these conditions phytoplankton abundance is probably higher than other seasons, except perhaps summer. However, long-term chlorophyll records showed no apparent seasonality (Chapter 3). This suggests that interactions between hydrodynamics, sediment resuspension and light presented here may be considered representative of processes within Lake Alexandrina that aren’t restricted to a specific time of year.

The dynamics and availability of light can determine the structure of a lake ecosystem. Phytoplankton have various advantages (such as buoyancy regulation and transient mixing through the photic zone) over benthic microalgae or rooted macrophytes that enable them to survive in low light intensities (1984). Thus during turbid conditions, phytoplankton can become the dominant primary producer in lakes. The maximum euphotic depth recorded in this study was 0.91 m. Given that
the depth at the study site was 2.2 m, light limitation almost certainly prevented photosynthesis at the benthos. This hypothesis was supported by the net loss of oxygen at depths of 1.25 m and 1.75 m during stratified conditions (Figure 6.9).

6.4.4 Light limitation in the pelagic

Light limitation in pelagic waters is generally taken to mean conditions that prevent phytoplankton from achieving their maximum growth potential provided other factors contributing to growth, such as nutrient supply, are sufficient (Ferris & Christian 1991; Prézelin et al. 1991). This is difficult to measure in situ and so other methods, including modeling (Reynolds 1997) and estimates from light availability and mixing regimes (Grobbelaar 1990; Lind et al. 1994), are more commonly used. A number of methods that examine the light climate for phytoplankton growth in Lake Alexandrina are presented here.

The optical density (OD) is the product of mixing depth \( z_{\text{mix}} \) and \( k_d \), which when greater than 16, provides strong empirical evidence of light limitation (Riley 1957; Talling 1971; Scheffer 1998). It follows that OD can be calculated from the relationship between \( k_d \) and turbidity, by substituting equation 6.11 to give equation 6.12:

\[
OD = z_{\text{mix}} (0.058 \text{Turbidity} + 0.69) \quad (6.12)
\]

Given that \( z_{\text{mix}} \) has a maximum value of 2.2 metres in Lake Alexandrina during this study, then the turbidity that leads to light limitation (where \( OD \geq 16 \)) during fully mixed waters must be greater than 113.5 NTU. This threshold was surpassed in the late afternoon of both 22 and 23 March. However, during periods of transient stratification, \( z_{\text{mix}} \) would be lower, thereby requiring higher turbidity to achieve the same optical density. As stratified conditions formed around noon, net photosynthetic production could occur when solar radiation was at its highest. A
The turbidity threshold of 113.5 NTU corresponds to a light attenuation of 7.27 m\(^{-1}\), a euphotic depth of 0.63 metres and a \(z_{mix} : z_{eu}\) ratio of 3.49.

Lind et al. (Scheffer 1998) showed that light limited phytoplankton biomass in Lake Chappala at a \(z_{mix} : z_{eu}\) ratio of less than four, similar to values observed here. This is in contrast to Talling (1994) who suggested that a \(z_{mix} : z_{eu}\) ratio of greater than 5 is required for light-limitation in temperate lakes. Lind et al. (1971), suggest that a ratio of 5 is too high for warmer tropical lakes, where plankton respiration is likely to be significantly higher. As Lake Alexandrina is located in a semi-arid region, the respiration rate of plankton would likely be higher than that of temperate lakes due to warmer average temperatures and will therefore influence light requirements (1994).

As the results from this study are taken from the deepest section of Lake Alexandrina, a number of factors will influence its comparison with light-limitation in shallower sections of the lake. Notably, because \(z_{mix}\) has a lake-wide maximum of 2.2 metres, light attenuation would need to be higher at other sites to achieve an optical density or \(z_{mix} : z_{eu}\) ratio large enough to suggest light-limitation. At sites with shallower water, less wind energy is required to overcome the shear stress at the sediment surface such that sediments can be more prone to resuspension and corresponding higher values of \(k_d\).

One major drawback from these estimates of light limitation from mixing regimes is their independence from the dynamic interplay between transient stratification (that effects mixing depth) and sediment resuspension (that influences \(k_d\)) in shallow, polymictic lakes. Furthermore, these interactions need to be considered in the context of changing levels of incoming solar radiation over diurnal periods. While the observed changes in suspended sediment in this study resulted in changes to light attenuation that may have resulted in light limitation, the likely effect on pelagic productivity would have been minimal because SPM concentrations were highest in the evening of both days when solar radiation was negligible. The declining wind
speeds overnight also provided long periods for which particles could settle out of
the water column, reducing the impact of a previous resuspension event on the light
climate the following day. Ultimately, light limitation should be based on integrated
light availability over a diel period to account for these factors.

To enable the results of this study to be compared with earlier studies of Lake
Alexandrina when water levels were higher (Beklioglu et al. 2007), the average
irradiance through the water column (\( \overline{E}_z \)) was estimated to give a quantitative
description of the effects of light availability on phytoplankton growth according to
equation 6.13 (Geddes 1984):

\[
\overline{E}_z = \frac{E_o(1 - e^{-k_d z_{\text{mix}}})}{k_d z_{\text{mix}}} \quad (6.13)
\]

Earlier work on Lake Alexandrina showed that between October 1975 and March
1978, \( \overline{E}_z \) varied from 38.1 to 168 \( \mu \text{mol photons m}^{-2} \text{s}^{-1} \) when calculated with the
mean depth at the time of 2.86 m (Geddes 1984). This suggested light-limitation
because the required irradiance for phytoplankton production was shown to be at
least 180.9 \( \mu \text{mol photons m}^{-2} \text{s}^{-1} \) in the Sargasso Sea by Riley (1957). In Mount Bold
reservoir, South Australia, at least 150 \( \mu \text{mol photons m}^{-2} \text{s}^{-1} \) was required to
stimulate phytoplankton growth (Oliver 1981). Conversely, during the low water
levels of this study, the mean \( \overline{E}_z \) was 207.2 \( \mu \text{mol photons m}^{-2} \text{s}^{-1} \) when calculated
using the maximum depth of Lake Alexandrina, and 350.4 \( \mu \text{mol photons m}^{-2} \text{s}^{-1} \)
when calculated with the mean depth (1.3 metres) as was done by Geddes (1984,
Figure 6.9). This suggests that in this study phytoplankton were not light-limited in
Lake Alexandrina, as they were between 1975 and 1978. During a later period of
low inflows with low turbidity (as the contribution of inflows from the turbid Darling
River were low) and increasing salinity (to 0.7 g.L\(^{-1}\)), Geddes (1988) showed that
water clarity increased, with \( \overline{E}_z \) reaching 240 \( \mu \text{mol photons m}^{-2} \text{s}^{-1} \) and \( z_{\text{eu}} \) over 3
m. The current experiment was done under conditions of low inflows, drought-
induced brackish salinity (3.9 g.L\(^{-1}\)), similarly low contribution of flows from the
Darling River (Table 2.1) and substantially lower water levels. While the average irradiance in the water column was similar to the previous clear phase of Lake Alexandrina, $z_{eu}$ was much lower (1981: 3.1 m; maximum recorded in 2010: 0.91 m). This is consistent with increased resuspension of sediments as a result of substantially lower water levels.

6.4.5 Proportion of littoral sediments after water level declines

Sediments below shallow water in Lake Alexandrina are dominated by coarser substrate to that found below profundal waters (Chapter 4), but resuspension and redistribution of fine sediments appears to be relatively spatially homogenous (Chapter 5). It follows that these resuspended sediments would affect the light attenuation in a similar manner throughout the lake. If we define the littoral zone as the area where benthic primary production occurs (i.e. within the euphotic zone), a hypsographic curve for Lake Alexandrina (South Australian Department of Environment and Heritage) can be used to estimate the surface area of littoral sediments. The maximum euphotic depth recorded in this study was 0.91 m, which would give a littoral zone of 236.2 km$^2$, or 39% of the lake area. The minimum euphotic depth recorded directly in this study of 0.68 m corresponds to a littoral zone of 202.7 km$^2$ or 33% of the lake area. As wind speeds vary over a larger range than measured in this study, these estimates of littoral area would no doubt be substantially affected. However, the estimates do suggest that over 61% of sediments in Lake Alexandrina have insufficient light for photosynthetic growth in the benthic zone. Performing the same calculations using the euphotic depths reported by Geddes for historical water levels, it can be shown that during the turbid period between 1975 and 1978, the littoral zone encompassed between 3 and 15% of the sediment surface area. During the clear phase in 1981, where external conditions were similar to those of this study, the littoral zone was approximately 74% of the lakes surface area.
It is evident that declining water levels have affected sediment resuspension, through higher wave-induced turbulence at the sediment surface for a given wind speed. The implications for pelagic primary producers may be determined by the timing of resuspension events relative to the diurnal variation in solar radiation and thermal stratification. Pelagic light availability (average irradiance) increased as a result of declining water levels and lower maximum mixing depth, despite an increase in turbidity and light attenuation. Conversely, benthic light availability decreased after water levels had declined compared with previous drought periods. These results suggest that the light climate during extremely low water levels present in Lake Alexandrina favour primary production in the pelagic over the benthic zone.
Table 6.1: Critical shear stress and settling velocities for six particle size categories. Data were calculated according to equations 6.3 - 6.6. The reported mean sediment particle size was taken from sediment samples collected on 24 March 2010 for Chapter 5.

<table>
<thead>
<tr>
<th>Nominal particle size diameter (µm)</th>
<th>Calculated critical shear stress $\tau_{cr}$ (N m$^{-2}$)</th>
<th>Stokes settling velocity $V_s$ (m hr$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean (Range)</td>
<td>Mean (Range)</td>
<td>Mean (Range)</td>
</tr>
<tr>
<td>3.25 (2.7 – 3.8)</td>
<td>0.0042 (0.0035 – 0.0049)</td>
<td>0.005 (0.0035 – 0.007)</td>
</tr>
<tr>
<td>5.3 (4.43 – 6.24)</td>
<td>0.0069 (0.0058 – 0.008)</td>
<td>0.014 (0.0095 – 0.019)</td>
</tr>
<tr>
<td>13.6 (7.36 – 19.9)</td>
<td>0.018 (0.0095 – 0.026)</td>
<td>0.090 (0.026 – 0.19)</td>
</tr>
<tr>
<td>43.4 (23.5 – 63.3)</td>
<td>0.056 (0.030 – 0.082)</td>
<td>0.92 (0.27 – 1.95)</td>
</tr>
<tr>
<td>130.4 (74.7 – 186)</td>
<td>0.169 (0.096 – 0.22)</td>
<td>8.26 (2.7 – 16.8)</td>
</tr>
<tr>
<td>360 (219 – 500)</td>
<td>0.47 (0.28 – 0.65)</td>
<td>63 (23 – 122)</td>
</tr>
</tbody>
</table>

Mean sediment particle size at study site = 28.09 µm

$\tau_{cr}$ = 0.0305

$V_s$ = 0.38
Figure 6.1: Location of instrument deployment within Lake Alexandrina. High-resolution data was collected in Lake Alexandrina at the location marked with a black circle for a 70-hour intensive field campaign. The location of the wind recording station is shown by the 'W'.
Figure 6.2: Period of measurements recorded using different instruments during the 70 hr intensive field campaign. The duration of field experimentation is indicated at the top and was between 1 pm 21 March and 11 am 24 March 2010. Three instruments, DO sensors, thermistors and wind were being recorded from the 4th of March and are therefore shown to be extended prior to the field campaign with arrows to the left. Note: for the sake of comparison, figures 6.2, 6.3, 6.4, 6.7, 6.8 and 6.10 have their ordinate axes begin at 21 Mar 12 pm and end at 24 Mar 12 pm.
Figure 6.3: The influence of temperature stratification and wind speed on changes to turbidity. (Top) The temperature measured with a thermistor chain up until 9:30 am 22 March and extrapolated temperature data using the corrected LISST data from 1.9 m below the water surface and sonde temperature data recorded at 0.7 m below the water surface. (Bottom) The wind speed and turbidity, where wind speed has been smoothed with a 15-minute moving average filter.
Figure 6.4: Wind speed and direction recorded on Lake Alexandrina for the duration of the experiment. Wind speed data has been smoothed with a 15 minute moving average filter. Arrows indicate the wind direction, where up is north.
Figure 6.5: Frequency spectra of three components of water velocity (u, v and w) measured by the Acoustic Doppler Velocimeter. Inset values of $\bar{U}$ show the mean wind speed over the measurement period and $\tau_b$ is the measured wave-induced bottom shear stress.
Figure 6.6: Rate of change in turbidity and the logarithm of the shear stress ($\tau_b$) calculated from orbital velocity measurements with an Acoustic Doppler Velocimeter. Data were used to estimate the critical shear stress ($\tau_{cr}$). Sedimentation occurs when the rate of change in turbidity is negative, whereas erosion and sediment resuspension occurs when the rate of change in turbidity is positive. An estimate of $\tau_{cr}$ is made at the transition between sedimentation and erosion.
Figure 6.7: In situ particle size distribution of suspended sediments between 1 pm 21 March and 11 am 24 March 2010. Top panel shows the change in volume of six particle size groups. Bottom panel shows the change in volume of all particle sizes measured with the measured benthic shear stress (black diamonds) and the critical shear stress for resuspension (grey dashed line). All data has been smoothed with a 15 minute moving average filter for visual clarity. Two arrows indicate the point of destratification and the onset of increasing turbidity at 3:00 pm on 22 March and 2:50 pm on 23 March.
Figure 6.8: Time series of changes to suspended particles and the euphotic depth. Euphotic depth was measured between 9:00 am and 5:00 pm on 22 March and between 9:00 am and 6:00 pm on 23 March. Error bars on measured concentration of suspended particles show one standard deviation.
Figure 6.9: Vertical gradients of dissolved oxygen (top panel) compared with changes to wind speed and temperature stratification (bottom panel).
Table 6.2: Lake productivity estimates during well-mixed conditions in Lake Alexandrina. GPP: Gross primary production; CR: Community respiration; NEP: Net ecosystem production; Sol Rad: Solar radiation; Ave Wnd Spd: Average wind speed during daylight hours. Note: estimates do not include any gas exchange between the lake and the atmosphere.

<table>
<thead>
<tr>
<th>Date</th>
<th>GPP</th>
<th>CR</th>
<th>NEP</th>
<th>Sol Rad</th>
<th>Ave Wnd Spd</th>
</tr>
</thead>
<tbody>
<tr>
<td>6-Mar</td>
<td>102</td>
<td>-60</td>
<td>42</td>
<td>29109</td>
<td>4.3</td>
</tr>
<tr>
<td>8-Mar</td>
<td>29</td>
<td>-17</td>
<td>12</td>
<td>10303</td>
<td>13.4</td>
</tr>
<tr>
<td>9-Mar</td>
<td>92</td>
<td>-43</td>
<td>50</td>
<td>32502</td>
<td>9.8</td>
</tr>
<tr>
<td>10-Mar</td>
<td>103</td>
<td>-60</td>
<td>42</td>
<td>24361</td>
<td>6.9</td>
</tr>
<tr>
<td>11-Mar</td>
<td>120</td>
<td>-67</td>
<td>53</td>
<td>37504</td>
<td>5.7</td>
</tr>
<tr>
<td>19-Mar</td>
<td>106</td>
<td>-59</td>
<td>47</td>
<td>25405</td>
<td>5.4</td>
</tr>
<tr>
<td>21-Mar</td>
<td>123</td>
<td>-76</td>
<td>48</td>
<td>36261</td>
<td>6.3</td>
</tr>
</tbody>
</table>
Figure 6.10: Average irradiance in the middle of Lake Alexandrina calculated using $d = 1.3 \text{ m}$ (black diamonds) and $d = 2.2 \text{ m}$ (white squares). The dotted lines show the upper (168 $\mu\text{mol} \text{ m}^{-2} \text{ s}^{-1}$) and lower (38.1 $\mu\text{mol} \text{ m}^{-2} \text{ s}^{-1}$) bounds of average irradiance recorded by Geddes (1984) between October 1975 and March 1978. The unbroken grey line shows the highest average irradiance measured during a clear phase of Lake Alexandrina in 1981 where conditions (except water level) were similar to those in this study (Massey 1979; Lick et al. 1992).
7. The frequency of wind-driven sediment resuspension under different water levels in Lake Alexandrina

7.1 Introduction

Light is a dominant source of energy in aquatic ecosystems. In shallow, oligotrophic lakes, light has been shown to determine primary productivity levels (1984; 1988). Under eutrophic conditions, limited light availability inhibits productivity of benthic microalgae and rooted macrophytes (Karlsson et al. 2009) and even phytoplankton under certain mixing regimes (Lowe et al. 2001; Jackson 2003). In aquatic systems with significant pelagic photosynthetic production, an increase in the light penetration relative to the mixing depth is akin to nutrient enrichment (Scheffer 1998), enhancing primary productivity. While nutrient loading may be a better predictor of long-term biomass than light availability (Berger et al. 2006), variations in light attenuation can determine productivity levels over diel to seasonal time-scales. Interestingly, while there is a detailed understanding of the physiological responses of aquatic primary producers to changes in irradiance (Lowe et al. 2001), comparatively few field studies examine the temporal fluctuations in irradiance that drive these responses (Reynolds 1984; Cullen & Lewis 1988; Geider et al. 1996). This is despite the many processes affecting light availability that have a distinct periodicity in their occurrence. For example, lakes in coastal areas are subject to land
or sea breezes, driven by differential heating of the land and water surfaces that causes predictable daily wind patterns (Millie et al. 2003; Anthony et al. 2004; Torremorell et al. 2009). Large lakes may create their own daily wind cycles in a similar manner (Haurwitz 1947). Other examples include: predictable patterns in solar radiation that give rise to transient, daily temperature stratification in large, shallow polymictic lakes (Patrinos & Kistler 1977); tidal movement in shallow coastal estuaries that drives regular turbulence, increasing turbidity and light attenuation (Chapter 6, Rueda & Schladow 2003); and, periodicity in cloud cover, which can have an observable influence on the light available for photosynthesis (May et al. 2003). The timing and frequency of events that reduce water clarity relative to the amount of solar insolation are important for estimating lake productivity levels.

The time-scale over which variations in the light climate influence primary productivity is dependent on the factors driving the availability of light. Factors that cause light limitation include: eutrophication-enhanced phytoplankton standing crops (Anthony et al. 2004); resuspended inorganic particles (Phlips et al. 2000; Vadeboncoeur et al. 2001; Torremorell et al. 2007); dissolved organic carbon that results in highly coloured water (Geddes 1984; Lind et al. 1994; Lawson et al. 2007); or, the water itself, which absorbs incident solar radiation (Kirk 1976; Karlsson et al. 2009; Schelske et al. 2010). Attenuation of light by coloured compounds (following catchment runoff) or the water itself (after changes in the mixing depth of deep lakes) change over longer time frames in comparison to that caused by phytoplankton biomass or resuspended inorganic particles, which can vary relatively quickly. For example, changes to the light climate from resuspension in shallow lakes is possible in a matter of hours (Soto 2002).

Wind-induced resuspension of sediment has a dominant influence on light availability in shallow aquatic systems (Pierson & Weyhenmeyer 1994; van Duin et al. 2001; Chung et al. 2009). Depending on the sediment properties, resuspension
Periodicity and frequency of wind-driven resuspension

increases the absorption or scattering of incoming solar radiation, thereby increasing light attenuation (Hellström 1991; Nagid et al. 2001; Lawson et al. 2007).

Many studies that link sediment resuspension with light-limitation of primary productivity do so by examining discrete storm events (van Duin et al. 2001) or by integrating light availability through time (Hellström 1991). However, there have been comparatively few studies that link the temporal patterns in resuspension to light availability. Anthony et al. (Kristensen et al. 1992; Lawson et al. 2007) showed that photosynthesis could be predicted from modelled light availability determined by periodicity in cloud cover, tidal variation and wind-induced turbidity, provided that photo-acclimation was taken into account. These cyclical changes in light availability may be obscured by more random wind events that are of greater magnitude. In shallow lakes, when water levels decline, the wind speed required to resuspend sediments is likely to decrease. Consequently, weak periodic wind patterns may have a more pronounced effect on resuspension and light availability.

Lake Alexandrina is a large, coastal lake in South Australia, situated on the River Murray just prior to it entering the Southern Ocean. Due to its shallow morphology, the lake contains high levels of resuspended sediment and is thought to be light-limited for phytoplankton growth under historic water levels (2004). However, under extremely low water levels this relationship between sediment resuspension and light attenuation was shown to have a less pronounced effect on light availability due to the decreased total depth that could be mixed (Chapter 6). The present study aims to determine the frequency of wind-driven resuspension events in Lake Alexandrina, whether there is any periodicity to this resuspension and the extent to which this is affected by declining water levels. This is achieved by using a relationship derived with a three-dimensional hydrodynamic model between wind speed and bed shear stress to show the frequency of wind events that would be of sufficient magnitude to resuspend sediments.
7.2 Methods

7.2.1 Magnitude, frequency and periodicity of wind on Lake Alexandrina

Wind data measured in the northeast of Lake Alexandrina at 6.7 m above the water surface every 15 minutes were provided by DWLBC for the period between 27 October 2009 and 24 March 2010 (for the duration of the sediment trap study discussed in Chapter 5). Longer term, high-resolution (15 minute) wind data was collected at Langhorne Creek between 21 September 2005 and 30 September 2010. Langhorne Creek is situated 18 km northwest of the centre of Lake Alexandrina, with flat topography between. This data set was measured with an automatic weather station (Measurement Engineering Australia, Adelaide), 2 m above the ground. Wind speed from both datasets was corrected to 10 m above the water surface \(U_{10}\) according to equation 7.1 provided in the Lake Analyzer package (GLEON\(^1\)).

\[
U_{10} = U_{\text{measured}} \left( \frac{4.7}{\log \frac{h_{\text{measured}}}{0.0002}} \right) \tag{7.1}
\]

where \(U_{\text{measured}}\) is the wind speed measured at height \(h_{\text{measured}}\).

Both datasets for wind speed were tested for periodicity using modified wavelet analysis scripts for MatLab\(^2\) provided by C. Torrence and G. Compo (Geddes 1984). These scripts convert wind speed time-series data into a time-frequency matrix that enables oscillatory patterns in the wind record to be extracted and tested for significance against randomly generated Brownian noise (Torrence & Compo 1998). The signal can be summed through time to give a cumulative signal which shows whether any distinct periodicity is present in the wind record, and if so, the period of

\(^1\) available online at URL: [http://www.gleon.org/index.php?pr=Science-Technology](http://www.gleon.org/index.php?pr=Science-Technology)

\(^2\) available online at URL: [http://paos.colorado.edu/research/wavelets/](http://paos.colorado.edu/research/wavelets/)
Periodicity and frequency of wind-driven resuspension

its oscillation. Wavelet analysis was chosen over the more widely used Fourier transform because it not only provides information on the periodicity of an oscillation, but also gives an indication of how this frequency varies through time (Torrence & Compo 1998).

Modelled frequencies in wind speed extracted from the wavelet analysis were tested for congruence with measured data using the multiple regression equation for sinusoidal signals linking wind speed ($U_{10}$) against time ($t$) at a given frequency ($\nu$):

$$U_{10} = \alpha_0 + \alpha_1 \cos\left(\frac{2\pi t}{\nu}\right) + \alpha_2 \sin\left(\frac{2\pi t}{\nu}\right) \quad (7.2)$$

Multiple regression output between raw wind speed and the sinusoidal signal was used to estimate the three coefficients, $\alpha_0$, $\alpha_1$ and $\alpha_2$ for each tested frequency. For annual periodicity regressions, wind speed data were smoothed using a one week moving average filter to reduce the effect of short-term variability.

### 7.2.2 Hydrodynamic modeling (ELCOM-CAEDYM)

A three-dimensional hydrodynamic model (ELCOM: Estuary, Lake and Coastal Ocean Model), was coupled to a biogeochemical model (CAEDYM: Computational Aquatic Ecosystem Dynamics Model), both of which were developed at the Centre for Water Research, University of Western Australia (Lau & Weng 1995). This model (ELCOM-CAEDYM) was set up with 0.2 m vertical grid cells and was used to output the orbital velocity (the oscillatory water motion due to wave activity; Hipsey & Hamilton, 2008) at the sediment surface in the middle of Lake Alexandrina (Figure 7.1). The full model has undergone extensive set up, calibration and validation for Lake Alexandrina (Hipsey & Hamilton 2008; Hodges & Dallimore 2010) and the detailed set up of the model has been described elsewhere (Hipsey et al. 2009; Hipsey et al. 2010; Hipsey et al. 2010).
The model was run between 10 February 2010 and 25 March 2010 to overlap the period of field experimentation of Chapter 5 and 6. The model utilised high-resolution (15 minute) meteorological data from Tailem Bend (air temperature, solar radiation, net long wave radiation, relative humidity and rainfall, provided by SA Water) and the northeast of Lake Alexandrina (wind speed and direction, provided by DWLBC). Water temperature and salinity inputs were taken from direct measurements on 10 February 2010 at several locations within Lake Alexandrina (indicated on Figure 7.1). Data on the volume, temperature and salinity of inflows from the River Murray were recorded at Tailem Bend by SA Water and were included in the model run.

Wave generated turbulence modeled as orbital velocity \( u_b \) gave near sediment water velocities that were converted to bed shear stress \( \tau_b \) using equation 7.3

\[
\tau_b = \rho C_D u_b^2
\]

where \( C_D \) is a dimensionless friction coefficient, set at 0.0022 for mud (see Chung et al. 2009 for a recent overview) and \( \rho \) is the density of water equal to 1001 kg m\(^{-3}\). This bed shear was log-transformed and regressed against wind speed to establish the critical wind required to generate wave activity of sufficient magnitude to overcome the critical shear stress for different particle sizes (calculations shown in section 6.2.4). This procedure was repeated with ELCOM-CAEDYM output of \( u_b \) at 5 simulated water levels: +0.75 m AHD, +0.4 m AHD, 0 m AHD (mean sea level), -0.5 m AHD and -0.9 m AHD. In this way, the wind speed required to resuspend sediments in the middle of Lake Alexandrina was related to water level. Furthermore, the frequency of sediment resuspension was estimated as the proportion of time that wind speeds were greater than the critical wind for each simulated water level.
7.2.3 Modelling sedimentation in shallow lakes – a variant of the Courant-Friedrichs-Lewy constraint

A problem can arise for models attempting to simulate the sinking of particles with relatively high settling velocities in shallow lakes. The assumption of mass conservation can only be met by the model if settling particles fall less than the cell depth for each time step (Soulsby 1983), a criterion that can be difficult or impossible to meet in shallow, well-mixed lakes (Jorgensen 1994). This condition is a variant on the Courant-Friedrichs-Lewy constraint (Courant et al. 1928) where

$\frac{\psi \cdot \Delta t}{\Delta z} \leq 1$ \hspace{1cm} (7.4)

for the conservation of mass to be maintained. Here, $\psi$ is the settling velocity of particles, $\Delta t$ is the change in time and $\Delta z$ is the change in depth. In Lake Alexandrina, fine particles of low density may settle at up to $0.056 \text{ m s}^{-1}$ in quiescent conditions as a result of the salinity-induced flocculation of cohesive sediments (Appendix 2). This rate of sedimentation would require a time step of 39 seconds to maintain conservation of mass, provided that the maximum depth of the lake (2.2 m) was used as the depth of a cell. In other words, the maximum time step for a 1-D model is 39 seconds. Using a cell depth of 0.2 m such that the model could capture stratification and changes in orbital velocity with depth, the maximum time step required for the conservation of mass is 3.6 seconds. As a result of this constraint, modelling output was limited to the orbital velocity at the sediment surface and no biogeochemical modelling of sediment particle dynamics was undertaken. Instead, estimates of the critical shear stress for particles of known size and density were compared with the modelled hydrodynamic output to establish the wind speed that would generate an orbital velocity and bed shear stress sufficient to overcome this critical shear stress at different water levels.
7.2.4 Statistical analysis

Wavelet analysis and multiple regressions were undertaken with MatLab (r2009a). Linear and non-linear regressions between wind speed and bed shear stress were performed in GraphPad Prism (Prism Software Inc.). Log-transformed bed shear was related to wind speed using linear regression and any differences between slopes and intercepts under different water levels were tested using analysis of covariance (ANCOVA). An α value of 0.05 was deemed significant for all statistical analysis.

For both datasets of wind speed, a frequency distribution was undertaken in Microsoft Excel 2008 to determine the proportion of time that the lakes were subject to different wind speeds.

ELCOM-CAEDYM output of temperature was validated against high-resolution temperature measurements (15-minutes) in the middle of Lake Alexandrina. Temperature was measured with two thermistors (RBR Ltd. Ottawa, Canada) at the surface and 1.5 metres depth (Chapter 6). The root mean square error (RMSE) between recorded and predicted temperature output was calculated using MatLab (r2008a). Periodicity in the temperature record was tested in the same manner as for wind speed, where a difference between the surface temperature and that at 1.5 metres depth of 1.5°C was used to indicate stratification.

7.3 Results

7.3.1 Wind conditions and temperature stratification

Long term wind records collected at Langhorne Creek showed cycles in wind speed with a 24 hr period ($r^2 = 0.30$, Figure 7.2). An annual periodicity signal was also present ($r^2 = 0.26$) after data were smoothed with a one-week moving average filter. The diel signal peaked at a wind speed of 2.9 m s$^{-1}$ at 1:45 pm, while maximum
seasonal wind speeds occurred between 20 and 22 October (Table 7.1). Wind speeds recorded at Langhorne Creek were below 4 m s\(^{-1}\) for 92% of the time (Figure 7.3) and at or below 1 m s\(^{-1}\) for 39% of the time.

Wind speeds recorded on Lake Alexandrina showed diel periodicity (\(r^2 = 0.13\), Figure 7.4), but the strength of the correlation with a sinusoidal signal was weaker than that at Langhorne Creek. The diel signal for wind speeds recorded on Lake Alexandrina reached a maximum wind speed of 7.7 m s\(^{-1}\) at 6:00 pm, 4.25 hrs after the highest diel wind speed recorded at Langhorne Creek. Wind speed on Lake Alexandrina was below 4 m s\(^{-1}\) for 32% of the time (Figure 7.5). The average wind speed was 5.8 m s\(^{-1}\).

There was a distinct diel periodicity in the temperature difference between the surface and 1.5 metres depth (Figure 7.6) that was highest around midday when solar radiation was at its maximum. This resulted from transient periods of temperature stratification (< 4 hrs) where the surface layer was up to 8.7°C warmer than water at a depth of 1.5 m. There was a second periodic signal in the four-month water temperature record at 44 hours.

### 7.3.2 Modeled orbital velocity at different water levels

ELCOM-CAEDYM represented physical conditions in Lake Alexandrina well (Figure 7.7), with RMSE between modelled and measured surface temperature of 1.57°C and 1.53°C at a depth of 1.5 metres. However, wave-induced orbital velocity data contained multiple spikes that were not present in the wind record and lasted less than an hour. These spikes showed orbital velocities up to 18.9 m s\(^{-1}\), which would correspond to a bed shear stress of 786 N m\(^{-2}\) and where considered a glitch in the model (Hipsey, M. personal communication). Maximum recorded wind speed during the modelled period was 16.3 m s\(^{-1}\), which would induce an orbital velocity of 0.49 m s\(^{-1}\) and a shear stress at the sediment surface of about 0.52 N m\(^{-2}\).
Consequently, the orbital velocity data was filtered to remove all peaks larger than 0.55 N m\(^{-2}\) prior to further analysis. However, even after removing these peaks the relationship between wind speed and bed shear contained high variation in bed shear that was not associated with changing wind speed (Figure 7.8). These generally corresponded to the lower values of the spikes that had been filtered.

To overcome the effect of this noise, a period of prolonged rise and fall in wind speed, where no spikes were present, was used to examine the relationship between wind speed and bed shear (Figure 7.8). This had the additional advantage of minimising the delay caused by temperature stratification between increasing wind speeds and the resulting increase in bed shear stress. The period of persistent wind speeds used was between 7 and 9 March 2010. This period was considered representative of the overall relationship between wind speed and bed shear stress without the influence of spikes and temperature stratification (Figure 7.8).

Each simulated water level showed an exponential relationship between the calculated bed shear stress and wind speed for the period 7 to 9 March 2010 (Figure 7.9). Wind speed explained 89% of the variation in bed shear for the highest simulated water levels (+0.75 m AHD), while all other water levels had a relationship that explained 90% of the variation. This suggested that wind-induced waves were the dominant energy source for bed shear at all water levels tested. ANCOVA confirmed that the relationship between wind speed and bed shear stress was significantly different for all water levels except for the two highest water levels (+0.75 m AHD and +0.4 m AHD, \(p = 0.066\); all other combinations \(p < 0.05\)).

### 7.3.3 Critical wind speed, sediment resuspension and water level

The critical wind speed to overcome the critical shear stress of various particles decreased for each particle size with declining water levels (Figure 7.10). The change was greatest for smaller particles and the slopes of the regressions between water
level and critical wind speed were significantly different for each particle size shown in Figure 6.10 (p < 0.006). For fine sediments with density of 1250 kg m\(^{-3}\) and a diameter of 26 µm, the critical wind speed was 7.7 m s\(^{-1}\) at water levels of +0.75 m AHD and 2.4 m s\(^{-1}\) at water levels of -0.9 m AHD. Conversely, for particles of the same density but a diameter of 285 µm, wind speeds required for resuspension were 14.5 m s\(^{-1}\) and 13.3 m s\(^{-1}\) at +0.75 and -0.9 m AHD, respectively.

7.3.4 Frequency of sediment resuspension

Frequency distribution of wind speeds were combined with the critical wind speed estimates to produce a relationship between water level and the percentage time that conditions were conducive to resuspending particles of a given diameter (Figure 7.11). Once again, fine particles showed the most prevalent change with water levels. The finest particles (diameter = 26 µm) were shown to be resuspended 27% of the time when water levels were +0.75 m AHD, compared with 87% of the time when water levels were -0.9 m AHD. Particles with diameters of 63 µm were resuspended 8.5% of the time at water levels of +0.75 m AHD and 37.6% of the time at -0.9 m AHD. Water level declines affected the resuspension of largest particles least, with particles of diameter 285 µm resuspended 0.19% and 0.98% of the time when water levels were +0.75 and -0.9 m AHD, respectively. ANCOVA showed that the frequency of resuspension was significantly different between all particle sizes simulated (p < 0.004).

7.4 Discussion

7.4.1 Temperature stratification and wind driven sediment resuspension

The relationships between wind speed and resuspension presented here do not account for the influence of temperature stratification on the bed shear stress. When
present, temperature stratification would inhibit the propagation of wind energy through the water column to the sediment, a phenomenon that was shown to delay the resuspension of sediments in Chapter 6 and in other studies (e.g. Ji et al. 2007). The inclusion of this time delay would decrease the frequency of resuspension events shown in Figure 7.11. However, Lake Alexandrina is large, shallow and polymictic, so the overall change in duration of sediment resuspension events would be relatively minor. Although periods of thermal stratification were evident in Lake Alexandrina, stratification events were transient and for a majority of the time relatively weak (Figure 7.6). Perhaps the more significant effect of temperature stratification in delaying resuspension is not on the frequency, but on the timing of sediment resuspension events relative to diurnal solar radiation cycles, as discussed below.

7.4.2 Diel periodicity

Winds at Lake Alexandrina showed diel periodicity. The lower wind speeds at Langhorne Creek were more strongly influenced by this periodicity and the maximum wind speed occurred in the early afternoon. This is consistent with a sea breeze that develops as the land heats faster than the sea during daylight hours (Podsetchine & Huttula 1994; Jin & Sun 2007). Periodicity was weaker for the wind dataset measured on the lake directly, but this may have been a result of the shorter span of available data. Wind speeds on Lake Alexandrina were higher than those measured at Langhorne Creek, with the modelled diel wind signal oscillating between 7.7 m s^{-1} and 4.3 m s^{-1}. For these wind conditions, water levels would need to be below -0.5 m AHD to resuspend sediments with a diameter of 26 µm. These estimates support the findings in Chapter 5 that resuspension occurs with high frequency and approximately daily during low water levels (-0.9 m AHD). However, further work should be conducted on the periodicity of wind records taken on Lake Alexandrina when a longer data set is available. The influence of the choice of wind
dataset on subsequent calculations for benthic light availability should also be examined to improve the predictive capacity of these findings.

The relative timing between diel cycles in wind speed and diurnal temperature stratification determines their interaction and impact on light availability, and hence productivity. Similar interactions between the timing of periodic daily maximum wind speed with diurnal tidal flow have been shown to enhance or minimise resuspension in the shallow, San Fransisco Bay estuary with resulting changes to productivity (Haurwitz 1947). The peak in temperature stratification in Lake Alexandrina occurs around noon at the time of maximum solar radiation. In comparison, wind speeds peak at 6 pm on Lake Alexandrina, but increase throughout the afternoon. Stratification induces buoyancy in the surface mixed layer that must be overcome by wind-induced mixing. This buoyancy is more readily overcome in shallower water where the energy required to mix the water column is lower than for deeper waters. Consequently, lower wind speeds are required to degrade diurnal stratification as water levels decline so mixing and resuspension will occur earlier in the day.

Transient stratification provides optimal conditions for pelagic, buoyant cyanobacteria that can utilise these quiescent conditions to their advantage by maximising their access to light (May et al. 2003). Periods of stratification also disadvantage non-buoyant phytoplankton that settle out of the water column and must rely on resuspension to return them to the euphotic zone. As many cyanobacteria are highly adapted to conditions of low light intensity (Porat et al. 2001) they could also flourish during periods of mixing when sediment resuspension reduces the light availability. Indeed, cell counts revealed that cyanobacteria constituted over 98% of pelagic phytoplankton (Dominic Skinner, unpublished data).
7.4.3 Changing water levels

The frequency and intensity of sediment resuspension events are shown here to be higher at lower water levels in Lake Alexandrina. In addition, water level decline has resulted in an increase in the size of particles susceptible to resuspension. For example, at water levels of +0.75 m AHD, fine sediments with a diameter of 26 µm are resuspended 27% of the time. When water levels are -0.5 m AHD, particles with a diameter of 63 µm are resuspended with the same frequency. These results support the idea postulated in Chapter 4 that fine sediments are being more readily redistributed throughout the lake at lower water levels and therefore have a stronger signal in sediments collected from peripheral sites.

7.4.4 Water level declines and water clarity restoration

Simulations for Lake Alexandrina showed that the frequency of sediment resuspension and the size of sediments being resuspended increased with declining water levels. This suggests that in an already turbid system without macrophyte cover, declining water levels will exacerbate sediment resuspension. More resuspension means that the total surface area of suspended particles scattering or absorbing incoming light increases as a result (Reynolds 1997). This reduces the depth of light penetration into the water column, but does not have the same effect on the average light irradiance through the water column as the maximum mixing depth decreases with lower water levels (Chapter 6). As a result, increased sediment resuspension from water level declines may affect benthic primary producers more than it does those in the pelagic zone. The diel patterns of transient temperature stratification and wind speeds described earlier is likely to further promote the development of buoyant pelagic photosynthetic organisms over those in benthic regions and non-buoyant phytoplankton species.
In small shallow lakes, decreasing the water level has been used as a management strategy to increase the light availability to benthic macrophytes and microalgae (Davies-Colley & Smith 2001). By increasing the abundance of benthic macrophytes and microalgae, sediment stability increases, wave activity is dampened and resuspension lessened (Coops & Hosper 2002; van Geest et al. 2007; van Wichelen et al. 2007).

Large shallow eutrophic lakes present a particularly difficult challenge to lake restoration measures because the frequency of sediment resuspension and intensity of turbulence at the sediment surface prevent the recolonisation of macrophytes and maintain highly turbid conditions even after significant reductions to external nutrient input (Horppila & Nurminen 2001; Madsen et al. 2001; James et al. 2004; Huang et al. 2007; Li et al. 2008). Internal nutrient loading (Chapter 3) coupled with a high amount of sediment resuspension in Lake Alexandrina suggests that reductions in external nutrients alone may not prove sufficient to restore water clarity and macrophyte cover to Lake Alexandrina. Similarly, Bachmann et al. (1999), suggested that resuspended particles in Lake Apopka, USA, a large (124 km$^2$), shallow ($Z_{\text{mean}} = 1.7$ m) eutrophic system, would prevent a shift to a clear water state even if nutrient reduction measures were implemented. This was because turbidity resulted from resuspended particles, not nutrient supported phytoplankton, so would not be prevented by lower nutrient inputs. However, when coupled with macrophyte replanting and the biomanipulation of fish stocks, nutrient reductions in Lake Apopka did improve water clarity and led to a decrease in planktonic biomass, despite no significant shift in the wind conditions (Lowe et al. 2001).

Engineering solutions have been implemented in other large shallow lakes where nutrient reductions are unlikely to be sufficient to overcome the mechanisms that reinforce turbid conditions. For instance, restoration measures in the very large (2338 km$^2$), shallow ($Z_{\text{mean}} = 1.9$ m) Lake Taihu, the third largest freshwater lake in China, utilised a wave barrier 3.3 km long to reduce local sediment resuspension and TP
concentrations to 36% and 39%, respectively of values prior to the construction of the barrier (Hilt et al. 2006). Alternatively, localised deep holes were dug into sediments of Lake Loosdrecht, in the Netherlands, with the aim of creating quiescent conditions that would remove fine particles of low density from exposure to frequent resuspension (Huang & Liu 2009). This method reduced light attenuation from 2.5 m\(^{-1}\) to 2.2 m\(^{-1}\) and particle loads in the water column by 25% (Penning et al. 2010).

Lake Alexandrina is further degraded by the accumulation of fine sediments due to extensive river regulation and artificially maintained lake water levels. These fine sediments are readily resuspended and provide a poor quality substrate for macrophyte colonization (Barko & Smart, 1986). This study has shown that a large manipulation to water level is unlikely to improve water clarity in Lake Alexandrina because of its already degraded state. As a result, restoring water clarity to Lake Alexandrina would require a suite of measures in addition to nutrient reductions, such as the construction of breakwaters or sheltered regions to enable macrophytes to colonise, biomanipulation of food webs, and the active planting of macrophytes.
Periodicity and frequency of wind-driven resuspension

Table 7.1: Multiple regression output for a sinusoidal signal fitted to wind periodicity data. Annual periodicity calculations had data smoothed with a one-week moving average filter prior to regression.

<table>
<thead>
<tr>
<th>Site</th>
<th>Period</th>
<th>$r^2$</th>
<th>Maximum</th>
<th>Minimum</th>
<th>$\alpha_0$</th>
<th>$\alpha_1$</th>
<th>$\alpha_2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langhorne Creek</td>
<td>12 hrs</td>
<td>0.025</td>
<td>2:15 am/pm</td>
<td>7:45 am/pm</td>
<td>1.74</td>
<td>0.11</td>
<td>0.31</td>
</tr>
<tr>
<td></td>
<td>24 hrs</td>
<td>0.30</td>
<td>1:45 pm</td>
<td>1:45 am</td>
<td>1.74</td>
<td>0.94</td>
<td>0.67</td>
</tr>
<tr>
<td></td>
<td>Annual</td>
<td>0.26</td>
<td>20-22 Oct</td>
<td>20-22 Apr</td>
<td>1.74</td>
<td>3.66</td>
<td>0.00034</td>
</tr>
<tr>
<td>Northeast of Lake Alexandrina</td>
<td>24 hrs</td>
<td>0.13</td>
<td>6pm</td>
<td>6am</td>
<td>5.84</td>
<td>0.02</td>
<td>-1.5</td>
</tr>
</tbody>
</table>
Figure 7.1: Location of input parameters used to set up a three-dimensional hydrodynamic model (ELCOM-CAEDYM) for Lake Alexandrina. The open circles show sites where water quality measurements (temperature and salinity) were taken on 10 February 2010. The sources of wind data are indicated with a ‘W’. The black filled circle shows the site of output data for modelled bed shear stress.
Figure 7.2: Wavelet analysis of long-term wind data from Langhorne Creek. A: Raw wind speed data (in m s$^{-1}$); B: isopleths of wind speed periodicity through time; and C: the cumulative strength of the periodicity signals from the wavelet analysis.
Figure 7.3: Cumulative frequency distribution of long-term wind record at Langhorne Creek.
Figure 7.4: Wavelet analysis of wind data from the northeast of Lake Alexandrina. A: Raw wind speed data; B: isopleths of wind speed periodicity through time; and C: the cumulative strength of the periodicity signals of the wavelet analysis.
Figure 7.5: Cumulative frequency distribution of wind speed measured on the northeast of Lake Alexandrina.
Figure 7.6: Periodicity in temperature stratification in the middle of Lake Alexandrina. A: temperature difference between surface and bottom (recorded at 1.5 metres depth), B: wavelet power spectrum of temperature stratification, C: the cumulative strength of the periodicity signals of the wavelet analysis.
Figure 7.7: Meteorological inputs and temperature outputs of ELCOM-CAEDYM hydrological model for Lake Alexandrina. Upper two plots show meteorological inputs of solar radiation (A) and wind speed (B), while the lower two plots show the surface (C) and bottom (D) temperatures output by ELCOM-CAEDYM and measured in situ. RMSE is the root mean square error.
Figure 7.8: Relationship between wind speed and the bed shear stress derived from modelled orbital velocity in the middle of Lake Alexandrina. The full dataset (black dots) shows spikes in bed shear stress that are not influenced by wind speed. The relationship for the period between 7 and 9 March 2010 is also shown (grey dots).
Figure 7.9: Non-linear regressions of wind speed and modelled bed shear at different water levels. Data used are from 7 March to 9 March 2010.
Figure 7.10: Estimates of the critical wind speed required to resuspend sediment at different water levels. All particle sizes shown have a density of 1250 kg m$^{-3}$. $\tau_{cr}$ is the calculated critical shear stress to resuspend particles of diameter, D, as calculated in chapter 6.
Figure 7.11: Frequency of sediment resuspension under different water levels. \( \tau_{cr} \) is the calculated critical shear stress to resuspend particles of diameter, \( D \), as calculated in Chapter 6.
8. Discussion

8.1 Effects of water level decline in shallow lakes

8.1.1 Water use, climate change and water level decline

Lake Alexandrina underwent an unprecedented decline in water level that began in the austral summer of 2006-07, continuing into 2010, until floods increased flow in the River Murray and refilled the lake. The mean depth in Lake Alexandrina declined from 2.9 m to 1.3 m, with up to 70% of the lake volume lost. This provided a unique opportunity to examine the effects of water level decline on lake water quality. A specific focus of the research was to determine the role of water level decline on sediment resuspension and its effect on water clarity. The changes to water clarity during water level declines were used to infer potential changes to ecosystem state.

Water level declines in Lake Alexandrina resulted from a prolonged drought in the Murray-Darling Basin (Penning et al. 2010), with a 1 in 1500 year chance of natural occurrence (Leblanc et al. 2009). Indications are that the drought was exacerbated by climate change that shifted rainfall patterns and decreased associated run-off (Gallant & Gergis 2011). Additionally, upstream water extractions during the drought reduced flows to the lower reaches of the Murray-Darling Basin (Murphy & Timbal 2008; Timbal 2009; Kamruzzaman et al. 2011).
The effect of water level fluctuations on the water quality, biota, physical habitat and ecosystem structure in rivers and lakes has only received relatively recent attention (Crase & O'Keefe 2009; Lee & Ancev 2009; Paton et al. 2009). Studies examining the effects of extreme water level decline are uncommon (Leira & Cantonati 2008). As the combined effect of climate change, population growth and continued water resource development on water availability continue to increase the impacts of water level decline on lakes is expected to become increasingly common (Nõges & Nõges 1999; Baldwin et al. 2008; Zohary & Ostrovsky 2011). There are a number of other examples of extreme water level decline that highlight the combined threats of water extraction, climate change and drought. In Asia, shifting climate patterns and unsustainable extraction of freshwater for irrigation from the Amu Darya and Syr Darya tributaries have limited water inflows and resulted in an 80% reduction in volume of the Aral Sea (Wetzel 1992; Postel 1998; Milly et al. 2008). Lake Chad in North Africa parallels with Lake Alexandrina in that both are large, shallow lakes that receive the majority of inflows from a single river and are positioned at the end of their drainage basin. Declining catchment rainfall, and increased water extraction have restricted the surface area of Lake Chad to 5% of its former coverage (Stone 1999). A better understanding of the effects of water level decline on Lake Alexandrina, and shallow lakes in general, will improve our ability to manage changes associated with adverse water shortages as they occur.

8.1.2 Water level decline influences stable ecosystem states

The ecological theory of alternative stable states, while obviously a simplification of the multiple complex processes and feedbacks within shallow lakes (Coe & Foley 2001), proves a useful conceptual model for determining their responses to external disturbances (Scheffer & van Nes 2007). The interactions between water level decline and ecosystem state are influenced by factors including basin morphology,
the composition, distribution and abundance of vegetation, nutrient cycling, sediment resuspension and trophic interactions (Davis et al. 2003; Jeppesen et al. 2007).

For Lake Alexandrina, evidence suggests that a shift in state, from clear to turbid, occurred sometime in the early twentieth century, when land-use changes and river regulation altered the ecosystem functioning (Jeppesen et al. 1999; Scheffer & Jeppesen 2007; Scheffer & van Nes 2007). A transient clear water state was reported during drought in 1981 (Herczeg et al. 2001; Sim & Muller 2004) and a similar biophysical response was observed during the drought discussed in this study for the lower River Murray (upstream of Lake Alexandrina). Here, water clarity increased (Mosley et al. In prep) and macrophytes recolonised shallow sandy substrate where they had not been observed for many years (George Ganf, personal communication). In contrast, over the same period, Lake Alexandrina became increasingly turbid during water level decline with increased total nutrients, chlorophyll and suspended particulate matter.

The divergent response to drought of Lake Alexandrina and the lower River Murray can be explained by different responses to nutrient loadings. During drought periods, external nutrient inputs from catchment runoff tend to decrease (Geddes 1988), leading to a transient clear water state in many shallow lakes (Bond et al. 2008). However, water level decline can shift internal nutrient cycles towards an increase in total nutrient loads (Zohary & Ostrovsky 2011). If these changes to internal nutrient loading from water level decline are larger than the reduction in external nutrient supply, then lake waters will become eutrophied. Lower flows resulted in increased sedimentation, decreased turbidity and lower nutrient concentrations in the river (Mosley et al. In prep), whereas increased sediment resuspension led to higher nutrients and turbidity in Lake Alexandrina (Chapter 3).

The vegetation structure and basin morphology may also have influenced the decrease in water clarity during water level decline in Lake Alexandrina. This is in
contrast to other large, shallow lakes that have had increased water clarity resulting in higher macrophyte abundance and distribution as a result of water level decline (van Geest et al. 2007). During high water levels in the eutrophic Lake Okeechobee, Florida, light penetration to benthic sediments is low, minimising macrophyte establishment. The low abundance of macrophytes leads to increased chlorophyll concentrations and transport of fine mud particles from the pelagic zone towards the littoral zone (Wallsten & Forsgren 1989; Havens et al. 2004) creating a positive feedback loop of increased turbidity. However, drought-induced water level decline in Lake Okeechobee resulted in an increased distribution of submerged macrophytes as incident light on littoral sediments increased (Jin et al. 2000; Havens et al. 2004). The stabilisation effect of macrophytes decreased the resuspension and redistribution of fine sediments, reinforcing the shift to a clear water state.

For Lake Alexandrina, the barrages constructed in 1940 maintained static water levels in the lake that were artificially high for 67 years prior to the recent extended period of water level drawdown. The static water levels constricted the distribution of submerged macrophytes to a narrow band (~25 m) around the littoral region of the lake (Havens et al. 2004). Water level decline can lead to major changes in lake ecosystem structure and function if littoral macrophytes are disconnected from the lake with a receding shoreline (Walker et al. 1994; Blanch et al. 1999). The extent of this effect is mediated by the rate of drawdown compared with the rate that macrophytes can recolonise newly formed littoral zone (Furey et al. 2006). As the shoreline of Lake Alexandrina receded, aquatic plants became disconnected from the water and were lost completely from 2009 onwards (Coops et al. 2003; Turner et al. 2005; Zohary & Ostrovsky 2011). Consequently, stabilisation of sediments by increasing the abundance of macrophytes did not occur in Lake Alexandrina during this study. The effect was increased turbidity from higher resuspension and the redistribution of sediments throughout the lake (see Chapters 4 & 5).
8.1.3 Influence of drawdown on the species distribution of primary producers in shallow lakes

The loss of macrophytes from Lake Alexandrina after 2009 resulted in primary production being partitioned between phytoplankton and benthic microalgae. Consequently, lake productivity was influenced by hydrodynamic factors that change during water level decline and influence mixing regime, light availability and nutrient concentrations. These physical and chemical drivers have a strong effect on the phytoplankton species distribution and are discussed below.

Lower water levels correspond with a smaller lake volume and a consequent decreased thermal mass that gains and loses heat faster than deeper water (Gehrig et al. 2011). Warmer diurnal water temperatures improve conditions for cyanobacteria, which generally have an optimal growth rate at higher temperatures than eukaryotic phytoplankton (Wells & Sherman 2001).

The influence of water temperature on primary production also interacts with nutrient concentration (Jöhnk et al. 2008; Pael 2009), where higher temperatures amplify productivity when nutrients are in oversupply, but decrease productivity when nutrients are limited. In Lake Alexandrina, where total nutrients are abundant, but often undetectable in a soluble and bioavailable form (Tadonléké 2010), increased temperatures from water level decline could restrict primary production. However, nutrient limitation by oxidised nitrogen is common in Australian freshwaters (Geddes 1984) and in Lake Alexandrina (Harris 2001) and could be overcome by nitrogen-fixing cyanobacteria.

Changing light availability influences primary productivity of pelagic and benthic autotrophs as water levels decline. At low water levels, higher irradiance at the benthos can increase productivity (Cook et al. 2009). However, increases in turbidity will impact on productivity at the benthos more than phytoplankton productivity, which increases from mixing through waters of higher irradiance (Anthony et al.
2004; Lawson et al. 2007). The diurnal patterns of temperature stratification and sediment resuspension influenced by the afternoon sea breeze in Lake Alexandrina provide favorable conditions for buoyant phytoplankton. While sediment resuspension can return settled algal cells into suspension and increase productivity during mixing (Ferris & Christian 1991; Prézelin et al. 1991), intermittent stratification results in the sinking of non-buoyant phytoplankton (Carrick et al. 1993; Schelske et al. 1995; Schallenberg & Burns 2004). Additionally, lower water levels decrease the sedimentation time required to clear the water column of suspended particles after quiescent conditions are established. This results in increased irradiance to the surface mixed layer during stratification that can enhance the dominance of buoyant cyanobacteria (Sherman et al. 1998).

In Lake Alexandrina, physical interactions between holomixis and transient stratification, coupled with increased water temperatures and low concentrations of bioavailable nutrients provided ideal conditions for cyanobacterial development during low water levels. Throughout 2009, multiple species of cyanobacteria dominated phytoplankton cell counts in Lake Alexandrina (Table 8.1). In other shallow lakes exposed to water level decline, cyanobacteria also dominated species assemblages at the lowest water levels for reasons outlined above (Porat et al. 2001). Lake Alexandrina was also the site of one of the earliest recordings of toxic cyanobacterial blooms of Nodularia spumigera in 1878, during low water levels and high water temperatures (Tryfon & Moustaka-Gouni 1997; Nõges et al. 2003; Romo et al. 2004; Beklioglu et al. 2007).

### 8.2 Resuspension and redistribution of sediments in large, shallow lakes

Fine sediments became more prevalent in the littoral sediments of Lake Alexandrina as water levels decreased (Chapter 4). This finding is in contrast to other studies,
where fine sediments are often eroded from littoral sediments during drawdown and become focussed in deeper water (Codd et al. 1994). Water level declines in shallow lakes can increase the amount of sediment resuspension (Gottgens 1994; Effler & Matthews 2004; Furey et al. 2004), which has implications for sediment distribution, water quality and the ecological character of a lake.

Sediment distribution in a lake is a function of lake morphology, the depth of sediment undergoing resuspension, the horizontal mixing patterns of the lake, and the rate of particle sedimentation (Gibson & Guillot 1997; Håkanson et al. 2000; Nagid et al. 2001; Effler & Matthews 2004; Shantz et al. 2004). Lakes with a maximum depth that allows sediments to accumulate and remain relatively undisturbed should show evidence of increased sediment focussing during water level decline as the newly formed littoral sediments are scoured more frequently. Conversely, those lakes with a shallow morphology and small maximum depth will not provide a sheltered region for sediments to accumulate so that water level declines expose all sediments to greater shear stress. The relative difference between maximum depth and that of littoral sediments, as well as other morphological characteristics, are likely to dictate the extent of sediment focusing (Hilton 1985; Håkanson 2005).

The dynamic ratio (DR) of a lake (√Area/Z_{mean}) gives an indication of the amount of sediment that is exposed to forces that resuspend sediments (Whitmore et al. 1996). When DR is above 4, all surface sediments would be exposed to resuspension. This corresponds to all bottom sediments being classified as ‘erosion’ sediments under the framework proposed by Håkanson and Jansson (Hilton 1985; Håkanson 2005). For Lake Alexandrina, the DR increased from 8.3 to 15.9 as water levels declined from 0.75 m AHD to -0.9 m AHD, respectively. During low water levels, the DR and mean concentration of suspended particulate matter in Lake Alexandrina were comparable to a larger lake, such as Lake Okeechobee in Florida, USA (Area: 1750 km\(^2\), Z_{mean}: 2.7 m; Table 8.2). However, the DR of Lake Alexandrina suggests that,
irrespective of water levels, all sediment is expected to undergo resuspension, which would not explain the different distribution patterns observed during water level declines.

Shallow lakes are thought to undergo random redistribution of surface sediments (Håkanson & Jansson 1983), but resuspension events occur with relatively low frequency (Hilton et al. 1986; Blais & Kalff 1995). For example, over 70% of sediments in a small shallow lake ($Z_{\text{max}}$: 1.7 m) were shown by Carper and Bachman (Evans 1994) to undergo resuspension less than 0.1% of the time during summer. Conversely, in Lake Alexandrina at the lowest water levels, sediment resuspension occurred at relatively low wind speeds ($< 4$ m s$^{-1}$, Figure 6.3) at the maximum depth of the lake. At low water levels, fine sediments were frequently resuspended (63 µm particles: 38% of the time; 26 µm particles: 87% of the time; Figure 7.11).

As sediments in sheltered areas become exposed to increased shear stresses during water level declines, they are resuspended more frequently. Fine particles are a dominant proportion of resuspended sediments in shallow lakes (1984) and have an increased residence time in the water column (Hamilton & Mitchell 1996). This suggests that fine sediments have a disproportionate impact on suspended particle load, light climate and distribution patterns. Furthermore, fine particles are strongly influenced by horizontal water circulation patterns, such as wind driven transport, the coriolis effect (Kozerski 1994) and baroclinic pumping (Csanady 1975) that are dominant in shallow polymictic lakes where diurnal mixing reduces the influence of internal waves. This suggests that for large, shallow lakes, the frequency of resuspension events is an important factor in mapping sediment distribution patterns.

For Lake Alexandrina, the increased frequency of sediment resuspension at low water levels (Chapter 7) exposed more fine sediment to horizontal transport that resulted in more random deposition patterns than during higher water levels. Fine particles (4 – 20 µm diameter) were more prevalent in inundated littoral sediments following water level decline (Table 4.3).
Discussion

There has been very little research on sediment distribution in shallow lakes with surface area above 100 km$^2$ (Figure 8.1). Of these larger lakes, to the author’s knowledge, only two have had sediment distribution studied. Lough Neagh (area: 383 km$^2$, $Z_{\text{max}}$: 24 m, red cross on Figure 8.1), in Northern Ireland, showed random distribution of sediments (Rueda & Schladow 2003). Lake Võrtsjärv (area: 270 km$^2$, $Z_{\text{max}}$: 6 m), in Estonia, had highest sedimentation on one side of the lake that was deeper and leeward of the prevailing winds (Douglas & Rippey 2000). Consequently, this thesis presents the first empirical evidence of the effects of water level decline on sediment distribution in a large, shallow lake.

8.3 Implications for the management of Lake Alexandrina

So far, the causes of high turbidity have been discussed as resulting from sediment resuspension, which is clearly a dominant process in Lake Alexandrina. However, there are other causal factors to be considered in conjunction with sediment resuspension. Static water levels constricted the distribution of macrophytes, led to shoreline erosion and the accumulation of fine sediments. The ecosystem effects of the introduction of the Common Carp ($Cyprinus carpio$), a benthivorous fish, into Lake Alexandrina in the 1970s are not known (Nõges et al. 1999). However, they have been shown to reduce the abundance of macrophytes and macroinvertebrates in other shallow lakes (Paton 2010), suggesting their introduction may have altered the ecological structure of Lake Alexandrina. Increased dominance of phytoplankton (exacerbated by water level declines in Lake Alexandrina) promotes the accumulation of fine, organic-rich sediments that remain unconsolidated, are readily resuspended, provide poor substrate for macrophyte colonization, and hence act as a positive feedback to high turbidity (Miller & Crowl 2006; van de Haterd & Ter Heerdt 2007). This change was observed in Lake Alexandrina with an increase in organic matter in profundal sediments over the period of water level decline (Table 4.1).
The decline in macrophyte abundance and distribution of Lake Alexandrina is considerable when compared to historical evidence (Bachmann et al. 1999; Schutten et al. 2005; van Wichelen et al. 2007). Given that this extensive macrophyte coverage supported numerous species of fish and birds in large numbers (Bourman & Barnett 1995; Herczeg et al. 2001; Sim & Muller 2004; Marsland & Nicol 2009; Paton 2010; Gehrig et al. 2011), substantial improvements to the ecology of Lake Alexandrina are clearly possible and should be a primary management focus. The extreme perturbation to the biophysical dynamics, in the form of water level decline, can be used to highlight some lake characteristics that can provide guidance to managers during post-drought recovery and restoration efforts.

The role of water level fluctuations deserves attention as increases in water level can be equivalent to a reduction in trophic level (Paton 2010) and influence macrophyte abundance (Nõges & Nõges 1998). As the initial step in the post-drought recovery process, changes to the management of Lake Alexandrina will advocate the reintroduction of seasonal water level fluctuations to allow the distribution of macrophytes in the littoral region to expand (van Geest et al. 2007). This means that water levels will, on average, be lower than before the drought, fluctuating around 0.4 m AHD. Managing Lake Alexandrina at lower water levels will reduce shoreline erosion (DEH 2010) that contributes significantly to increased sediment loads in the lake (Lamontagne et al. 2004).

At these lower water levels, the wind speed required to resuspend fine sediments from the maximum depth of Lake Alexandrina will be about 6 m s\(^{-1}\) (Figure 7.10). This is at the top of the range of average wind speeds (4 - 6 m s\(^{-1}\)) that induce resuspension and have delayed or prevented other lakes from returning to a clear-water state following oligotrophication (Barnett 1994; Bourman & Barnett 1995; Fluin et al. 2007). The shallow depth of Lake Alexandrina means that even in a highly turbid state, over 11% of bottom sediment would have access to light, with a euphotic depth of 0.68 m (Figure 6.8). This suggests that light limitation is unlikely
to prevent initial macrophyte re-colonisation and any incremental decreases in turbidity would increase the area of substrate with sufficient light availability.

Water level manipulation has received more attention than nutrient reductions in management plans for Lake Alexandrina (Van Liere & Gulati 1992; Arfi & Bouvy 1995). Increased riparian revegetation is currently being used to enhance the resilience of the lake (Lamontagne et al. 2004). This may affect the input of nutrients from the local catchment that have previously been considered as more important than fluvial inputs from the River Murray, although data for this hypothesis is scarce (DEH 2010). Cook et al. (DEHAA 2000) showed a strong relationship between fluvial phosphorus input and changes to lake concentrations, which suggests that the River Murray (and local rivers) provide the vast majority of nutrients into Lake Alexandrina. Consequently, riparian revegetation, while providing ecological benefits in its own right, is unlikely to alter the condition of the lake ecosystem through any substantial reductions to the nutrient input into Lake Alexandrina.

Nutrient reductions are commonly the first stage of bioremediation in shallow, turbid lakes (2009). In Lake Alexandrina, extreme water level decline was required to increase turbidity and offset drought and declining nutrient inputs that would otherwise decrease turbidity (Jeppesen et al. 2003; Jeppesen et al. 2005; Beklioglu et al. 2007; Gulati et al. 2008). This suggests that the importance of eutrophication in maintaining a turbid state should not be underestimated and warrants further investigation. Suspended particles in Lake Alexandrina constituted both organic and inorganic sediments. During water level decline, the proportion of organic material in surface sediments increased, the nutrient composition of surface sediments shifted from inorganic to organic and the concentration of resuspended particles and chlorophyll increased. These factors all suggest that turbidity was maintained, in part, by phytoplankton that decreased light irradiance through the water column and increased the proportion of unconsolidated, fresh material of lower density that was more easily resuspended. High levels of resuspension of this ‘fluid mud’ has been
postulated to prevent a return to clear water in the large, shallow Lake Apopka (area 124 km²; \(Z_{\text{mean}}\) 1.7 m; (when compared with Geddes 1988). However, subsequent studies showed nutrient reductions to be effective in improving water quality despite these unconsolidated sediments in Lake Apopka (Bachmann et al. 1999) and in larger lakes of similar depth (Lowe et al. 2001).

Turbidity plays a key role in ecosystem processes in Lake Alexandrina (Havens et al. 2004), however the factors controlling turbidity levels have received much less attention. Significant inflows sourced from the highly turbid Darling River are known to increase turbidity in Lake Alexandrina (Geddes 1988; Lamontagne et al. 2004). This thesis has demonstrated that frequent sediment resuspension maintains suspended particle loads with a high proportion of fine sediments that are readily transported throughout the lake. This turbidity, contributes to the dominance of phytoplankton, especially cyanobacteria, in the shallow lake through mechanisms such as the shading of the benthos and suppression of macrophyte growth. However, this is a cyclical feedback mechanism, with the dominance of phytoplankton also a strong contributor to lake turbidity. As a result, nutrient reductions (coupled with the more natural water level fluctuations already planned) are likely to improve water clarity, increase the abundance of macrophytes and contribute to the restoration of the ecological character of Lake Alexandrina. These nutrient reductions could be achieved through changing land-use practices and bio-manipulation methods, for example by the removal of the exotic Common Carp from Lake Alexandrina.
Table 8.1: Seasonal variation in the proportion of algal species and functional groups for phytoplankton in Lake Alexandrina during 2009. Samples were collected in the middle of Lake Alexandrina 3 times each season, preserved in Lugol’s and counted by the Australian Water Quality Centre, South Australia.

<table>
<thead>
<tr>
<th>Season</th>
<th>Dominant species</th>
<th>Genus/Species</th>
<th>(cells/mL)</th>
<th>Type</th>
</tr>
</thead>
<tbody>
<tr>
<td>Summer</td>
<td>Anabaenopsis</td>
<td>(2600)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Ankistrodesmus</td>
<td>(3750)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aphanizomenon</td>
<td>(62800)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aphanocapsa</td>
<td>(51200)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Crucigenia</td>
<td>(4250)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Oocystis</td>
<td>(2350)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Planktolyngbya subtilis</td>
<td>(2260000)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td>Autumn</td>
<td>Ankistrodesmus</td>
<td>(2100)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aphanizomenon</td>
<td>(10700)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aphanocapsa</td>
<td>(278000)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Nodularia spumigena</td>
<td>(1120)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Oocystis</td>
<td>(1950)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Planktolyngbya subtilis</td>
<td>(1170000)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Pseudanabaena</td>
<td>(70200)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Staurosira</td>
<td>(1850)</td>
<td>Diatom</td>
<td></td>
</tr>
<tr>
<td>Winter</td>
<td>Ankistrodesmus</td>
<td>(3600)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aphanocapsa</td>
<td>(138000)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Oocystis</td>
<td>(3500)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Planktolyngbya spp.</td>
<td>(669000)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Scenedesmus</td>
<td>(1300)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td>Spring</td>
<td>Aphanocapsa</td>
<td>(132000)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Oocystis</td>
<td>(4900)</td>
<td>Green</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Planktolyngbya spp.</td>
<td>(437000)</td>
<td>Cyanobacteria</td>
<td></td>
</tr>
</tbody>
</table>
Table 8.2: Comparison of parameters relating to sediment resuspension in large, shallow lakes compared to Lake Alexandrina at different stages of the water level drawdown.

<table>
<thead>
<tr>
<th>Lake, Country</th>
<th>Area</th>
<th>Mean Depth</th>
<th>Dynamic Ratio</th>
<th>SPM</th>
<th>Resuspension Frequency</th>
<th>Critical shear stress</th>
<th>Method</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>km²</td>
<td>m</td>
<td>(√Area/Z&lt;sub&gt;mean&lt;/sub&gt;)</td>
<td>mg L&lt;sup&gt;−1&lt;/sup&gt;</td>
<td>g m&lt;sup&gt;−2&lt;/sup&gt; d&lt;sup&gt;−1&lt;/sup&gt;</td>
<td>%</td>
<td>N m&lt;sup&gt;−2&lt;/sup&gt; unless otherwise specified</td>
<td></td>
</tr>
<tr>
<td>Lake Arreso, Denmark</td>
<td>41</td>
<td>3</td>
<td>2.1</td>
<td>42</td>
<td>300</td>
<td>50</td>
<td>-</td>
<td>SPM &amp; modeling (Geddes 1988)</td>
</tr>
<tr>
<td>Lough Neagh, Northern Ireland</td>
<td>383</td>
<td>8.9</td>
<td>2.2</td>
<td>36*</td>
<td>50.9</td>
<td>-</td>
<td>-</td>
<td>Traps (Kristensen &lt;i&gt;et al.&lt;/i&gt; 1992)</td>
</tr>
<tr>
<td>Lake Võrtsjärv, Estonia</td>
<td>270</td>
<td>2.8</td>
<td>5.9</td>
<td>20 – 40</td>
<td>145 – 163</td>
<td>-</td>
<td>-</td>
<td>Traps &amp; mass balance (Douglas &amp; Rippey 2000)</td>
</tr>
<tr>
<td>Lake Markermeer, Netherlands</td>
<td>680</td>
<td>3.6</td>
<td>7.2</td>
<td>45</td>
<td>~1000</td>
<td>-</td>
<td>0.5 – 0.7 cm/s (water velocity)</td>
<td>Traps &amp; sediment cores (Nõges &lt;i&gt;et al.&lt;/i&gt; 1999)</td>
</tr>
<tr>
<td>Lake Alexandrina (+0.75 m AHD)</td>
<td>580</td>
<td>2.9</td>
<td>8.3</td>
<td>81</td>
<td>4</td>
<td>-</td>
<td>-</td>
<td>Water sampling &amp; modeling (Chapter 7) (Kelderman &lt;i&gt;et al.&lt;/i&gt; 2011)</td>
</tr>
<tr>
<td>Lake Okeechobee, USA</td>
<td>1730</td>
<td>2.7</td>
<td>15.5</td>
<td>70 – 200</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Water sampling (Mosley &lt;i&gt;et al.&lt;/i&gt; In Prep)</td>
</tr>
<tr>
<td>Lake Alexandrina (-0.9 m AHD)</td>
<td>429</td>
<td>1.3</td>
<td>15.9</td>
<td>105</td>
<td>1044</td>
<td>0.03 (26 µm)</td>
<td>Traps, water sampling &amp; modeling (Chapters 3,5,6,7)</td>
<td></td>
</tr>
<tr>
<td>Lake Taihu, China</td>
<td>2338</td>
<td>1.9</td>
<td>25.4</td>
<td>40</td>
<td>189 - 2507*</td>
<td>0.03 – 0.04</td>
<td>Traps &amp; Ex-situ measurement (Jin &amp; Sun 2007)</td>
<td></td>
</tr>
</tbody>
</table>

* Estimated for a maximum sedimentation rate of 100 g m<sup>−2</sup> d<sup>−1</sup> following each of three resuspension events over a fortnightly sampling period, see Douglas and Rippey (2000).

† Maximum values were recorded as a typhoon passed near Lake Taihu, see Huang and Liu (2009).
Figure 8.2: Area and maximum depth of lakes tested for sediment focussing (squares) and those shown in Table 8.2 (crosses). Area and depth of lakes tested are from Blais and Kalff (see text, Geddes 1988), Effler and Matthews (1995), Furey et al. (2004), Gottgens (2004), Hilton (1994), and Punning et al. (1985). Arrow indicates the change in position of Lake Alexandrina from 2007 when water levels were high to 2009 during low water level. The red cross is Lough Neagh in Northern Ireland that was shown to undergo random redistribution of sediments by Douglas and Rippey (2006).
A.1. Method selection when estimating the proportion of settling particles attributable to sediment resuspension

Three methods of estimating the proportion of trapped particulate matter (TPM) that results from sediment resuspension are applicable to shallow lakes. They are introduced and compared below using data presented in the sediment trap study for Lake Alexandrina (Chapter 5).

A.1.1 The label method

This method uses some label substance that is present in the sediment, TPM and suspended particulate matter (SPM) to estimate the proportion of TPM associated with sediment resuspension (R) according to equation A.1.1 from Gasith (Huang & Liu 2009)

\[ R = TPM \left[ \frac{f_S - f_T}{f_R - f_T} \right] \]  

(A.1.1)

where \( f_S \) is the label fraction in TPM, \( f_R \) is the label fraction in the surface sediments and \( f_T \) is the label fraction in SPM. This method is considered to be valid if the label substance is present in a significantly different proportion in the sediment compared to SPM (1975). The most commonly used label is the organic fraction present in each of the three groups being compared (Blomqvist & Hakanson 1981).
The label method is able to resolve spatial variations between sites, within certain limitations, as it provides an estimate for each trap or set of traps. However, comparing some label component of local sediments with the corresponding label in trap contents assumes that the effect of horizontal transport of particles on the fraction of label substance caught in the trap is negligible. Additionally, by assuming that the fraction of label component of the surface sediment is representative, necessitates that surface sediment is resuspended uniformly. If the surface sediment is not homogenous, and the fraction of label component differs by particle size or density, then distinct components of the sediment can be preferentially resuspended, which will affect the validity of the calculation.

One-way ANOVA (JMP-IN, SAS Institute) was used to show that the organic fraction of sediments was significantly different than that of SPM for all samples collected during the sediment trap study (Chapter 5). The values of R calculated with the label method at each site were averaged across each deployment period to give a lake-wide estimate of resuspension that could be compared with estimates using the next two methods.

A.1.2 The TIPM/TPM method

The second method was proposed by Weyhenmeyer et al. (Bloesch 1994), whereby any resuspended sediment is assumed to contain a constant fraction of inorganic matter. From linear regression of TIPM and TPM, therefore, the ordinate intercept can be taken as the fraction of autochthonous phytoplankton that settles into a trap over any single deployment period. This level of autochthonous phytoplankton production is assumed to be constant irrespective of the amount of resuspension (1995). The remaining fraction of TPM that isn’t attributed to autochthonous production and sedimentation of phytoplankton is taken to be resuspended sediment (equation A.1.2).
The TIPM/TPM method only utilizes characteristics of trap contents, making it independent of the local sediment conditions. Consequently, if only a distinct fraction of sediment is resuspended, and provided that this fraction contains a constant concentration of inorganic matter, the method should be valid. However, it requires that external input of particles and productivity by diatoms in the lake are both negligible as these could affect the ratio of trapped TIPM and TPM (Weyhenmeyer et al. 1995). Further, achieving adequate spatial resolution is restricted by the large number of traps required, at each site, to get a strong regression between TIPM and TPM. Consequently, this method is usually used to estimate lake-wide resuspension (Weyhenmeyer et al. 1995). All regressions performed for sediment traps deployed in Lake Alexandrina were undertaken with the Prism 5.0c software package (GraphPad Software, Inc.).

A.1.3 Correcting with long-term accumulation rates

A third method is commonly used in the literature (Weyhenmeyer et al. 1995; Weyhenmeyer 1997; Effler & Matthews 2004) without being explicitly stated as a method for proportioning sediments caught in traps to resuspension. In effect, long-term sedimentation rates, as determined by deep sediment cores, are subtracted from the gross sedimentation (TPM) measured with traps. This assumes negligible spatial variability in primary sedimentation and sediment resuspension. It is also dependent on the location of the cores used for determining the sedimentation rate and whether these were affected by sediment focusing (e.g. Douglas & Rippey 2000). Furthermore, it is assumed that primary sedimentation (settling of newly formed particles) is the sole contributor to sediment accumulation and that resuspension is constant over long time periods. However, this method is less sensitive to issues of carbon mineralization, grazing and physical artefacts involved in using traps as these
change the amount of gross sedimentation, rather than the composition of trapped particles as for the previous two methods (Hilton 1985; Blais & Kalff 1995).

The estimates of long-term sedimentation rates in Lake Alexandrina have been estimated by Barnett (1994) and Herczeg et al. (2001). Both studies took cores from the main basin of Lake Alexandrina, but sedimentation rates differed by 7.5 g m$^{-2}$ d$^{-1}$. Barnett (1994) took a single core from the southern side of Lake Alexandrina, and showed a sedimentation rate of 1.73 g m$^{-2}$ d$^{-1}$, whereas Herczeg et al. (2001) collected three cores from the northern side of the lake and estimated sedimentation to occur at 9.2 g m$^{-2}$ d$^{-1}$. This gave a range from the two studies, of 1.3 – 9.2 g m$^{-2}$ d$^{-1}$, which was used to estimate resuspension from the gross sedimentation values reported during the sediment trap study of Lake Alexandrina (Chapter 5).

_A.1.4 Method comparison and optimization for Lake Alexandrina_

Sensitivity analysis comparing the first two methods outlined above has suggested that the TIPM/TPM method varies less than the label method from errors in trapped organic matter concentrations, whereas the label method was more stable to seasonal fluctuations in primary productivity (2001). Consequently, it was suggested that the label method be used preferentially in lakes with high cyanobacteria biomass, while the TIPM/TPM method be used when bacterial activity is high (Horppila & Nurminen 2005). This would suggest that the label method is more appropriate for Lake Alexandrina. A comparison of resuspension rates estimated in Lake Alexandrina using the label and the TIPM/TPM methods is shown in Figure A.1.1.

Estimates of sediment resuspension in Lake Alexandrina were generally lowest for the label method and highest when correcting with long-term accumulation rates (Table A.1.1). Conversely, estimates of primary sedimentation are higher with the label method (up to 543 g m$^{-2}$ d$^{-1}$) than with the TIPM/TPM method (up to 122 g m$^{-2}$ d$^{-1}$) and when correcting with long-term accumulation rates (5 – 6 g m$^{-2}$ d$^{-1}$). The
discrepancy between methods to calculate the resuspended fraction of TPM may be explained by the assumptions required for each. The artificially quiescent environment created inside the sediment trap can selectively trap cohesive organic material, enhancing the amount of autochthonous production and sedimentation estimated with both trap-specific correction methods (Horppila & Nurminen 2005).

In shallow lakes with high turbulence, resuspension events can be so frequent as to prevent the complete sedimentation of autochthonously produced material (Chapter 7). This would further enhance the amount of TPM attributed to primary sedimentation when using the label and TIPM/TPM methods.

The correction for long-term accumulation rates is used in Chapter 5 for a number of reasons. First, Lake Alexandrina is a large, shallow lake with frequent sediment resuspension. Offshore water currents in Lake Alexandrina have been measured above 17 cm s\(^{-1}\) during relatively calm conditions (Chapter 6) suggesting that particles could be transported over 600 m in an hour. This renders the label method insufficient, as it is reliant on the assumption of negligible horizontal transport. The TIPM/TPM method has been shown to fluctuate most as a result of variations in primary productivity (Kozerski 1994). The phytoplankton in Lake Alexandrina are dominated by cyanobacteria (> 98% of cell counts), suggesting that the label method is less prone to errors, according to Horppila and Nurminen (2005). Furthermore, that sedimentation in Lake Alexandrina has mostly increased from the long-term rate of 1.7 – 9.2 g m\(^{-2}\) d\(^{-1}\) to between 3 and 120 g m\(^{-2}\) d\(^{-1}\) using the TIPM/TPM method, despite a substantial decline in water depth and consequent increase in resuspension, seems unlikely. The primary sedimentation rates using the TIPM/TPM method are also high compared with similar primary sedimentation rates taken from the literature. In a survey of 25 lakes in Sweden, Håkanson (2005) reported a maximum primary sedimentation rate of 6.8 g m\(^{-2}\) d\(^{-1}\). During a bloom of Chlorobium phaeobacteroides in Lake Kinneret, Israel, Yacobi and Ostrovsky (1995) reported a maximum rate of organic matter sedimentation of 2.42 g m\(^{-2}\) d\(^{-1}\). In contrast, when
correcting for long-term accumulation rates, the assumption that sediments are not significantly focused is consistent with the theory about shallow lakes. The spatial variation of long-term accumulation rates in Lake Alexandrina (i.e. the difference between estimates by Barnett (1994) and Herczeg et al. (2001)) is small relative to the temporal variation in sedimentation shown in table A.1. This suggests some level of homogeneity in the long-term sediment record. As a result of these factors, the third method (correcting TPM with the long-term sediment accumulation rates) is used in Chapter 5 to estimate the amount of sedimentation that is attributable to resuspended sediments.
Table A.1.1: Estimates of the proportion of TPM attributed to sediment resuspension and autochthonous sedimentation using the three calculation methods.

<table>
<thead>
<tr>
<th>Date</th>
<th>Gross Sedimentation (g m(^{-2}) d(^{-1}))</th>
<th>Resuspension (%)</th>
<th>Primary Sedimentation (g m(^{-2}) d(^{-1}))</th>
<th>Resuspension (%)</th>
<th>Primary Sedimentation (g m(^{-2}) d(^{-1}))</th>
<th>Resuspension (%)</th>
<th>Primary Sedimentation (g m(^{-2}) d(^{-1}))</th>
<th>Accumulation rate correction(^1)</th>
</tr>
</thead>
<tbody>
<tr>
<td>13 Nov 09</td>
<td>918</td>
<td>74.1</td>
<td>238</td>
<td>96.5</td>
<td>32</td>
<td>&gt;99</td>
<td>2.2 – 8.7</td>
<td></td>
</tr>
<tr>
<td>24 Nov 09</td>
<td>920</td>
<td>55.7</td>
<td>408</td>
<td>95.1</td>
<td>45</td>
<td>&gt;99</td>
<td>2.2 – 8.7</td>
<td></td>
</tr>
<tr>
<td>18 Dec 09</td>
<td>1670</td>
<td>76.6</td>
<td>391</td>
<td>99.8</td>
<td>3</td>
<td>&gt;99</td>
<td>3.2 – 9.7</td>
<td></td>
</tr>
<tr>
<td>29 Dec 09</td>
<td>933</td>
<td>42.5</td>
<td>537</td>
<td>93.4</td>
<td>62</td>
<td>&gt;99</td>
<td>2.2 – 8.7</td>
<td></td>
</tr>
<tr>
<td>15 Jan 10</td>
<td>1000</td>
<td>45.7</td>
<td>543</td>
<td>99.1</td>
<td>9</td>
<td>&gt;99</td>
<td>3.2 – 9.7</td>
<td></td>
</tr>
<tr>
<td>10 Feb 10</td>
<td>1001</td>
<td>54.7</td>
<td>454</td>
<td>91.7</td>
<td>83</td>
<td>&gt;99</td>
<td>3.2 – 9.7</td>
<td></td>
</tr>
<tr>
<td>4 Mar 10</td>
<td>1076</td>
<td>60.9</td>
<td>421</td>
<td>99.2</td>
<td>9</td>
<td>&gt;99</td>
<td>2.2 – 8.7</td>
<td></td>
</tr>
<tr>
<td>10 Mar 10</td>
<td>1631</td>
<td>60.1</td>
<td>652</td>
<td>92.5</td>
<td>122</td>
<td>&gt;99</td>
<td>2.2 – 8.7</td>
<td></td>
</tr>
<tr>
<td>20 Mar 10</td>
<td>513</td>
<td>41.1</td>
<td>302</td>
<td>95.1</td>
<td>25</td>
<td>99</td>
<td>2.2 – 8.7</td>
<td></td>
</tr>
<tr>
<td>24 Mar 10</td>
<td>781</td>
<td>50.1</td>
<td>389</td>
<td>96.8</td>
<td>25</td>
<td>&gt;99</td>
<td>3.2 – 9.7</td>
<td></td>
</tr>
</tbody>
</table>

\(^1\) Accumulation rate taken as 1.7 – 9.2 g m\(^{-2}\) d\(^{-1}\), which is the average sedimentation calculated by Barnett (1994) and Herczeg et al. (2001).
Figure A.1.1: A comparison between two methods to calculate the proportion of TPM attributable to sediment resuspension in Lake Alexandrina ($r^2 = 0.91$).
A.2. The effect of salinity on flocculation and sedimentation rates

The rate of sedimentation under different salinity regimes is expected to change as salinity induced flocculation of cohesive particles alters the effective size and density of particles. The simple experiment described here gives an indication of the settling velocity and size of flocs under quiescent conditions.

A.2.1 The SETCOL technique

A standard concentration of fine sediments collected from the thalweg at the entrance to Lake Alexandrina was suspended in water of different salinities. Water with different salinities were prepared by dissolving 0 g L\(^{-1}\), 1 g L\(^{-1}\), 2 g L\(^{-1}\) and 3 g L\(^{-1}\) of Instant Ocean ® salt in deionised water at 25°C. The ionic composition of Instant Ocean ® is shown in Table A.2.1. For each salinity treatment, sediments were added to give a suspended particulate matter (SPM) concentration of 500 mg L\(^{-1}\) and the mixture was added into a settling column which conformed to specifications given in Bienfang (2001). The column had a height of 0.47 metres, a volume of 2.81 L and fitted with three off-take points for the topmost 0.2 L, the middle 2.4 L and the bottom 0.21 L. Sediments were suspended uniformly and harvested (the entire column was emptied) in duplicate at 5 time points (t = 0, 0.5, 1, 2, 4 hours).

Harvested samples, which had SPM concentrations proportional to the settling velocity, were analysed for SPM, suspended inorganic particulate matter (SIPM) and
suspended organic particulate matter (SPOM) gravimetrically. The mean settling velocity \( (\Psi) \) of sediments was calculated with equation A.2.1

\[
\Psi = \frac{V_s b_s - V_s (b_{0,0} + b_{0,t})/2}{V_t (b_{0,0} + b_{0,t})/2} \left( \frac{l}{t} \right)
\]  

(A.2.1)

where \( V_s \) is the volume of the harvested sample, \( b_s \) is the SPM concentration after the allotted time period \( t \), \( l \) is the height of the column, \( b_{0,0} \) is whole column SPM at the beginning of the trial, \( b_{0,t} \) is the whole column SPM at the end of the trial and \( V_t \) is the total volume of the column (1981).

A.2.2 Settling velocity and floc size

The settling velocity of particles changed significantly under different salinity treatments (Figure A.2.1), fitting well with other studies (Bienfang 1981). The loss rates were best fitted to equation A.2.2 for exponential decay of the form

\[
Y = (Y_0 - P)e^{-K \times t} + P
\]  

(A.2.2)

where \( Y_0 \) is the initial \( Y \) value of SPM concentration (mg L\(^{-1}\)) at time zero, \( P \) is the asymptote and \( K \) is the rate constant. Values for each salinity treatment are shown in Table A.2.2. Calculated settling velocities are depicted in Figure A.2.2 for sediments accumulating in the bottom section of the column. Maximum settling velocity for salinity treatments is shown in Table A.2.3. These settling velocities were used to estimate the size of flocs using Stoke’s law (Burban et al. 1990; Spears et al. 2008) assuming that the floc density was equivalent to the density of the individual particles. Mean particle size of suspended sediments was 39 µm before settling, whereas this apparent size increased to 603 µm for the highest (3 g L\(^{-1}\)) salinity treatment over the first time period.
Table A.2.1: Composition of salt in Instant Ocean® and at three sites along the salinity gradient of Lake Alexandrina. Data are from October 2009. Note that the concentration of salts have been standardized to the sodium concentration in Instant Ocean® to enable comparison. NR indicates that this parameter was not measured in water samples.

<table>
<thead>
<tr>
<th></th>
<th>Instant Ocean®</th>
<th>Goolwa</th>
<th>Lake Alexandrina</th>
<th>Wellington</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>mg L⁻¹</td>
<td>mg L⁻¹</td>
<td>mg L⁻¹</td>
<td>mg L⁻¹</td>
</tr>
<tr>
<td>Na⁺</td>
<td>10.780</td>
<td>10.780</td>
<td>10.780</td>
<td>10.780</td>
</tr>
<tr>
<td>K⁺</td>
<td>0.420</td>
<td>0.425</td>
<td>0.394</td>
<td>0.439</td>
</tr>
<tr>
<td>Mg²⁺</td>
<td>0.132</td>
<td>1.439</td>
<td>1.464</td>
<td>1.710</td>
</tr>
<tr>
<td>Ca²⁺</td>
<td>0.400</td>
<td>0.822</td>
<td>0.865</td>
<td>1.829</td>
</tr>
<tr>
<td>Sr²⁺</td>
<td>0.009</td>
<td>0.011</td>
<td>0.015</td>
<td>0.023</td>
</tr>
<tr>
<td>Cl⁻</td>
<td>19.290</td>
<td>21.094</td>
<td>21.080</td>
<td>17.174</td>
</tr>
<tr>
<td>SO₄²⁻</td>
<td>0.266</td>
<td>3.916</td>
<td>2.979</td>
<td>3.256</td>
</tr>
<tr>
<td>HCO₃⁻</td>
<td>0.200</td>
<td>0.985</td>
<td>2.840</td>
<td>8.475</td>
</tr>
<tr>
<td>Br⁻</td>
<td>0.056</td>
<td>NR</td>
<td>NR</td>
<td>NR</td>
</tr>
<tr>
<td>F⁻</td>
<td>0.001</td>
<td>0.003</td>
<td>0.006</td>
<td>0.013</td>
</tr>
</tbody>
</table>
Table A.2.2: Values of coefficients and fit for equation A.2.2 under different salinities.

<table>
<thead>
<tr>
<th></th>
<th>$Y_{0}$</th>
<th>$P$</th>
<th>$K$</th>
<th>$r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>$Y$ at $t = 0$ hrs</td>
<td>asymptote</td>
<td>$hr^{-1}$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>$0 \text{ g L}^{-1}$</td>
<td>448</td>
<td>277</td>
<td>1.1</td>
<td>0.90</td>
</tr>
<tr>
<td>$1 \text{ g L}^{-1}$</td>
<td>472</td>
<td>136</td>
<td>1.5</td>
<td>0.96</td>
</tr>
<tr>
<td>$2 \text{ g L}^{-1}$</td>
<td>471</td>
<td>68</td>
<td>2.4</td>
<td>0.99</td>
</tr>
<tr>
<td>$3 \text{ g L}^{-1}$</td>
<td>489</td>
<td>72</td>
<td>2.7</td>
<td>0.98</td>
</tr>
</tbody>
</table>
Table A.2.3: Settling velocity and calculated floc radius under different salinity regimes. Data shown are for the first time period (0.5 hours) where the largest particles settled out of suspension.

<table>
<thead>
<tr>
<th>Salinity (g L(^{-1}))</th>
<th>Settling velocity (m hr(^{-1}))</th>
<th>Calculated floc radius ((\mu m))</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 g L(^{-1})</td>
<td>12</td>
<td>147</td>
</tr>
<tr>
<td>1 g L(^{-1})</td>
<td>67</td>
<td>348</td>
</tr>
<tr>
<td>2 g L(^{-1})</td>
<td>103</td>
<td>434</td>
</tr>
<tr>
<td>3 g L(^{-1})</td>
<td>200</td>
<td>603</td>
</tr>
</tbody>
</table>
Figure A.2.1: Loss rate of SPM from the top section of the settling column under different salinities.
Figure A.2.2: Settling velocity for particles under different salinity regimes at different samples harvested from the settling columns. As SPM settled, the settling velocity of the remaining particles decreased with time.
References


References


References


Russell, B. D., J. I. Thompson, L. J. Falkenberg and S. D. Connell (2009). Synergistic effects of climate change and local stressors: CO₂ and nutrient-


Skinner, D. S. (2007). A Culture for Change: Reinterpreting the Environmental Crisis as a Cultural Inconsistency. Flinders University of South Australia,


